LOW-ENERGY WASTEWATER RECYCLING THROUGH WETLAND ECOSYSTEMS:

Apalachicola Study—Experimental Use of a Freshwater Shrub Swamp

Third and Final Summary Progress Report
Covering the Period from 1985 to 1987

By

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1989

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PREFACE

A research proposal was submitted to the Florida Department of Environmental Regulation in 1981 to assess the response and assimilative capacity of a titi shrub swamp ecosystem to addition of secondarily treated wastewater effluent from the City of Apalachicola, Florida. The long-term goal of the project was to assess the role of this type of wetland for wastewater treatment. After funding began in November 1981, research continued until termination of the project in February 1988. The first summary progress report (Best *et al.* 1983) covered the period from November 1981 to March 1983, encompassing an 8-mo hiatus in funding during which sampling and analyses continued. This report summarized research focused on establishing basic information on the chemical, biological, and hydrological status of the shrub swamp prior to wastewater discharge. The second summary report (Best *et al.* 1987) covered the period from July 1983 to November 1984 and summarized continued development of pre-application baseline data along with field and laboratory experiments demonstrating the wastewater treatment potential of the swamp. This third and final report summarizes post-application sampling and analyses covering the period from March 1985 to May 1986 and from January 1987 to December 1987. Although laboratory analysis continued, sampling suffered a 7-mo break due to another hiatus in funding. No monitoring of the wastewater-impacted shrub swamp has been done since the end of the research project, although wastewater discharge has continued up to the present.

APALACHICOLA PROJECT RELATED PUBLICATIONS
(As of 1987)

Best, G.R., L.N. Schwartz, L. Sonnenburg, S. Kidd, and J.J. McCreary. 1983. Low-Energy Wastewater Recycling through Wetland Ecosystems: Experimental Use of a Freshwater Swamp. Center for Wetlands Tech. Report No. 39. University of Florida, Gainesville, Florida.

Haack, S.K. 1984. Aquatic Macroinvertebrate Community Structure in a Forested Wetland: Interrelationships with Environmental Parameters. M.S. Thesis. University of Florida, Gainesville, Florida.

Haack, S.K., G.R. Best, and T.L. Crisman. 1987. Changes in Aquatic Macroinvertebrate Community Structure along Environmental Gradients within a Titi Shrub Swamp in Northern Florida. Symposium on Freshwater Wetlands and Wildlife: Perspectives on Natural, Managed and Degraded Ecosystems. Charleston, South Carolina. March 26, 1986.

Pezeshki, C. 1987. Response of Benthic Macroinvertebrates of a Shrub Swamp to Discharge of Treated Wastewater. MS Thesis (CFW-87-07). Gainesville, FL: Univ of FL, pp. 115.

Schwartz, L.N., and G.R. Best. 1984. Pre-Wastewater Disposal Characterization and Demonstration of Treatment Potential of a Titi Shrub Swamp in Apalachicola, Florida. Submitted to Wetlands.

Sonnenberg, L.B. 1984. Validation of a Fugacity Based Environmental Fate Model using 2,4 Dichlorophenol in Laboratory Microcosms. M.S. Thesis. University of Florida, Gainesville, Florida. pp. 109.

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Haack, S.K. 1984. Aquatic Macroinvertebrate Community Structure in a Forested Wetland: Interrelationships with Environmental Parameters. Center for Wetlands Seminar. University of Florida, Gainesville, Florida. February 8, 1984.

Haack, S.K., G.R. Best, and T.L. Crisman. 1986. Changes in Aquatic Macroinvertebrate Community Structure along Environmental Gradients within a Titi Shrub Swamp in Northern Florida. Symposium on Freshwater Wetlands and Wildlfe: Perspectives on Natural, Managed and Degraded Ecosystems. Charleston, South Carolina. March 26, 1986.

Schwartz, L.N. 1984. Apalachicola Project Seminar: Phase of the Natural Wetland Wastewater Treatment Project. Florida Department of Environmental Regulation, Tallahassee, Florida. February 28, 1984.

Schwartz, L.N. 1984. Experimental use of a Titi Shrub Swamp at Apalachicola, Florida, for Low-Energy Recycling of Wastewater: Ecological Setting and Development of Emperical Models. Center for Wetlands Seminar, University of Florida, Gainesville, Florida. April 11, 1984.

Schwartz, L.N. 1984. Pre-Wastewater Disposal Characterization and Demonstration of Treatment Potential of a Titi Shrub Swamp in Apalachicola, Florida. Wetlands Seminar, Greater Orlando Chamber of Commerce. November 14, 1984.

Schwartz, L.N., G.R. Best, S.K. Haack, and L.B. Sonnenberg. 1984. Pre-Wastewater Disposal Characterization of a Titi Shrub Swamp in Apalachicola, Florida. Poster Session: Wetland Ecology. Co-Sponsored by the ESA and BSA at the 35th Annual Meeting with the AIBS. Fort Collins, Colorado: Colorado State University. August 9, 1984.

Schwartz, L.N., and G.R. Best. 1985. Pre-Wastewater Disposal Characterization and Demonstration of Treatment Potential of a Titi Shrub Swamp in Apalachicola, Florida. Annual Meeting, Society of Wetland Scientists. Durham, New Hampshire. July 30, 1985.

OVERVIEW

Purpose

Section 17-4.243(4) of the Florida Administrative Code provides for an exemption from water quality criteria to allow for the experimental use of wetlands for low-energy water and wastewater recycling. The goal of this exemption is to encourage experiments leading to new information regarding low-energy approaches to advanced treatment of domestic, agricultural, and industrial wastes and to encourage conservation of wetlands and fresh waters. A permit may be issued specifically exempting certain sources of pollution discharging into restricted areas of wetlands from water quality criteria, provided that:

1) scientifically valid experimental controls are provided by the applicant to monitor the long-term ecological effects and waste recycling efficiency, and

2) the discharger affirmatively demonstrates that the wetland ecosystem may reasonably be expected to assimilate the waste discharge without significant adverse impact on the biological community within the receiving waters.

A task force was established to assist the City of Apalachicola, Florida, in addressing water quality and public health issues related to Apalachicola Bay. State and federal agencies determined that the discharge of municipal wastewater to a tributary of the Apalachicola River had to be eliminated. Based on site-specific studies and analysis of other available disposal alternatives, the most appropriate and cost-effective alternative was to discharge the municipal wastewater to a titi shrub swamp. The above exemption from water quality standards was granted to the City of Apalachicola, Florida. Research was initiated by the Center for Wetlands, University of Florida, to insure compliance with the provisions of the exemption and to demonstrate an assimilative capacity of the wetland ecosystem.

Since wetland systems do not normally meet the Class III water quality standards for pH and dissolved oxygen, it is the practice of the Department of Environmental Regulation to require an assessment of baseline conditions on these and other chemical, physical, and biological parameters. These background water quality conditions become the criteria from which to evaluate compliance with provisions of the exemption. After wastewater discharge begins, monitoring of the significant chemical, physical, and biological characteristics of the wetland system receiving the discharge, as well as a non-impacted control system, is required.

Although there is evidence that wetlands are effective in tertiary treatment of wastewater, their long-term response to wastewater discharge is unclear. Since the predictive capability for assessing the long-term assimilative capacity of these wetlands is limited, a model was developed to evaluate ecosystem response to the increased influx of nutrients and water and to predict capacity and effective lifetime of wastewater assimilation within the wetland. Thus primary objectives of this research project were to determine if a titi shrub swamp can effectively remove nutrients and pathogens from secondarily treated municipal wastewater and to assess the long-term assimilative capacity and response of the wetland to wastewater addition. To achieve this goal, research was organized into discrete, definable tasks as follows:

1) Chemical analysis of water quality—characterize and monitor chemical parameters within the wetland to determine response to wastewater discharge;

2) Response of biota—characterize and monitor biological parameters within the wetland to determine long-term response to wastewater discharge;

3) Hydrology—predict transport of wastewater within the system;

4) Variable advancing front—indicate nutrient treatment potential of wetland; and

5) Ecosystem response model—simulate long-term effects of wastewater discharge to the wetland.

Research Objectives

Research objectives during the first two funding periods, or phases, consisted of two primary tasks. The first was development and summary of pre-application baseline data on chemical water quality and biological parameters in the Apalachicola titi shrub swamp. A total of 15 mo of baseline data was collected, including values for surface water, groundwater, benthic invertebrates, vegetation, and bacteria. Secondly, the wastewater treatment potential of the swamp was assessed. Three experiments were conducted to complete this task. The fate of nitrate added to laboratory microcosms consisting of sediment cores and water taken from the wetland was determined. Phosphorus adsorption maxima determined from phosphorus adsorption isotherms were used to determine potential for phosphorus adsorption in wetland soils. A pilot field mesocosm flow-through experiment was also conducted to assess the fate of phosphorus and nitrogen added at secondarily treated wastewater levels. The primary task of the third and final phase of funding was to monitor chemical and biological parameters as listed above, in order to assess the response and assimilative capacity of the wetland to added wastewater. Discharge from the wastewater treatment plant began in May 1985. In this report, monitoring data from 2 mo prior to discharge until December 1987 will be presented and discussed. For the sake of comparison, the baseline data previously reported (Best *et al.* 1983, 1987) will be presented and reviewed again. In addition, surface discharge measurement data, facilitated by a USGS water level recorder and flow meter, will be combined with water quality data to give an estimation of water and nutrient budgets in the wastewater-impacted swamp. Finally, this report will evaluate the variable advancing front concept under direct discharge conditions and synthesize all available information into a predictive model simulating the long-term effects of wastewater discharge to the wetland.

Overview of Region and Study Site

Geology and Physiography of the Area

The City of Apalachicola is located at the western edge of the Big Bend region of the state (Figure 1). The entire Big Bend region of Florida is underlain by a bedrock of limestone, which dates back no later than the early Miocene age (Clewell 1971). The first limestone encountered beneath the Apalachicola area is located approximately 40 m below the surface (Schmidt 1978). Above the limestone lies an assortment of various Miocene clastics and above them a veneer of Pleistocene sands. These materials were deposited during ancient sea level fluctuations. Usually there is a shell bed in a sand and clayey matrix, overlain by a gravel and coarse sand unit, overlain by a clayey sand and finally by a medium fine sand composed of sand, silt and clay, and organic debris. In addition to peat deposits there are beds of humate along the coast (up to 1 m thick). The humate is dark brown to black firmly cemented sand of late Pleistocene to Recent age and was probably formed in an ancient swamp when sea level was a few feet higher than the present.

The western portion of the Big Bend region lies in the Apalachicola Coastal Lowlands unit of the Gulf Coastal Lowlands Physiographic Province (Schmidt 1978). These coastal lowlands are low in elevation due to the reworking by coastal processes and are generally poorly drained. A large amount of land area in this unit is covered by swamp. The impermeable clastics contain a large amount of fine grained clay and silt (as indicated above) which retards water movement. The permeability is low and groundwater is perched near the surface. This is enhanced by low relief, making it difficult for surface water to run off the land surface.

The swamps occupy irregularly shaped shallow depressions that mostly do not join to form drainages (Clewell 1971) . These depressions are likely the result of gentle undulations of a former Pleistocene sea-bottom. These swamps may have been accentuated by more recent localized slumping of the surface that would slowly form a depression having a higher water table than the surrounding lands. These geomorphic features are interlevee swamps, oriented parallel to the coast indicating their formation through marine forces (Schmidt 1978).

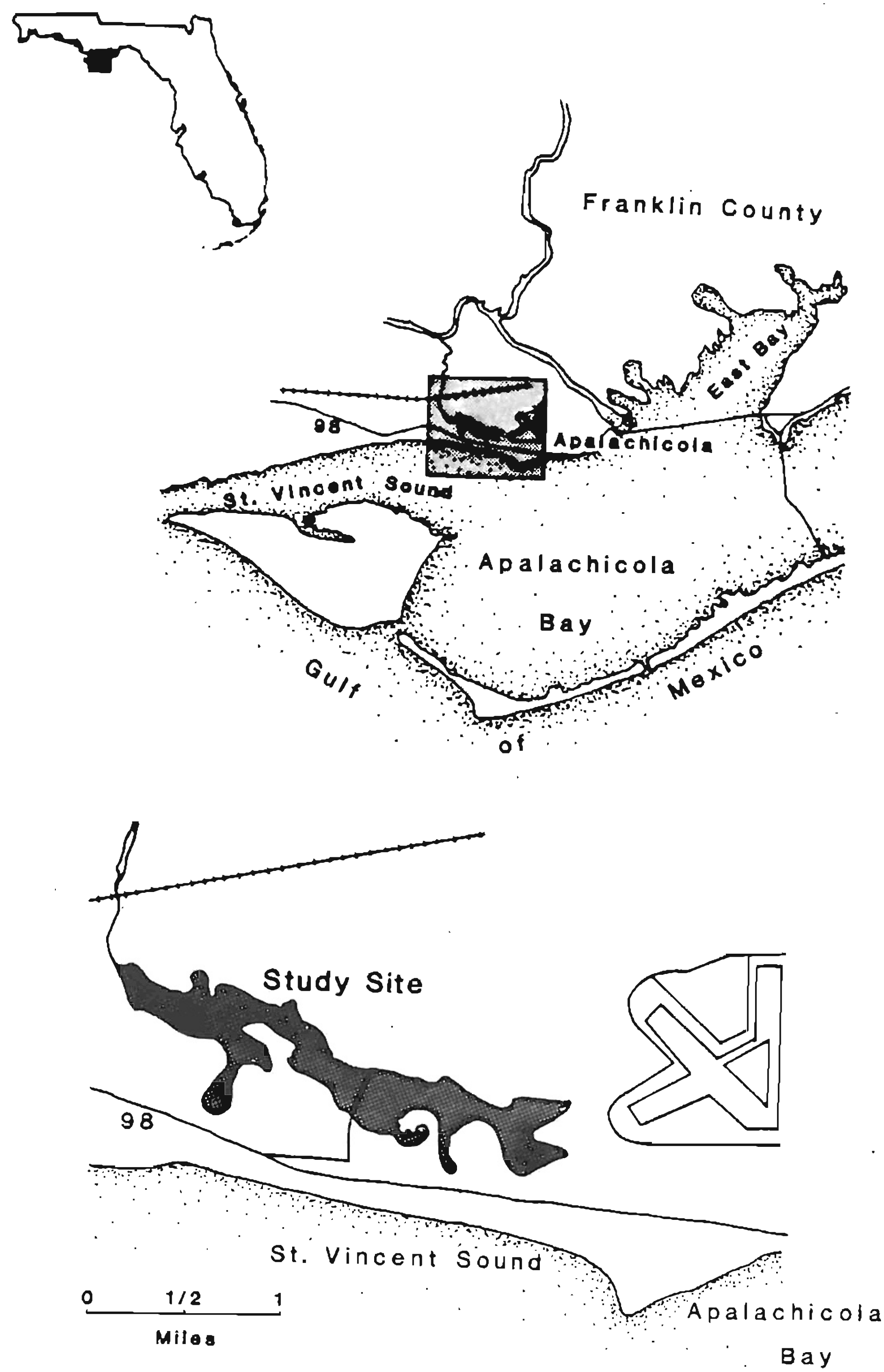


Figure 1. Location of the titi shrub swamp study site in Apalacicola, Florida.

These type of systems are referred to as bog and bog-fed streams by Wharton *et al.* (1982). The depressions that feed the streams are areas of internal perched drainage underlain by clay aquicludes. Surface drainage occurs through slow moving streams originating from flat swamp areas. These streams have limited distribution and generally occupy the linear depressions or swales between adjacent sand ridges and reworked relict coastal lowland deposits.

Study Site Description

The research site (Figure 2) is a titi shrub swamp wetland located 6 km west of Apalachicola, Florida, in the panhandle region of the state. The wetland is approximately 3 km in length and varies from 300 to 800 m in width and is located 500 to 1000 m inland and parallels the coast of the Gulf of Mexico. The wetland covers 63.7 ha and drains an area of 2.26 x 10^6 m^2 (Best *et al.* 1983). The outlet of the wetland is Whortleberry Creek, a tributary of the Apalachicola River. The wetland itself occupies a dune swale in an area planted in *Pinus elliottii* (slash pine). Mineral soils underlie the wetland: average bulk density is 0.56 g/cm^3 and average organic carbon is 5.5%. Water flow runs parallel to the coast from east to west. The wetland is dominated by *Cyrilla racemiflora* (red titi) and *Cliftonia monophylla* (black titi) in shallower areas (0 to 40 cm water) and by *Taxodium distichum* var. *nutans* (pond cypress) and *Nyssa sylvatica* var. *biflora* (swamp black gum) in deeper areas (20 to 50 cm).

The activated sludge wastewater treatment plant became operational in May 1985 and discharged an average of 3.42 x 10^6 l/day (0.9 MGD) of secondary effluent during the study period. After dechlorination in a holding pond, wastewater enters the wetland at its approximate midpoint via a submerged, slotted pipe, yielding an effective treatment area of about 32 ha. The hydroperiod, or time of inundation with water, varied from 8 to 10 mo prior to wastewater discharge, increasing to almost continuous inundation in some areas after discharge began. The original scientific design of the experiment, which called for an elevated, perforated, pipeline, was compromised by the submerged concrete outflow pipe housing. During construction the area immediately surrounding the point of wastewater discharge as, cleared of vegetation and dredged, causing water to pond at the site. In addition, large "paths" up to 50 yd across, apparently during surveying of the site, were cut and bulldozed approximately perpendicular to the concrete outflow pipe housing. Although the impacts were caused during construction, we made an attempt correct this situation, for the long term, by planting trees (as will be discussed in the *Vegetation* section of this report).

The original naming of the sampling sites (see Best et al., 1983) of 1, 2, 3, 4, 5, 6, and 7 (Figure 2) was changed to reflect distance from the wastewater outfall. Several additional sites downstream of the outfall were sampled after wastewater flow began. Three upstream control sites were sampled monthly during the 1983-84 sampling period and quarterly during the 1985-87 sampling period. Site U1200 (U = upstream and discharge, 1200 meters) is the farthest upstream of the effluent outfall, and corresponds with site 1 of the previous naming. Site U400 is 400 m upstream of the effluent outfall. Site U25,

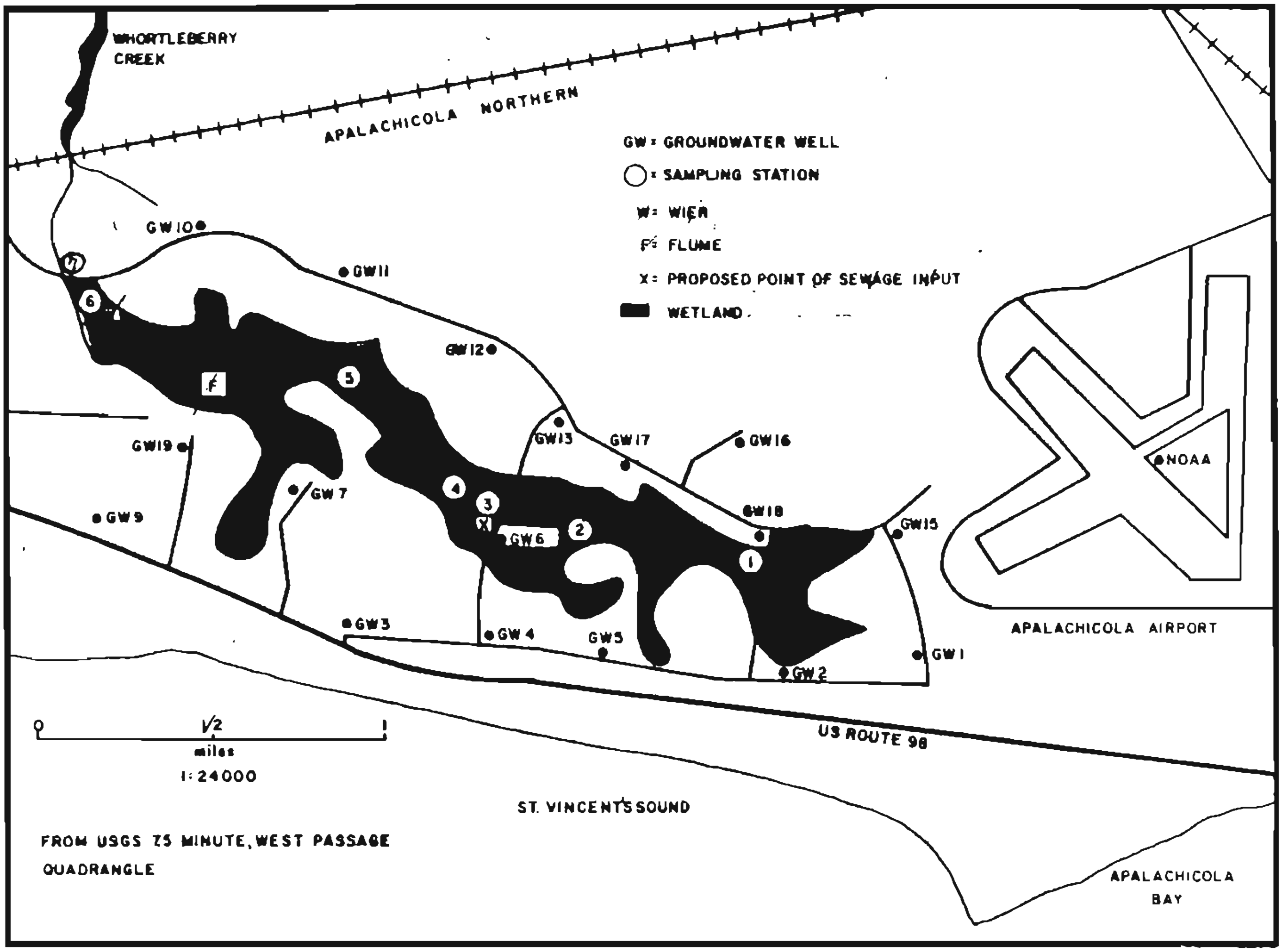


Figure 2. The titi shrub swamp study site in Apalachicola, Florida, surface water and groundwater sampling stations. Listed below is a summary of sampling site designation prior and after to contruction of outfall pipe.

Original Site Number	Current Site Number
1	U1200
2	U400
3	U25
4	D25
-	D200
-	D400
5	D900
6	D2600

corresponding with site 3, is just upstream of the wastewater discharge (25 m) and also served as a control, although, due to its close proximity to the wastewater outfall site, may have been contaminated with wastewater. Site D25, corresponding with site 4, is 25 m downstream of the wastewater outfall. Sites D200 and D400 are 200 and 400 m, respectively, downstream from the wastewater outfall. These sites were not previously sampled. Site D900 is 900 m downstream of the outfall and corresponds to site 5. These downstream sites were sampled monthly after wastewater discharge began. Site D2600 is at the point of discharge of the wetland into Whortleberry Creek and corresponds to site 6.

The following site descriptions are from Haack (1984:17-20):

> Site 1 (U1200) is an open water site, continuously wet throughout the year. Vegetation at sampling points is dominated by *Hypericum* sp. and *Utricularia* sp. but *Lacnanthes tinctoria* and *Xyris* sp. were also collected in some samples. The site grades to *Taxodium distichum* var. *nutans* and *Nyssa sylvatica* var. *biflora* to the south and east, pine and grassland to the north, and *Cyrilla racemiflora* and *Lyonia lucida* to the west. The site is somewhat ponded due to a road berm to the west, but water is usually observed flowing across the road at several points. The substrate is primarily sand with a thin (1-4 cm) layer of well-decomposed detritus and algal mat.
>
> Sites 2 (U400) and 5 (D900) are intermediate between a *Taxodium* sp.–*Nyssa* sp. deepwater swamp and a *Cyrilla* sp.–*Cliftonia monophylla* titi shrub swamp. Trees are larger and the water is deeper. Aquatic macrophytes are sparse. The substrate is a layer of leaf litter and flocculent particles approximately 5-10 cm deep overlaying a brown to blackish sediment and there is a fine, tough, intertwined root system at 3 cm depth and below.
>
> Prior to wastewater discharge, sites 4 (D25), D200, and D400 were characterized as *Sphagnum* sp.–shrub sites. Water levels were low during most of the year, and occasionally below ground level. *Sphagnum* sp. was abundant in the water column, while *Hypericum* sp. and *Lacnanthes* sp. dominated as emergents. Larger mixed shrubs (*Lyonia* sp., *Cyrilla* sp., *Cliftonia* sp.) were also present but sampling points were relatively open. The substrate was a fine mud with a 2-5 cm litter layer grading to sand at depths which varied greatly over short (30-50 cm) distances.
>
> Site 6 (D2600)is most like a stream channel. Flowing water can usually be observed and a distinct channel is recognizable. The channel contains *Xyris* sp., *Sagittaria* spp., and unidentified sedges. The banks are dominated by *Hypericum* sp., *Xyris* sp., *Sagittaria* sp., and *Ludwigia repens*. The sediment is extremely flocculent and muddy. Leaf litter collects near the banks, but is eroded from the main channel. Larger shrubs and trees typical of the wetland surround the site.

Locations of the sampling stations, shallow groundwater wells, and point of wastewater input (Figure 2) were selected to include areas that could serve as controls during plant operation. Many parameters were sampled on regular monthly intervals and others were sampled on a quarterly sampling scheme centered around a seasonal schedule as follows:

First, dormant season—December, January, and February;
Second, early growing season—March, April, and May;
Third, growing season—June, July, and August;
Fourth, senescent season—September, October, and November.

WATER CHEMISTRY

Purpose

Water quality data were collected to establish a baseline of normal chemical characteristics of this wetland ecosystem and to evaluate the changes that occurred in the measured parameters as a result of wastewater addition. Establishment of natural ranges for each parameter allowed an accurate assessment of the chemical responses of the wetland. In addition, baseline data were necessary in order to model chemical flux through the system. Samples from both surface water and groundwater were collected and analyzed. Monitoring of groundwater chemistry both before and after effluent discharge indicated the wetland's ability to contain added nutrients within the ecosystem (nutrient sink), as opposed to allowing nutrients to pass into groundwater (nutrient exporter).

Literature Review

Wastewater Discharge to Wetlands

Wetlands are often viewed as highly dynamic and adaptable ecosystems. Nutrient transformation processes may enable some wetlands to assimilate and store increased levels of nutrients and other contaminants from wastewater (USEPA 1983). Many wetlands have been shown to efficiently process wastewater (Whigham 1982), tolerating anoxic conditions associated with BOD removal and eutrophication, and effectively remove nutrients from wastewater (Ewel *et al.* 1982). In nearly all instances, wetlands act to renovate or improve water quality to some extent, but pollutant removal efficiencies are extremely variable (Chan *et al.* 1982).

There is great promise for the use of some wetland ecosystems as an effective medium of wastewater organic carbon removal (Khalid *et al.* 1982). The components remaining in wastewater that will exert oxygen demand, measured as BOD, are very effectively removed in wetland systems by the microbial flora (Kadlec and Tilton 1979). Optimal BOD removal is correlated with high surface area available for microbial growth, and shallow vegetated wetlands maximize this removal capability (Chan *et al.* 1982). BOD removal in natural wetlands ranges from 70% to 96% (Tchobanoglous and Culp 1980).

Wetlands may also provide a high degree of removal of suspended solids that originate in wastewater (Kadlec and Tilton 1979). Long detention times and thick vegetation filter suspended solids. Suspended solids removal ranges from 60% to 90% in wetlands (Tchobanoglous and Culp 1980).

Pathogens (bacteria and viruses) in wastewater are reduced by any processes that promote sedimentation or filtration and increase detention time (Chan *et al.* 1982; USEPA 1983). Thus, large, shallow, non-channelized wetlands encourage die-off of microbes (Chan *et al.* 1982). Kadlec (1981) reviewed studies that documented the introduction of significantly elevated levels of fecal coliforms into wetlands. The levels of fecal coliforms were reduced with passage of wastewater through these wetlands.

It has been amply demonstrated that some wetland ecosystems are capable of removing nitrogen and phosphorus compounds via a variety of mechanisms (Kadlec and Tilton 1979). Whereas nitrogen processing is largely biologically mediated, redistribution of phosphorus to internal sinks is a result of adsorption/precipitation reactions (Ewel *et al.* 1982). Adsorption and precipitation by soils are not necessarily permanent sinks for wastewater phosphorus, as these processes are at least partially reversible (Richardson and Nichols 1985). Therefore, some wetlands may eventually lose their ability to immobilize large quantities of phosphorus, but may retain their ability to immobilize or dissipate large quantities of nitrogen (Kadlec and Kadlec 1979). Wetland removal efficiencies for total nitrogen and total phosphorus are variable, ranging from 10% to 90% (Richardson 1985).

The capacity for nitrogen removal in wetlands is large (Chan *et al.* 1982); processes include volatilization, plant uptake, soil uptake, microbial uptake, sedimentation, nitrification, and denitrification. The major mechanism for removing nitrogen from wastewater applied to wetlands seems to be denitrification (Sloey *et al.* 1978; Kadlec and Tilton 1979; Nichols 1983), but Richardson and Nichols (1985) suggest that the disappearance of nitrogen from acid organic soils may be due as much to the chemical breakdown of nitrite as to denitrification.

Because the phosphorus cycle has no gaseous phase less phosphorus is removed from wastewater added to wetlands, although high, short-term removal efficiencies have been observed (Nichols 1983). The magnitude of phosphorus retention capacity varies considerably among wetland types (Richardson and Nichols 1985; Kelly and Harwell 1985). Successful phosphorus immobilization by wetland soils is related to contact time with organic matter (Kadlec and Tilton 1979), but the quantity of phosphorus adsorbed depends on the exchange equilibrium with the dissolved phase (Kadlec 1987). Plant uptake is generally less important than soil adsorption/precipitation reactions for retaining phosphorus in wetland ecosystems (Ewel *et al.* 1982) but the best possibilities for using wetland plants for nutrient removal appear to occur when the nutrients are stored in woody plants (Ewel and Odum 1978).

Flow through a wetland in northern Canada reduced orthophosphate by more than 95% (Hartland-Rowe and Wright 1975). A similar reduction of phosphorus occurred in a northern peatland receiving sewage

(Richardson *et al.* 1976). Greater exports of phosphate from channelized as compared with natural Coastal Plain streams occurred as a result of a reduction in their capacity to assimilate phosphate (Kuenzler *et al.* 1977). Most of the phosphorus added to surface water accumulated in the sediments in an alluvial swamp forest in the North Carolina Coastal Plain (Holmes 1977). The floodplain of a small Coastal Plain stream in North Carolina was a sink for phosphorus (Yarbro 1979). In the Santee River Swamp, phosphorus was adsorbed or deposited as sediments as water coursed through the floodplain from the river (Kitchens *et al.* 1975). Phosphorus accumulated in the floodplain of a tupelo swamp in southern Illinois (Mitsch *et al.* 1979).

In Wildwood, Florida, secondarily treated wastewater has been released for over 20 yr into a series of three wetlands. The wetland that directly receives the wastewater is dominated by *Typha latifolia* (cattail) and *Salix* sp. (willow). This marsh is covered by *Lemna* sp. (duckweed). The discharge from this wetland flows through a ditch to a mixed hardwood swamp dominated by *Fraxinus profunda* (ash), *Taxodium distichum* (bald cypress), and *Nyssa biflora* (black gum). The discharge from this wetland flows through another ditch to a much larger mixed hardwood swamp with similar species composition.

The first two wetlands receive higher nutrient loadings than the third wetland as most of the nutrient removal takes place within them (Brown *et al.* 1975). After flowing through the swamp, the concentration of nutrients in the water was reduced to values equal to or less than those found in a control swamp (Boyt *et al.* 1977). This effectiveness was evaluated in terms of nutrient concentrations. Reductions in terms of mass loading were calculated to be 87% for phosphorus. No visible stress or damage to the natural system was evident. Dilution rather than chemical or biological processes played the key role in reducing nutrient and organic loads. No buildup of nutrients in sediments was indicated. Results from tree borings showed significant differences in tree growth during a 19-yr period as compared to a previous 19-yr period. Therefore, trees did play an active role in removing nutrients (Brown *et al.* 1975). In addition, the number of fecal coliforms declined to background levels within 1 km of the point of wastewater discharge to the wetland (Boyt *et al.* 1977).

A cypress strand in Waldo, Florida, dominated by *Taxodium ascendens* (pond cypress), black gum, and *Acer rubrum* (red maple) has been receiving wastewater since 1934 from overflow of a community septic tank. This wetland reduced nutrient concentrations to background levels due to phosphorus retention in the sediments (Nessel 1978). Total phosphorus concentrations were reduced 51% in surface waters leaving this cypress strand and 77% after passing through the soil profile into shallow groundwater (Nessel 1978). Infiltration was a major route for water leaving this system. This facilitates phosphorus removal and explains the long-term effectiveness of this wetland in terms of phosphorus assimilation (Richardson and Davis 1987). Pond cypress tree growth was found to be stimulated and increased nutrient concentrations in wood and foliage were recorded (Nessel *et al.* 1982), but this represented only 1% of the estimated phosphorus inflow to the system (Nessel and Bayley

1984). Bacteria had low survival rates; 99% reduction was achieved in 32 days for the viruses tested (Butner and Bitton 1982).

Another cypress strand, Basin Swamp, in Jasper, Florida, has been receiving raw wastewater or primary or secondarily treated wastewater since 1914 (Tuschall *et al.* 1981). Total nitrogen and phosphorus concentrations in the surface water were effectively reduced by 69% and 36%, respectively between the inflow and outflow of the swamp. A portion of the reduction was attributed to dilution by surface runoff into the swamp. Discharge of raw and primary wastewater in the swamp decreased growth rates in pond cypress; however, discharge of secondarily treated wastewater enhanced growth over controls (Lemlich and Ewel 1984). The rate of fecal coliform export depended on the detention time of the strand (Brezonik *et al.* 1981). Based on their findings at the Jasper site, Fritz and Helle (1981) indicated that the use of a flow-through wetland system for additional treatment of secondarily treated wastewater is a workable and economical alternative to conventional physical-chemical treatment methods.

Most of the wastewater from the Walt Disney World Complex has been discharged into a mixed hardwood swamp since 1977. The site is dominated by red maple, black gum, bald cypress, and *Pinus elliottii* (slash pine). This wetland was isolated by berms and the discharge, which ultimately reaches Reedy Creek, is artificially controlled. This is the largest full-scale forested wetland effluent discharge system that has been extensively monitored in the U.S. (Knight *et al.* 1987). The long-term average removal rate was 75% for BOD and 80% for suspended solids. Total nitrogen concentration was reduced 88% but no total phosphorus reduction was observed (Kohl and McKim 1981). A net release of phosphorus from this system occurred, probably because the retention capacity of the swamp had rapidly become saturated (McKim 1982). Removal efficiency depended on input concentration as lower removal efficiencies resulted from lower input concentrations over the range of values observed (Knight *et al.* 1987).

Pottsburg Creek Swamp, a mixed hardwood swamp, in Jacksonville, Florida, has been receiving secondarily treated wastewater since 1967. This wetland is vegetated by a mixture of species including ash, red maple, black gum, pond cypress and *Liquidambar styraciflua* (sweetgum). Based on mass balance calculations, total nitrogen loadings were reduced by 87% and total phosphorus loadings by 62% (Winchester and Emenhiser 1983). There were no net concentrating or diluting effects and, therefore, nutrient reduction was due to infiltration within the swamp.

Cypress domes are a common type of swamp in Florida (Ewel and Odum 1984). These forested wetlands are dominated by pond cypress and often have large numbers of black gum (Penfound 1952). The term "dome" comes from the characteristic profile of these wetlands, because the trees are taller in the center and decrease in size toward the edges (Mitsch 1984). A study of the use of cypress domes for the advanced treatment of domestic wastewater was conducted from 1975 to 1979.

Biochemical oxygen demand was not substantially reduced as the wastewater traveled from the center to the edge of the domes (Dierberg and Brezonik 1978). In contrast to this, the concentrations of nutrients were generally lower in the surface waters at the edges of domes receiving wastewater than at the center, but the overall reduction of nutrient concentrations in the surface waters was less than 33% (Dierberg and Brezonik 1983a). Infiltration of secondarily treated effluent through organic sediments lining the basins of the cypress domes reduced nitrogen and phosphorus concentrations to background levels (Dierberg and Brezonik 1983a).

Eighty seven percent of the total nitrogen entering the system was stored in peat, roots, and wood, or was released to the atmosphere by denitrification, and approximately 92% of the phosphorus entering the system was removed by plant uptake or sediment deposition (Dierberg and Brezonik 1983b). Based on leaching studies using laboratory columns, organic soils in the domes have a large phosphorus adsorption capability (Dierberg 1980) and this removal capability could continue for a long time (Dierberg and Brezonik 1983a). The cypress trees accounted for storage of 24% of the estimated nitrogen inflow but only 1% of the estimated phosphorus inflow to the system (Dierberg and Brezonik 1984).

After 5.5 yr of wastewater disposal, the understory vegetation and existing trees showed no detrimental effects (Ewel *et al.* 1981). The most striking response of understory vegetation was the development and persistence of a thick layer of duckweed over the entire surface of the domes receiving wastewater (Ewel 1984). Tree growth rates were also unaffected (Ewel *et al.* 1981). The number of fecal coliforms (Fox *et al.* 1984) and viruses (Scheuerman 1978) were reduced during infiltration of surface water to the shallow groundwater aquifer. Binding of viruses may not be permanent (Scheuerman 1978) and the dome substrate may not be a perfect filter (Wellings *et al.* 1975). In summary, the cypress domes studied and their associated sediments can reduce the levels of major wastewater constituents to levels comparable to those of conventional tertiary treatment processes (Dierberg and Brezonik 1983b) and can thus serve as a natural tertiary treatment system (Dierberg and Brezonik 1983a).

Results from the studies reviewed are difficult to generalize upon in a quantitative way. However, some qualitative conclusions about wetland transformation and assimilation of different forms of nitrogen and phosphorus can be reached (Richardson and Davis 1987). First, nitrogen removal from water was consistent and substantial over a range of loading rates. Removal efficiency was generally 75% or more on a mass loading basis. Soils provided a finite and reversible sink for ammonium and phosphorus, and retention capacity depended on a complex of factors. In contrast to nitrogen removal, efficiency of phosphorus removal varied greatly. Natural wetlands can process significant amounts of nitrogen, and can be managed to assimilate even more. Phosphorus retention is highly variable and highly dependent on the characteristics of the wetland ecosystem involved and the loading rates.

Wetlands differ in their ability to store and release nutrients. Some types of wetlands dominated by woody plants (swamps) may be capable of assimilating excess nutrients through microbial processes and long-term storage in the soil and in vegetation. Caution must be used when making generalizations about nutrient removal efficiencies from a diverse and sparse data set which includes a variety of wetland types and a wide range of years of application (Richardson and Nichols 1985). However, trends from the most complete studies show a general pattern of decreased nutrient removal efficiency with time and with higher loading rates (Richardson and Nichols 1985).

Chemistry in Acidic Waters

Most natural waters are buffered principally by a carbon dioxide-bicarbonate system. By observing the equilibrium chemistry (dissociation relationships) of a system the proportions of carbonic acid (plus dissolved carbon dioxide), bicarbonate, and carbonate at various pH values can be evaluated. Because of the ubiquitous nature of carbonate rocks and the equilibrium reactions of carbon dioxide, bicarbonate and carbonate are present as bases in most natural waters (Stumm and Morgan 1981) but all waters with a pH less than 8.5 contain acidity (Sawyer and McCarty 1978). Uncombined carbon dioxide, organic acids (such as tannic or fulvic), and salts of strong acids are responsible for the acidity of natural waters (Wetzel 1975). In waters with a pH below 5, carbonic acid (plus dissolved carbon dioxide) dominates the carbonate equilibria (Wetzel 1975) but this alone will not depress the pH below a value of about 4.5 (Sawyer and McCarty 1978). Depression of pH below 4.5 is due to mineral acidity which is exhibited by waters containing acids stronger than carbonic acid (Stumm and Morgan 1981). At a pH of 3 to 4.5, carbonate and bicarbonate are not buffering the water; rather, organic acids are the buffer (Thurman 1985).

The proportions of carbonate in surface waters come from the weathering of rocks, and the solubility of carbon dioxide in water increases markedly in water that contains carbonate (Wetzel 1975). Therefore, if there is a high concentration of carbonates in surface waters due to contact with carbonate containing parent material, then a definite amount of carbon dioxide will remain free in solution after equilibrium is reached between calcium, bicarbonate and carbonate (Wetzel 1975). If surface waters are isolated from the carbonate rich Floridan Aquifer (Fernald and Patton 1984) then there is probably very little free carbon dioxide present in those surface waters. Conductivity is a useful indicator of whether the source of water entering a peatland is primarily from precipitation and shallow mineral soil inflow (and therefore not in contact with carbonate containing parent material) or groundwater (Verry 1975). Values less than 80 µmhos/cm are indicative of a perched water table. Values greater than 80 µmhos/cm indicate a groundwater table.

The carbonate equilibria for Austin Cary cypress dome (mean pH = 4.5) was examined by Dierberg (1980). Only trace amounts of bicarbonate existed in the water as there was no titratable alkalinity. Therefore, it is appropriate to measure phenolphthalein acidity rather than alkalinity in acidic waters.

Phenolphthalein acidity is a measure of the free (or uncombined) carbon dioxide and the mineral acidity present in the surface water (Sawyer and McCarty 1978). Highly colored natural surface waters typically have low pH due to the acidic nature of humic substances that are present. The high color is principally due to tannins, humic acid, humates, and the decomposition of lignins (Sawyer and McCarty 1978) but color in surface waters in Florida streams and canals may be of organic or mineral origin (Kaufman 1975b). The inorganic sources are metallic substances such as iron and manganese compounds (Christman *et al.* 1967).

Low pH values are found in natural waters rich in dissolved organic matter, especially in systems that contain large amounts of sphagnum (Wetzel 1975). In wetlands, dissolved organic matter usually exceeds dissolved inorganic matter, which is not the usual case in surface waters (Thurman 1985). The most likely major sources of hydrogen ions in these waters are the dissociation of H_2SO_4 derived from H_2S (Gorham 1956) and the active cation exchange in the cell walls of sphagnum where the release of hydrogen ions occur (Clymo 1964). Hydrogen ions are also produced by organic decomposition (Clymo 1967).

Increases in acidity occurs whenever the production of organic matter is greater than decomposition, as in systems where peat is present (Stumm and Morgan 1981). This is because the assimilation of ammonium produces hydrogen ions. Areas with low relief and slowly moving water provide conditions for peat formation and for the gradual accumulation of partly decomposed organic materials (Davis 1946). The chemical nature of the plant tissues forming the peat (humic acids) tend to make this peat material acid and the most acid peats are those formed from swamp plants and sphagnum moss (Davis 1946). In addition, the poor buffering capacity of precipitation reaching a wetland can further accentuate low pH (Thurman 1985).

Information on nitrogen transformations in acidic, highly organic flooded soils is limited and these processes may occur in unique ways (Haack 1984). Compounds found in naturally occurring humic-colored waters reduce dissolved oxygen levels and, therefore, these waters act as a sink for dissolved oxygen (Dierberg 1980). Low dissolved oxygen can lead to anaerobic conditions where net ammonification (the release of ammonium during microbial decomposition of organic matter) is often noted (Tusneem and Patrick 1971). In addition, low pH as well as organic compounds have been shown to inhibit the nitrification of ammonium to nitrate (Dierberg 1980). Therefore, low dissolved oxygen, low pH and the presence of organic compounds contribute to the dominance of ammonium rather than nitrate plus nitrite in these waters. Through the inhibition of nitrification, ammonium becomes the dominant inorganic nitrogen species and this leads to conservation of nitrogen in the system (Dierberg 1980).

Nitrate plus nitrite concentrations may also be low in these waters due to rapid plant uptake and denitrification, although denitrification has also been shown to be inhibited at low pH (Mitchell 1974;

Brezonik 1977). Nitrate was added to jar and core microcosms composed of water and soil from the study site (Haack 1984). Nitrate loss did occur in both the jar and core microcosms but sediment was necessary for the nitrate loss. No mechanism for nitrate loss was substantiated, although it may be due to denitrification, which has been shown to occur in wetlands. Chemical reduction (as apposed to biological reduction) of nitrate at low pH has been proposed to occur through several pathways. Wetlands with low pH, high organic matter, and humic compounds should have pathways of nitrate loss other than biological denitrification (Haack 1984). Under highly reduced conditions, nitrate reduction to ammonium and organic nitrogen is possible (Buresh and Patrick 1978). These processes would also account for the dominance of ammonium in these waters.

The shallow nature of surface water in cypress domes and hence the close proximity of soil and water suggests that the phosphorus content of the surface water may be controlled by the interaction of the phosphorus with the soil (Dierberg 1980). The nature of soil/phosphorus reactions is complex. In general, the inorganic phosphorus is partitioned between the solution phase (small fraction of total system phosphorus) and the solid phase (a larger portion of total system phosphorus). The chemical species of solution phosphorus are a function of the reactions of protonation and soluble metallic complex ion formation (Bohn *et al.* 1979). At low pH, iron and aluminum ions on solid (soil) surfaces form bonds with solution species (Stumm and Morgan 1981). The resulting precipitate removes phosphorus from the water column. The oxygen content of the water and soil also effects the amount of phosphorus in solution as phosphorus becomes more soluble under reduced anaerobic conditions (Stumm and Morgan 1981). Therefore, soluble metallic ion complex formation (phosphate and hydrous oxides of iron and aluminum) plays a great role in controlling phosphorus levels in natural waters.

The limit for the phosphorus concentration in solution is set by the dissolution and precipitation of these sparingly soluble phosphorus compounds and the adsorption of phosphorus on the surface of soil particles. In general the overall solubility of these metal phosphate complexes is inversely related to pH while adsorption and precipitation of phosphorus is directly related to pH (Stumm and Morgan 1981). Therefore, the lower the pH the greater the solubility of the metal phosphate complexes and the greater the adsorption and precipitation of phosphorus in the soil. Removal of phosphates from solution can also be linked to pH because of the dependency of the reactions upon soil aluminum (Dubuc *et al.* 1986). At a low pH in cypress domes studied by Dierberg (1980) aluminum rather than iron controlled phosphorus solubility. In addition, at a pH less than 6, organic phosphorus also precipitates as a complex with iron and aluminum (Dubuc *et al.* 1986).

Solubilization of the sparingly soluble compounds can also occur due to the production of organic acids. These organic acids exist in water as negatively charged colloids which exert a holding action for metallic ions such as iron and aluminum (Kaufman 1975b). The sorption of phosphate by these organometallic complexes occurs but the dynamics of the transformations are still unclear. Phosphate can react with metal ions to form complexes in the presence of organic ligands such as fulvic and humic acids (Boto and Patrick 1978). Phosphate ions may be acting as ligands in organometallic compounds (Sinha 1971). In either case the retention is a function of pH.

Biological immobilization of phosphorus also occurs within wetlands (Chan *et al.* 1982). Wetland trees have been shown to assimilate phosphorus (Brown *et al.* 1975; Nessel *et al.* 1982; Dierberg and Brezonik 1983b). In addition high cation exchange capacity exhibited by peat can lead to the absorption of phosphate anions (Moore and Bellamy 1974). Therefore, it appears that through biological and chemical processes in wetlands, low levels of phosphorus can be maintained in surface waters.

Methods

All water samples taken at the site were grab samples. Samples were collected monthly in 1982, quarterly in 1983-84, and monthly in 1986-87. During the background and post-effluent data collection period, the following parameters were determined: orthophosphate (OP), total phosphorus (TP), ammonia-nitrogen (NH_3), nitrate plus nitrite-nitrogen (NO_3+NO_2), total Kjeldahl nitrogen (TKN), chloride (Cl), conductivity, turbidity, five-day biochemical oxygen demand (BOD_5), acidity, pH, dissolved oxygen (DO), temperature, and color.

APHA (1985) and EPA (1979) methods were used for all analyses (Table 1). Temperature, dissolved oxygen (DO), and pH were determined in the field. Dissolved oxygen was measured with a YSI meter model 54. Measurements were made at mid-level depths in the water column; no attempt was made to determine dissolved oxygen at different water column levels. Four acid-washed Nalgene bottles were filled at each surface water site, with samples taken at mid-level depths when possible. Two Nalgene bottles were used to collect groundwater samples using a gasoline-powered pump. Wells were not consistently pumped dry before sampling. Two ml/l of concentrated sulfuric acid was added in the field as a preservative to bottles used for NH_3, NO_3+NO_2, TKN, and TP analyses. Samples to be analyzed for OP were filtered in the field through Gelman 0.45 μm membrane filters. Once collected, samples were kept at 4°C until analysis, or frozen if held longer than the EPA's recommended holding time (EPA 1979). To ensure the accuracy of data generated, at least five standards were run before and after each analysis, and between every 40 samples. One duplicate sample and one spiked sample were included for each 10 samples analyzed. Reference standards obtained from US EPA were also analyzed with samples for TKN, TP, Cl, NO_3+NO_2, and NH_3. Values determined for these reference standards all fell within EPA's 95% confidence interval. Mean recoveries of spikes were 104.1% for Cl, 93.0% for TKN, 94.0% for TP, 99.7% for NH_3, and 96.6% for NO_3+NO_2. Sample concentrations were calculated from calibration standard data using linear regression. The coefficient of determination (R^2) was always greater than 0.990.

Table 1. Analytical methods used.

Water Quality Parameter	Method Used
Total Kjeldahl Nitrogen	315.2 (EPA 1979)
Total Phosphorus	365.4 (EPA 1979)
Chloride	325.2 (EPA 1979)
Conductivity	205 (APHA 1985)
Orthophosphorus	365.1 (EPA 1979)
Nitrate + Nitrite	353.2 (EPA 1979)
Ammonia	350.1 (EPA 1979)
Biochemical Oxygen Demand	507 (APHA 1985)
Acidity	402 (APHA 1985)
Turbidity	214A (APHA 1985)
Color	at 420 nm (filtered sample and ambient pH adapted from APHA 1985)
Dissolved Oxygen	421 F (APHA 1985)
pH	423 (APHA 1985)
Temperature	212 (APHA 1985)

Laboratory Instruments Used

Technicon Autoanalyzer II
Perkin Elmer Model 552 Spectrophotometer
Perkin Elmer Model 5000 Atomic Absorption Spectrophotometer
Hach Nephelometer
YSI Model 31 Conductivity Bridge
YSI Model 54A Oxygen Meter
Orion pH Meter

Sampling and preservation, analytical procedures, and quality assurances are described in more detail by Best *et al.* (1983, 1987).

Results and Discussion

Surface Water Chemistry

Baseline review. Statistical comparisons of pre-effluent water quality sampling data indicated that there were significant differences ($p<.05$) between the mean values for any of the parameters at sites upstream of the (then) proposed effluent discharge location when compared to sites downstream. The surface water was characterized (Table 2) as highly colored (mean color = 306 c.v.) and acidic (mean pH = 3.9). The mean level of dissolved oxygen (2.8 mg/l) and the mean levels of BOD_5 (2.8 mg/l),

Table 2. Cumulative mean and standard error for surface water chemistry parameters measured monthly during **pre-effluent baseline period** from January-December 1982 and January-May 1985 at various sampling locations in the titi shrub swamp near Apalachicola Florida. Sites correspond to locations noted in Figure 2.

Site		Temp. (deg. C)	Turbid. (ntu)	Tot.Phos. (mg/L)	TKN (mg/L)	B.O.D. (mg/L)	pH	Diss.Ox. (mg/L)	Conduct. (umho/cm)	Chloride (mg/L)	Orthophos. (mg/L)	Ammonia (mg/L)	Nitrate/Nitrite (mg/L)	Acidity	Color
U1200	MEAN	20.7	1.2	0.01	0.92	3.4	4.1	2.9	51	6	0.008	0.11	0.01	27	210
	SE	1.4	0.2	0.00	0.17	0.4	0.1	0.4	3	1	0.007	0.04	0.01	2	20
U400	MEAN	20.0	1.5	0.01	0.98	3.0	3.9	1.2	67	5	0.003	0.08	0.00	41	334
	SE	1.4	0.3	0.00	0.23	0.3	0.1	0.3	5	0	0.002	0.02	0.00	2	27
U25	MEAN	20.1	1.2	0.01	0.94	2.6	3.4	2.7	85	6	0.003	0.08	0.01	43	341
	SE	2.5	0.2	0.00	0.16	0.3	0.3	0.7	11	2	0.002	0.05	0.00	7	56
POND	MEAN	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	SE	-	-	-	-	-	-	-	-	-	-	-	-	-	-
D25	MEAN	21.2	2.0	0.01	0.98	2.8	3.3	3.6	85	6	0.002	0.08	0.02	40	366
	SE	2.8	0.4	0.01	0.22	0.4	0.3	0.7	9	2	0.001	0.02	0.01	8	65
D200	MEAN	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	SE	-	-	-	-	-	-	-	-	-	-	-	-	-	-
D400	MEAN	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	SE	-	-	-	-	-	-	-	-	-	-	-	-	-	-
D900	MEAN	21.4	3.2	0.01	0.62	2.4	4.0	2.5	53	5	0.005	0.10	0.01	34	162
	SE	1.4	0.7	0.00	0.06	0.3	0.1	0.3	3	0	0.003	0.03	0.00	3	23
D1500	MEAN	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	SE	-	-	-	-	-	-	-	-	-	-	-	-	-	-
D1800	MEAN	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	SE	-	-	-	-	-	-	-	-	-	-	-	-	-	-
D2700	MEAN	21.9	1.9	0.01	0.87	2.4	4.0	3.6	64	6	0.003	0.11	0.01	28	254
	SE	1.5	0.5	0.00	0.16	0.3	0.1	0.4	2	1	0.002	0.05	0.00	1	18

TOC (41.6 mg/l), and COD (90.8 mg/l) were low, indicating a low rate of decomposition. The mean conductivity (66.2 μmhos/cm) was low, indicating a low dissolved solids content. The absence of any buffering capacity (mean of 39.1 mg $CaCO_3$/l) indicated that the wetland community could be vulnerable to the effects of increased pH, which occurs with wastewater discharge.

Mean nutrient concentrations were consistent with other forested freshwater wetlands (Carter *et al.* 1973; Boyt 1976; Dierberg and Brezonik 1980; Richardson 1981). Most of the nitrogen was found to be in the organic form (mean TKN = 0.984 mg N/l), much greater than most abundant soluble nitrogen form, ammonia (mean = 0.084 mg N/l), indicating a reduced system. Mean ammonia levels were 0.011 mg N/l. Dissolved oxygen levels were also low, limiting nitrification and probably accounting for the low nitrate levels found. Low phosphorus concentrations (mean TP = 0.014 mg P/l) and high N:P ratio (94:1) indicated a phosphorus-limited, unenriched natural water body. Primary nutrient sources were

Table 3. Mean nutrient levels in precipitation (mg/l).

Variable	Mean	n	Minimum	Maximum
TKN	0.816	3	0.364	1.125
NO_3	0.125	5	0.096	0.159
TN	0.942	3	0.502	1.235
TP	0.047	5	0.011	0.092

from precipitation (mean TKN = 0.816 mg N/l and mean TP = 0.047 mg P/l) (Table 3). Total nitrogen levels were found to increase slightly with passage through the swamp, indicating some export of organic forms, whereas total phosphorus levels remained relatively constant, reflecting limited availability and more complete assimilation of this nutrient.

Conductivity and chloride. Conductivity and chloride (Figure 3) were monitored as conservative "tracer" parameters and indicated movement of wastewater within the wetland. Effluent had reached site 5 (900 m from point of wastewater discharge) within 8 wk of effluent introduction, as shown by increased conductivity and chloride values. After 16 wk, wastewater had reached the outlet of the wetland (site 7) approximately 2400 m from the wastewater discharge pipe. The residence time for wastewater within the wetland was therefore about 16 wk.

Conductivity was above background levels throughout the entire wetland after 1 yr of wastewater discharge ($p = 0.005$) (Figure 3). At the outlet, conductivity was greater than 107 μmhos/cm above

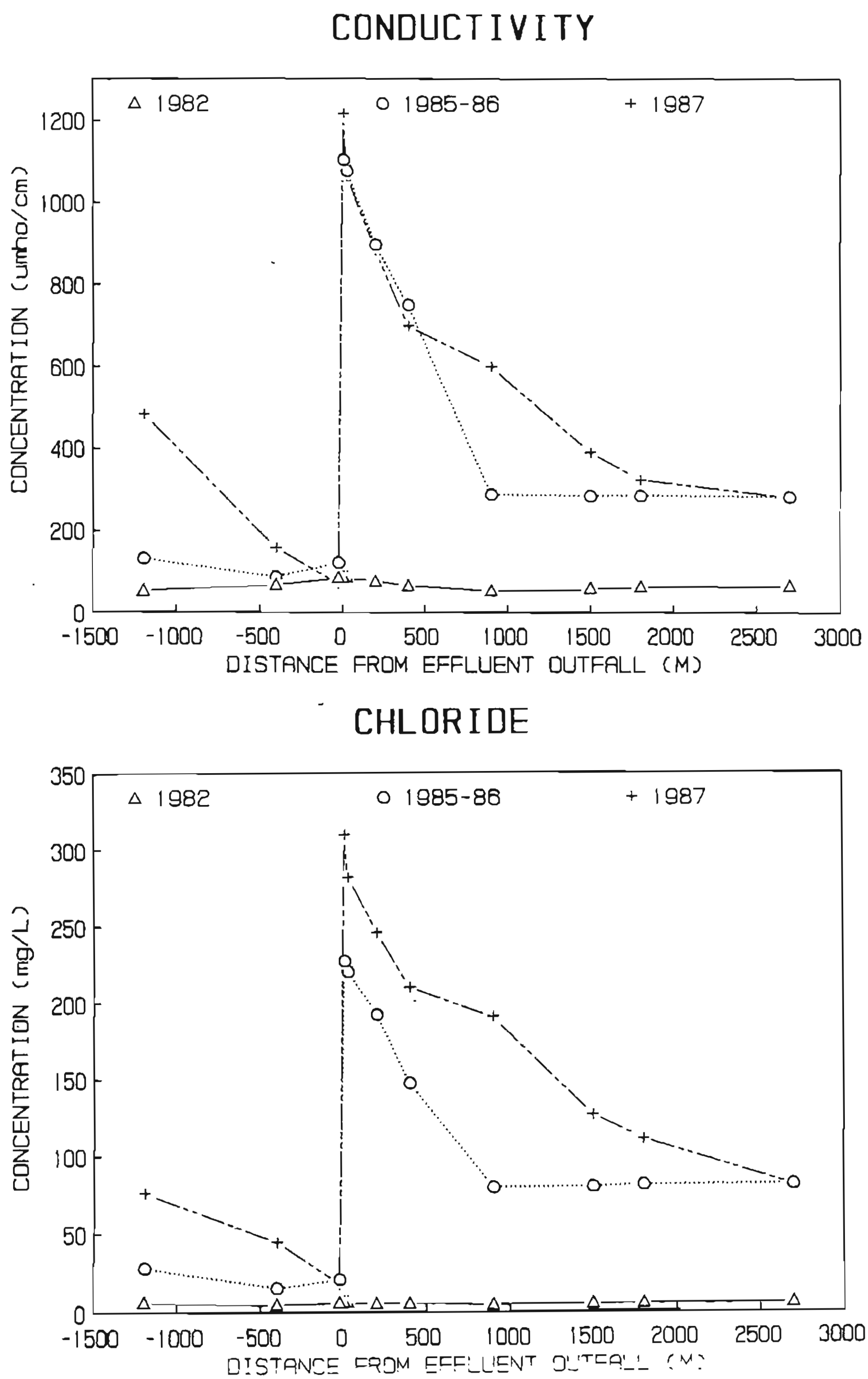


Figure 3. Conductivity, (umbo/cm) and chloride concentration (mg/l) at various distances upstream ("-" distances on figure) and downstream ("+" distances on figure) from point of wastewater discharge to titi-shrub swamp near Apalachicola, Florida. Data are means of replicate monthly values for preapplication (1982), one-year (1985-86) and two-year (1987) postapplication.

natural baseline ($p = 0.007$). Using simple linear regression, distance from discharge explained 39.2% of the variation in conductivity readings, while flow velocity explained 31.3%, sewage treatment plant output explained 23.4%, and precipitation explained 5.8%. When these four factors were combined using multiple linear regression, they accounted for 58.4% of the variation in conductivity values in the wetland.

Chloride is a conservative, non-reactive ion and its reduction in concentration over distance can largely be explained by dilution (Figure 3). Assuming that the difference in chloride concentration between the wastewater input and the wetland output was due solely to dilution, then about 65% of the reduction in nutrient concentrations in the treatment wetland could be explained by dilution processes. Using linear regression, chloride concentration correlated best with flow velocity in the wetland ($r = -0.675$). When the independent variables of distance from discharge point and time were added using multiple regression, 59.2% of the variation in chloride concentration was explained.

Nitrogen. Nitrogen, as measured by TKN in the wastewater, was effectively retained by the wetland ecosystem. The mean value for TKN over the 1-yr post-effluent monitoring period was not significantly different from the background TKN mean at the wetland outlet (95% confidence level) (Figure 4). TKN levels in the effluent were comparatively low, averaging 4.6 ± 3.1 mg TKN/l (data from treatment plant).

Total ammonia nitrogen in the treatment plant effluent averaged 2.2 ± 1.2 mg NH_3-N/l (data from treatment plant), but was reduced to background levels at distances greater than 800 m from the discharge point ($p = 0.01$) (Figure 4). Nitrate plus nitrite-nitrogen input was 4.0 ± 3.9 mg NO_3+NO_2-N/l in the effluent (data from treatment plant), and was reduced to background levels within 200 m of the discharge ($p = 0.01$) (Figure 4).

Multiple linear regression using five independent variables of time, precipitation, sewage treatment plant output, distance from discharge, and flow velocity explained only about 15% of the variation in nitrogen concentrations as measured by TKN. Biological processes of the nitrogen cycle and natural water quality variability could explain the bulk of variation in nitrogen compound levels in the treatment plant wetland.

Mechanisms for nitrogen removal in wetlands include nitrification, denitrification, utilization by plants, and incorporation into soil organic matter. Ammonium-nitrogen in flooded soils is first nitrified and then denitrified in a cycle that occurs by diffusion at the aerobic sediment/anaerobic sediment interface (Patrick and Reddy 1976). The possibility of nitrate-nitrogen leaching to the groundwater is of public health concern regardless of where wastewater is discharged on the landscape. In wetlands, Bartlett *et al.* (1979) showed that 90-95% of nitrate added to wetland soil was denitrified by dissimulatory reduction to nitrous oxide and nitrogen gas. Aquatic macrophytes can provide temporary storage for

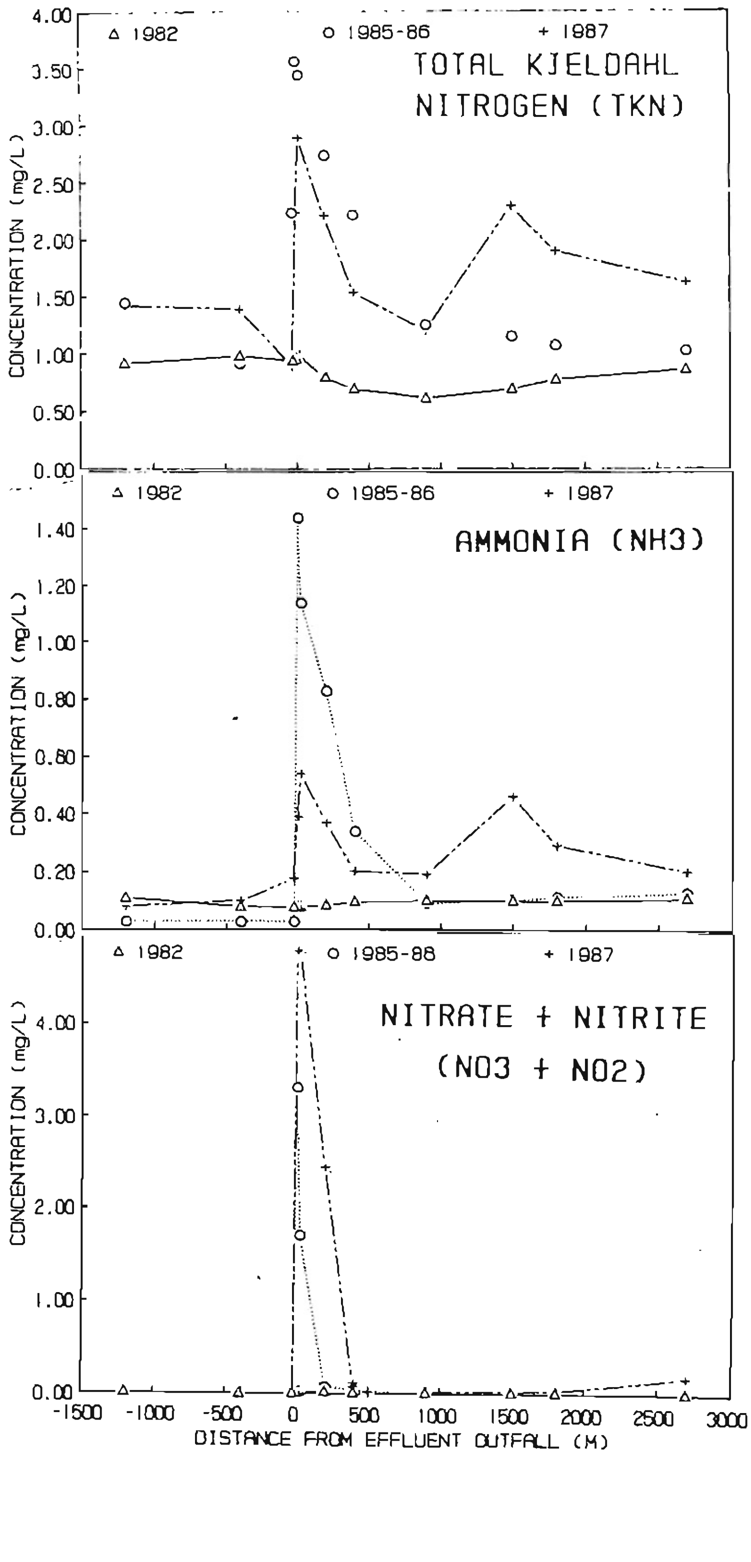


Figure 4. TKN, Ammonia and Nitrate and Nitrate concentrations (mg/l) at various distances upstream ("-" distances on figure) and downstream ("+" distances on figure) from point of wastewater discharge to titi-shrub swamp near Apalachicola, Florida. Data are means of replicate monthly values for preapplication (1982), one-year (1985-86) and two-year (1987) postapplication.

excess nitrogen. A water hyacinth system receiving sewage effluent provided an 87% concentration reduction of added nitrogen compounds (DeBusk *et al.* 1983). Nitrogen loss by decomposition of organic matter is accelerated by frequent cycles of aerobic and anaerobic conditions (Lance and Whisler 1972; Reddy and Patrick 1975).

Phosphorus. Unlike nitrogen compounds in wastewater, phosphorus was less effectively removed in the flow-through wetland at Apalachicola. Total phosphorus inputs from the sewage treatment plant averaged 2.4 (± 0.9) mg P/l (data from treatment plant). An arbitrary phosphorus concentration of 1 mg/l was chosen to monitor the movement of phosphorus through the wetland. The 1 mg/l phosphorus front advanced to 75 m downflow within 2 wk of sewage treatment plant startup. It was located at 150 m after 1 mo of operation, 250 m after 6 wk, and 350 m after 3 mo. After 1 yr (May 1986), the 1 mg/l phosphorus front stabilized at about 600 m downgradient from the wastewater discharge pipe. Levels of phosphorus at the wetland outlet (2400 m downflow from the effluent input) were significantly higher ($p = 0.005$) 1 yr after discharge began, but still well below the Florida DER regulatory limit of 0.2 mg P/l (Figure 5). Linear regression showed that 38.6% of the variation in phosphorus concentrations was due to variation in distance from the discharge point.

Phosphorus removal from wastewater applied to wetlands depends on many processes. Uptake of dissolved phosphorus by aquatic plants and algae is a significant temporary sink for added phosphate. Unless biomass is harvested, most accumulated phosphate will be re-released to the water column following death and decay of plant material (DeBusk *et al.* 1983; Dolan *et al.* 1981). Accumulation of detritus (soil formation) is a removal mechanism for organic forms of phosphorus, but is a slow, long-term process (Richardson 1985). Several studies have shown that adsorption of phosphate by underlying sediment is the principal removal mechanism for added phosphate contained in wastewater applied to wetlands (Dierberg 1980; Dierberg and Brezonik 1983; Dolan *et al.* 1981).

Phosphate adsorption by soils is a function of pH. As pH increases, phosphate adsorption decreases (Lopez-Hernandez and Burnham 1974). Redox potential also affects phosphate sorption. Anaerobic sediments can adsorb more phosphate than aerobic soils from water with a high concentration of dissolved phosphorus (greater than 5 mg/l), but when concentrations of phosphorus are low (less than 5 mg/l), anaerobic soils will release more phosphorus to the surrounding water than will aerobic soils (Holford and Patrick 1979; Patrick and Khalid 1974). According to field observations, after addition of wastewater to the Apalachicola wetland, sediment conditions have been continuously anaerobic, whereas previously, seasonal drying cycles permitted occasional aerobic periods.

Perhaps the most important factor influencing adsorption of phosphorus by soils and sediments is the amount of amorphous iron and aluminum present in the soil. Many investigators have demonstrated that the best predictor of phosphorus sorption capacity of a soil is the amount of oxalate-extractable iron and aluminum contained in the soil (Fox and Kamprath 1971; Khalid *et al.* 1977; Larsen *et al.* 1959;

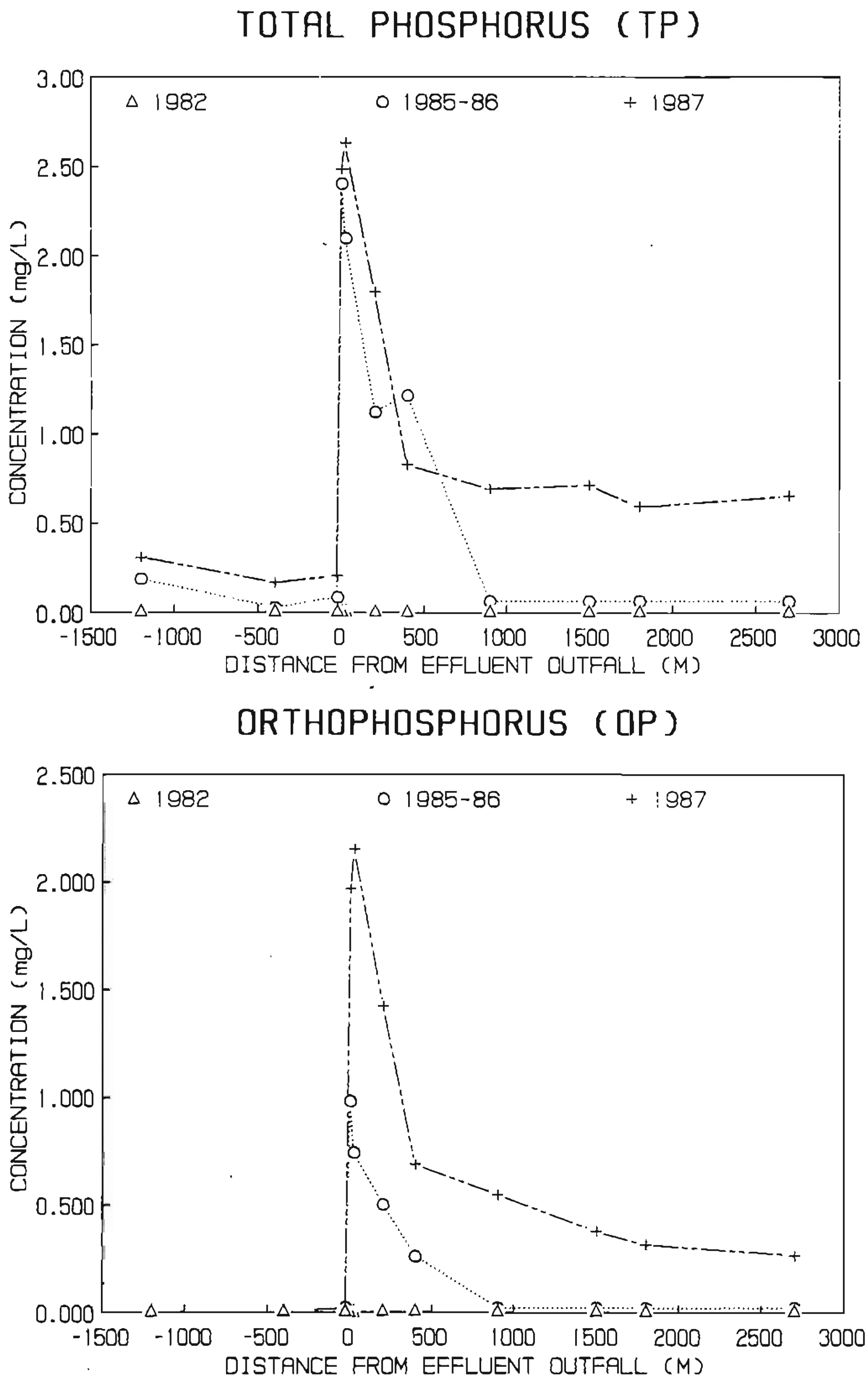


Figure 5. Total phosphorus and orthophosphorus concentrations (mg/l) at various distances upstream ("-" distances on figure) and downstream ("+" distances on figure) from point of wastewater discharge to titi-shrub swamp near Apalachicola, Florida. Data are means of replicate monthly values for preapplication (1982), one-year (1985-86) and two-year (1987) postapplication.

Richardson 1985). Soil at the Apalachicola site was not analyzed for iron or aluminum content; however, surface water contained considerable iron (215 ± 33 μg/l) (Best *et al.* 1983).

Because soil adsorption of phosphate occurs mainly at the thin soil-water interface, water depth and water velocity are also important influences on phosphorus removal potential of a wetland. Less phosphorus is adsorbed by sediments as water depth and/or water velocity increase. Tilton and Kadlec (1979) found significantly higher total dissolved phosphorus concentrations in deeper (30 cm) areas of a wetland receiving wastewater compared to shallower (6 cm) areas. Peverly (1982) found that a flow-through natural wetland functioned as a nutrient sink only at low flow rates. The phosphorus removal demonstrated at the flow-through wetland in Apalachicola, Florida, agrees with other studies done on flow-through systems (Kadlec 1983; Spangler *et al.* 1977; Vega and Ewel 1981). The phosphorus retention results of this study, however, contrast with those of stagnant or shallow water wetland treatment systems, such as cypress domes (Dierberg and Brezonik 1983; Dolan *et al.* 1981; Tilton and Kadlec 1979).

DO, BOD, pH, color, turbidity, and acidity. Part of the wetland's response to wastewater input was prolific algal and macrophyte growth in the impact zone as noted in field observations. Plant growth increased the DO content of the water above background levels in the discharge area (Figure 6).

Effluent from the treatment plant was low in BOD, averaging 5.8 ± 2.5 mg/l (data from treatment plant), and BOD remained at background levels throughout the wetland for the 1-yr monitoring period following wastewater discharge (Figure 6).

The mean pH of the effluent was 7.2 ± 0.2 (data from treatment plant), well above the wetland background mean pH of 3.8 ± 0.3. Because of the influence of sewage treatment plant input, pH was elevated above background in the wetland (Figure 7). At 1400 m downflow, the pH was greater than 1.5 units above natural background ($p = 0.0046$) after 1 yr of wastewater discharge.

Post-effluent color and turbidity did not differ significantly from background levels. Acidity was reduced because water discharged from the sewage treatment plant was higher in calcium carbonate (being from a groundwater source). Buffering capacity of the wetland system was thus substantially increased over natural levels.

Post-effluent. The pH was used as an on-site indicator of water quality changes, since an increase in pH was correlated to direct contamination of the otherwise low pH system with wastewater. Comparing pre-effluent and post-effluent (1985-86) mean pH at site D25 (close to the effluent outfall), an increase in pH of from 3-4 to 6-7 was observed (Figure 7). The control sites U1200, U400, and U25 did not exhibit a change in mean pH during any of the sampling periods. The mean pH at downstream sites D900 and D2600 rose only slightly during the 1985-86 post-effluent sampling period.

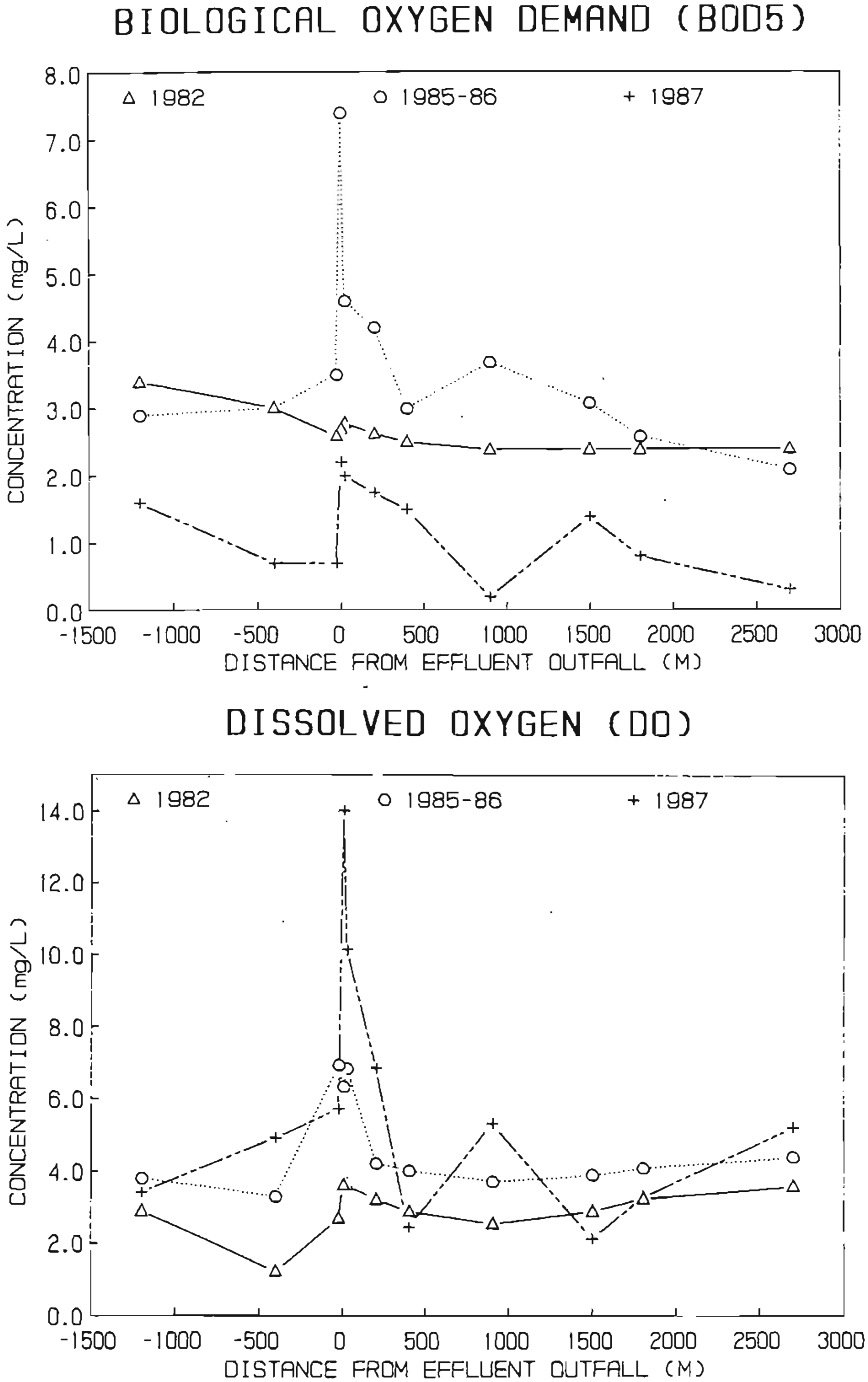


Figure 6. Biochemical oxygen demand and dissolved concentrations (mg/l) at various distances upstream ("-" distances on figure) and downstream ("+" distances on figure) from point of wastewater discharge to titi-shrub swamp near Apalachicola, Florida. Data are means of replicate monthly values for preapplication (1982), one-year (1985-86) and two-year (1987) postapplication.

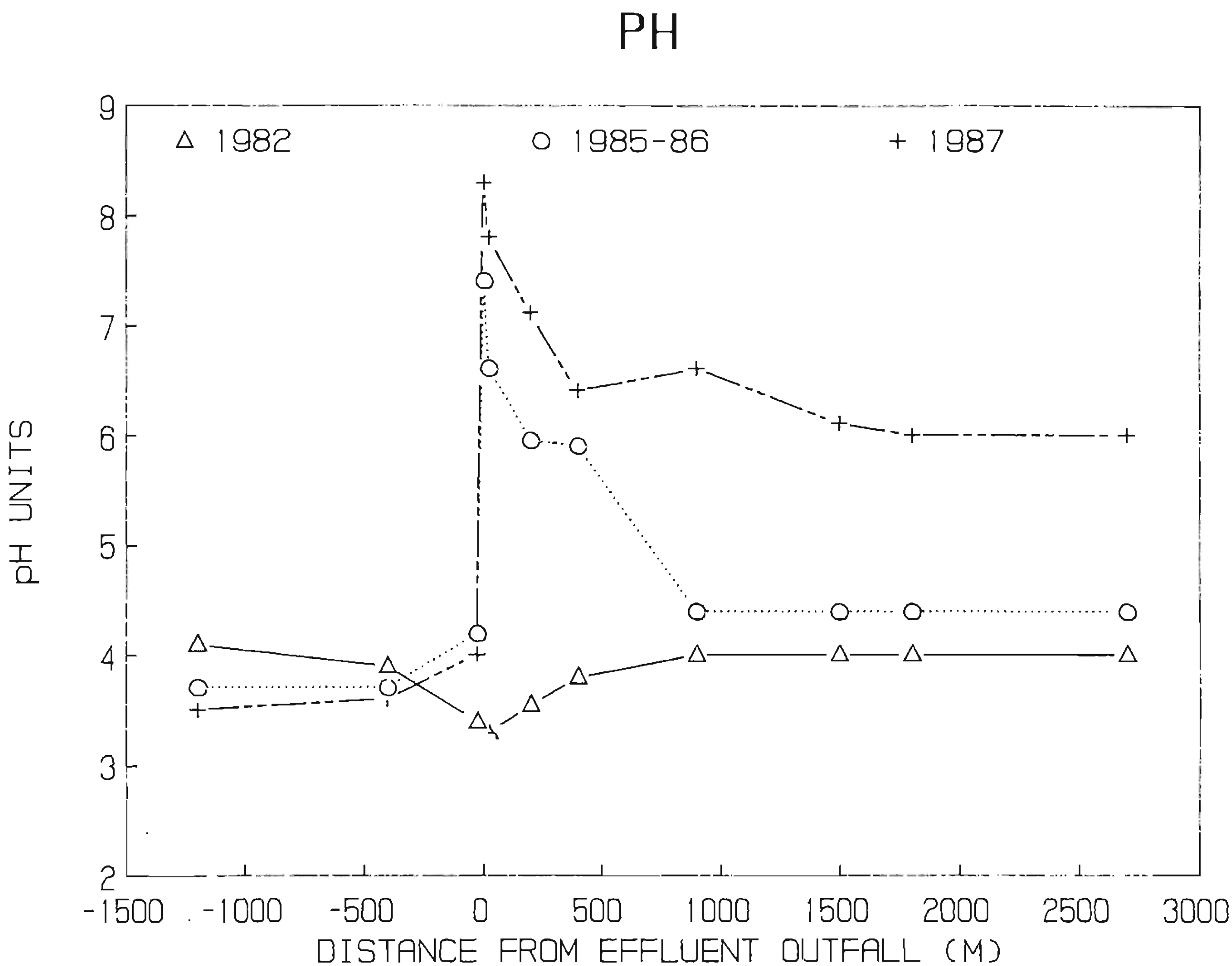


Figure 7. Ph at various concentrations (mg/l) at various distances upstream ("-" distances on figure) and downstream ("+" distances on figure) from point of wastewater discharge to titi-shrub swamp near Apalachicola, Florida. Data are means of replicate monthly values for preapplication (1982), one-year (1985-86) and two-year (1987) postapplication.

Measurments of mean of pH, TKN, TP, dissolved oxygen, conductivity, and chloride for sites in the area of wastewater discharge (Table 4 & 5) were higher than at background sites (see "U" upstream sites in Table 4 & 5) and pre-effluent conditions (Table 2) at the same sites. Values were also higher at the sites in the area of wastewater discharge than the control sites U1200 and U400 of the same year. The decreases in measured values with distance from effluent outfall followed a quadratic pattern in all parameters monitored, with possible exceptions of BOD and DO. Sharp increases were observed at D25, the point of effluent outfall, with gradually decreasing concentrations downstream. At the point of discharge into Whortleberry Creek (Site D2600), all parameters were close to background levels.

Problems arose with measurements of some parameters due to large particles, often including vegetation, unavoidably being collected with water samples. Biological oxygen demand measurements were particularly affected and are therefore suspect. Filtering of water samples was used where permissible for other parameters (APHA 1980). Mean dissolved oxygen did not show the quadratic trend of concentration decrease with distance from outfall as clearly as the other parameters. Measurements were affected by time of day, presence or absence of vegetation, and water depth. Dissolved oxygen followed a similar pattern during the pre-effluent and 1985-86 sampling periods at the control sites U1200, U400, and U25, although elevated during the 1985-86 post-effluent period. Increased concentrations of nitrogen, dissolved oxygen, and biological oxygen demand at site U25 suggest possible contamination by wastewater. (Note: this site is directly adjacent [upstream] to the wastewater outfall and may have inadvertently been exposed to wastewater addition.) For the other sites, increased DO may have been an artifact of a different meter being used in a different year.

Measurements of water chemistry samples taken between site D900 and D2600 indicate increase of concentrations of all parameters above those of site D900 during the sampling period. This indicates that site D900 is an area in which wastewater is not directly contacting the wetland, either because of surface water flow patterns or possible input of groundwater leading to dilution of wastewater.

Low coefficient of determination values obtained from regression of nutrient concentrations versus distance from wastewater inflow were due in large measure to the anomalously low concentrations for all parameters, including conductivity and chloride "tracers," by an order of magnitude. At 1000 m and beyond, concentrations rose again to expected levels. A possible explanation for the low concentrations between 700 m and 900 m is that sampling stations chosen in this range were located in "dead spots" within the wetland, and wastewater flow was around rather than through these sites. It is also possible that a substantial groundwater influx occurred between 700 and 900 m in the wetland. Other samples that produced outlying data (detrimental to regression analysis) were collected on 7 September 1985, immediately following a hurricane which passed through the study area and deposited 25.9 cm of rain

Table 4. Cumulative mean and standard error for surface water chemistry parameters measured monthly during the **1985-86 post-effluent discharge period** (period immediately after initiation of wastewater discharge) at upstream (background) and downstream sampling locations in the titi shrub swamp near Apalachicola, Florida. Sites donated by "U" are upstream background sites. Pond "is at point of wastewater Appalachicola, Florida sites denoted by "D" are the number of meters downstream from pont of discharge.

Site		Temp. (deg. C)	Turbid. (ntu)	Tot.Phos. (mg/L)	TKN (mg/L)	B.O.D. (mg/L)	pH	Diss.Ox. (mg/L)	Conduct. (umho/cm)	Chloride (mg/L)	Orthophos. (mg/L)	Ammonia (mg/L)	Nitrate/Nitrite (mg/L)
U1200	MEAN	15.8	1.5	0.19	1.45	2.9	3.7	3.8	130	28	0.000	0.03	0.00
	SE	1.5	-	0.08	0.62	1.3	0.2	0.5	13	5	-	0.01	0.00
U400	MEAN	14.5	1.6	0.03	0.92	3.0	3.7	3.3	87	16	0.000	0.03	0.00
	SE	1.5	-	0.02	0.27	1.9	0.1	1.8	8	3	-	0.01	0.01
U25	MEAN	16.3	1.8	0.09	2.24	3.5	4.2	6.9	122	21	0.020	0.03	0.00
	SE	0.8	-	0.04	-	2.6	0.2	2.6	-	1	-	0.01	0.01
POND	MEAN	23.2	-	2.40	3.56	7.4	7.4	6.3	1103	227	0.980	1.44	3.31
	SE	1.6	-	0.24	0.38	1.8	0.3	1.5	95	30	-	0.52	1.67
D25	MEAN	29.9	-	2.10	3.44	4.6	6.6	6.8	1075	220	-	-	-
	SE	1.1	-	0.20	0.44	0.6	0.2	0.9	105	33	-	-	-
D200	MEAN	21.7	-	1.13	2.74	4.2	5.95	4.16	896	192	-	0.83	0.08
	SE	0.9	-	0.15	0.36	1.35	0.28	0.89	77	32	-	0.25	0.05
D400	MEAN	17.8	-	1.22	2.22	3.0	5.9	4.0	749	147	-	0.34	0.03
	SE	1.1	-	0.11	0.29	1.0	0.2	0.7	96	22	-	0.11	0.01
D900	MEAN	19.6	2.3	0.07	1.25	3.7	4.4	3.7	288	80	0.020	0.09	0.01
	SE	1.5	-	0.01	0.27	0.8	0.3	0.6	54	15	-	0.06	0.00
D1500	MEAN	-	-	-	-	-	-	-	-	-	-	-	-
	SE	-	-	-	-	-	-	-	-	-	-	-	-
D1800	MEAN	-	-	-	-	-	-	-	-	-	-	-	-
	SE	-	-	-	-	-	-	-	-	-	-	-	-
D2700	MEAN	16.8	1.1	0.07	1.03	2.1	4.4	4.4	281	83	0.020	0.13	0.00
	SE	1.6	-	0.01	0.19	0.8	0.3	0.5	32	11	-	0.06	0.00

Table 5. Cumulative mean and standard error for surface water chemistry parameters measured monthly during the **1987 post-effluent discharge period** at upstream (background) and downstream sampling locations in the titi shrub swamp near Apalachicola, Florida. Sites donated by "U" are upstream background sites. Pond "is at point of wastewater Appalachicola, Florida sites denoted by "D" are the number of meters downstream from pont of discharge.

Site		Temp. (deg. C)	Turbid. (ntu)	Tot.Phos. (mg/L)	TKN (mg/L)	B.O.D. (mg/L)	pH	Diss.Ox. (mg/L)	Conduct. (umho/cm)	Chloride (mg/L)	Orthophos. (mg/L)	Ammonia (mg/L)	Nitrate/Nitrite (mg/L)
U1200	MEAN	14.8	2.7	0.31	1.42	1.6	3.5	3.4	482	76	0.013	0.08	0.01
	SE	2.1	1.0	0.14	0.40	1.4	0.1	0.7	137	22	0.005	0.03	0.00
U400	MEAN	15.4	3.2	0.17	1.39	0.7	3.6	4.9	157	45	0.008	0.10	0.01
	SE	2.4	0.9	0.09	0.57	0.6	0.1	0.7	30	16	0.004	0.04	0.00
U25	MEAN	12.6	2.7	0.21	0.89	0.7	4.0	5.7	69	18	0.014	0.18	0.01
	SE	2.2	0.5	0.19	0.52	0.0	0.3	1.2	16	4	0.007	0.08	0.00
POND	MEAN	20.5	8.4	2.48	2.24	2.2	8.3	14.0	1215	309	1.963	0.39	4.79
	SE	1.9	1.8	0.92	0.49	0.7	0.2	1.9	204	119	0.270	0.10	1.02
D25	MEAN	19.5	9.0	2.63	2.89	2.0	7.8	10.1	1062	281	2.149	0.54	4.77
	SE	1.7	1.8	0.66	0.50	1.5	0.1	1.7	161	91	0.202	0.14	0.73
D200	MEAN	-	-	-	-	-	-	-	-	-	-	-	-
	SE	-	-	-	-	-	-	-	-	-	-	-	-
D400	MEAN	17.8	6.5	0.83	1.54	1.5	6.4	2.4	698	209	0.680	0.20	0.11
	SE	1.8	1.0	0.24	0.34	1.0	0.3	0.5	156	71	0.194	0.05	0.07
D900	MEAN	17.1	4.6	0.70	1.20	0.2	6.6	5.3	599	190	0.543	0.19	0.02
	SE	2.1	0.7	0.16	0.38	0.1	0.1	1.0	110	63	0.085	0.06	0.00
D1500	MEAN	16.26	5.77	0.72	2.30	1.4	6.1	2.1	391	127	0.375	0.46	0.01
	SE	1.80	0.89	0.19	0.59	0.8	0.2	0.4	54	34	0.041	0.20	0.00
D1800	MEAN	16.0	3.8	0.60	1.90	0.8	6.0	3.3	323	111	0.311	0.29	0.03
	SE	2.0	1.0	0.21	0.48	0.8	0.3	0.5	45	28	0.040	0.11	0.02
D2700	MEAN	14.8	2.2	0.66	1.63	0.3	6.0	5.2	278	82	0.265	0.20	0.18
	SE	1.7	0.4	0.23	0.46	0.1	0.1	0.6	33	18	0.036	0.06	0.08

(30 August to 1 September 1985). All parameters for 7 September 1985 samples had significantly lower concentrations, presumably because of dilution by rainwater.

Groundwater Chemistry

Several shallow wells 2.0 - 4.5 m deep were installed at the wetland by the USGS (Figure 2). Gamma-ray well logs (measuring natural radioactivity of rocks and sediments in a bore hole) performed in the area showed that the mineral soil graded into clay at about 6.8 m below the surface (Best *et al.* 1983). Groundwater flow was in an east-southeast direction (Best *et al.* 1983). Chloride and conductivity increases indicated that wastewater had reached one well (well B), which was adjacent to the discharge area. Other wells have not been impacted thus far.

Nitrate-nitrogen was the only parameter monitored that was of public health concern. The possibility of migration of nitrate to groundwater was examined, and during the course of the study nitrate-nitrogen levels remained well below the US EPA 10 mg/l regulatory standard. Post-effluent concentrations of NO_3+NO_2 in groundwater were equivalent to background levels at the 99% confidence level. Phosphorus, TKN, and NH_3 levels in groundwater also remained within the natural background range.

Chloroorganics in a Model Forested Wetland

Chlorophenols have been found in chlorinated wastewater effluent (Garrison *et al.* 1978) an may pose impacts to receiving water bodies. With the advent of wetland use for tertiary treatment of wastewater the study of chlorophenolic behavior in a receiving wetland is of interest and should be investigated before the practice becomes more widespread. A widely published fugacity-based fate mode (Mackay 1979; Mackay and Paterson 1981) was used by Sonnenberg (1984) to evaluate the fate of chloroorganics in forested wetlands. The accuracy and efficacy of the model was assessed for prediction of the behavior of a model chlorophenolic compound, 2,4 dichlorophenol (DCP) (an EPA priority pollutant), in laboratory microcosms that simulated forested wetlands. A chlorophenol was selected for study because, as indicated, the occurrence of these compounds has been found in wastewater and because of the similarity of the phenolic moiety to some of the polymeric subunits of humus. Humic substances in wastewater are susceptible to chlorination, and phenols in particular are easily chlorinated.

There were two phases of experimentation where DCP was added to the laboratory microcosms. The first phase involved a system in static equilibrium (level I), which provided preliminary information about the significance of suspended solids and the migration of DCP through the detritus (litter). The second phase involved a system in a steady-state equilibrium and thus describes degradation of the compound as outlined by the level II model. Three microcosms were set up in each phase: two experimental and one control. The microcosms consisted of 40 l aquaria containing detritus and water from a cypress dome with the following mean water quality characteristics (Dierberg and Brezonik 1980):

pH	4.5
Dissolved Oxygen (mg/l)	2.0
BOD (mg/l)	2.9
Total Organic Carbon (mg/l)	39.9
Color (c.v.)	456.0

The characteristics of this cypress dome are very similar to the titi shrub swamp in Apalachicola (see Table 2).

An in-depth description of the experimental procedures, sampling methodology, and analytical procedures for both the static microcosm (level I) and the steady-state microcosm (level II) experiments are presented in Sonnenberg (1984) along the with model variables determined from site-specific and chemical-specific data. For the static system both the predicted model results and the experimental results indicate that the bulk of DCP resided in detritus (Table 6). The experimental water concentration, amount, and percent of total were almost an order of magnitude larger than those predicted by the model. Model values and experimental results were closer for the detritus samples. The model predicted a higher concentration and higher amount of DCP in the detritus by about 1.5 times. For the steady-state flow through systems, the distribution of DCP was also primarily into the detritus (Table 6). The quantity and concentration of DCP in the open water was about 10 times that predicted by the model, as with the static systems. However, the theoretical and experimental quantities and concentrations in the detritus were almost exactly the same. The model prediction of degradation in the detritus, based on rate constants derived from literature data, was more rapid than actually occurred.

Both model and experimental results indicate that the primary compartment of residence of DCP is detritus. This partitioning of DCP into the detritus from the water column appears to be a rapid and extensive process. There was less partitioning of DCP from the water to the detritus than predicted by the model, leaving higher DCP concentration in the water column in both systems. Probable causes ofthese discrepancies were the high aqueous solubility of DCP and the large particle size of the detritus compared to soil and sediments from which model variables were selected. In spite of the difference between model and experimental distribution ratios, the results indicate that detritus and sediment in wetlands will provide a sink for chlorinated organics that have structures similar to DCP, provided that there is adequate contact time between surface water and litter. It can be expected that in systems with particulate matter that is rich in organic carbon, these compounds will exhibit a high affinity for that particulate matter. It is also reasonable to expect that the higher the humus and organic carbon content of the abiotic phases, the greater the portioning of DCP into the litter and particulates will be.

In wetlands systems such as a cypress dome or a titi shrub swamp, it is reasonable to expect microbiological degradation of DCP. Anaerobic microbe degradation has been reported to be an

Table 5a. Model vs. experimental values of the distribution of DCP to water and sediment in static microcosms.

	Water		Detritus	
	Experimental	Model	Experimental	Model
Concentration	90. ug/L	11. ug/L	2.9 ug/g	4.3 ug/g
Amount	1.26 mg	0.15 mg	7.41 mg	10.95 mg
% of amount introduced	11.	1.	66.	98.
% of amount recovered from water and detritus	15.	1.	85.	99.

Table 5b. Model vs. experimental values of the distribution of DCP to water detritus in flow-through microcosms.

	Water		Detritus	
	Experimental	Model	Experimental	Model
Concentration	21. ug/L	2. ug/L	0.80 ug/g	0.79 ug/g
Amount	288. ug	28. ug	1.98 mg	1.96
% of amount introduced[a]	1.	<1.	6.	6.
% of total recovered	13.	1.	87.	98.[b]

[a]Amount introduced--33.3 mg

[b]Remaining 1% assumedly partitioned to the air compartment not measured.

important degradation pathway in sediment (Suflita *et al.* 1982). Indeed, in this investigation that type of degradative mechanism may have been the most significant removal pathway in the system, as shown by experimental and, particularly, model results. In addition to the physicochemical affinity of DCP for the detritus, it appears from model results that most biological transformations will occur in the detritus and, overall, detrital biodegradation is the most significant transformation process. Chemical degradation in water and detritus is probably not as significant as biological degradation in either compartment. In summary, the fugacity model results are within the specifications for validity of evaluative models (with an order of magnitude of actual values). They are easily used and can provide important evaluative information about a chemical in a specific or general environment.

BENTHIC MACROINVERTEBRATES

Purpose

The major purpose of this study was to determine whether or not studies of benthic macroinvertebrates can be useful in quantitatively assessing the effects of wastewater on a wetland system, and whether or not changes in benthic community structure are indicators of organic pollution in these systems.

Questions that were addressed in this study included:

1. Does abundance and/or diversity of macroinvertebrates change when wastewater is added to a wetland?
2. What changes (if any) in taxonomic composition of the macroinvertebrate assemblage are observed?
3. Can the use of functional feeding guilds (Merritt and Cummins 1978) provide information on changes in community structure of macroinvertebrates resulting from wastewater addition?
4. Are changes in the water chemistry parameters TN, TP, BOD, pH, and DO directly or indirectly (through stimulation of primary productivity) responsible for changes in macroinvertebrate abundance or diversity?
5. Is benthic macroinvertebrate response a reliable indicator of organic pollution in this system?

Literature Review

Benthic macroinvertebrates have historically been used as indicators of pollution in aquatic systems such as streams and lakes (Richardson 1921; Patrick 1949; Gaufin and Tarzwell 1952, 1955, 1956), but a limited amount of work has been done in this area in wetland systems (Brightman 1976; Schwartz 1980). An indictor organism can be defined as a plant or animal, the presence or absence of which is indicative of some fact or facts with respect to its environment (Beck 1954). When an environment os altered, the degree of alteration logically should be demonstrable by a shift in population structure or density of properly selected indicator organisms (Beck 1954). Although several groups of organisms have been suggested for water quality monitoring, macroinvertebrates are often used for this purpose.

Since macroinvertebrates as a group are sensitive to habitat alteration, especially of constituents typically found in municipal wastewater, analysis of ranges in abundance and distribution of macroinvertebrate populations may provide data of significant importance in environmental impact assessment (Resh and Price 1984).

Chemical-physical analyses of water samples would seem to be the most direct method of detecting water pollution (Milbrink 1973). However, pollutants released intermittently or which undergo dilution effects, are easily missed by sampling designs unless samples are continually obtained at very short intervals, which is both time-consuming and expensive. A biological analysis is often the most direct assessment of pollution in such situations (Milbrink 1973).

There are several reasons why bottom fauna are studied (Cairns and Dickson 1971):

1) Many species are extremely sensitive to and respond quickly to alteration of water quality.

2) Benthic fauna usually have a complex life cycle of a year or more, and if at any time during their life cycle environmental conditions are outside the tolerance limits, the population may be stressed.

3) Since they exist in a more or less restricted area and are not subject to rapid migrations, they serve as natural monitors of water quality.

Gaufin and Tarzwell (1952, 1955, 1956) were among the first to use aquatic invertebrates in stream pollution investigations. However, their application of the indicator organism idea stressed the interference of other factors. These will be discussed later.

Historically, investigators have tried to relate presence or absence of specific indicator species to varying degrees of water quality (Richardson 1921; Patrick 1949; Gaufin and Tarzwell 1952, 1955, 1956). According to this approach, organisms which are traditionally considered to be "clean-water" organisms are found in regions of high water quality. Conversely, pollution tolerant organisms are found in areas of poor water quality (Gaufin and Tarzwell 1952).

Diversity Indices

Because of the limitations or the indicator species approach, consideration of species abundance in the community became important (Ruggiero and Merchant 1979). According to Wilhem and Dorris (1968), the structure of a benthic community can be summarized in diversity indices derived from information theory, first suggested by Margalef (1958).

Various investigators have shown a negative correlation between species diversity and environmental stress (Itow 1963; Gordon and Gorham 1963; Woodwell 1967; Wilhm and Dorris 1966; 1968, Syke 1968; Nutall 1972; Nutall and Purves 1974; Ruggiero and Merchant 1979). Wilhm and Dorris (1966) proposed values of diversity, D, in the biotic community of less than 1 in areas of heavy pollution, values from 1 to 3 in areas of moderate pollution, and values greater than 3 in clean water areas. These values for diversity indices, however, may not hold true for wetland systems. Shepard (1984) maintains that a log series distribution (Pielou 1977) is most appropriate for summarizing ecological diversity, rather than indices based on information theory, for describing population responses to stress. Traditional diversity indices assume that species in the sample are represented proportionally to their true abundances, that "Community" spatial boundaries can be precisely defined, and that all taxa interacted ecologically before collection. These assumptions frequently cannot be fulfilled in aquatic studies (Shepard 1984). Some of the assumptions of the log series distribution are 1) abundances of samples vary greatly, 2) the number of species not collected is unknown, 3) most species are rare, and 4) the samples have high heterogeneity. All of these assumptions are appropriate for benthic macroinvertebrate data (Shepard 1984).

Non-Effluent Factors

For this study, it was appropriate that factors other than effect of effluent discharge be taken into account when analyzing benthic macroinvertebrate data. Two of the most important of these are the type of substrate and vegetation found at different sites. Many studies have shown the relationship of organisms to substrate (Ruggiero and Merchant 1979). All of these studies support the hypothesis that absence or presence of certain organisms could be due to something other than water quality, namely adequate substrate (Ruggiero and Merchant 1979). Substrate differences between sites can have an effect on the diversity of species by changing the frequencies at which the various kinds of organisms occur (Ruggiero and Merchant 1979).

Changes in abundance and of higher plants, particularly submergents, will have profound effects on the species composition of benthic invertebrates (Schwartz 1980). Lack of appropriate habitat, and not water quality, may also be a cause for not finding "clean water" organisms (Arthur and Horning 1969).

Voights (1976) studies the relationship between invertebrate populations and vegetative cover. Total invertebrate abundance increased as the emergent vegetation was replaced by submerged vegetation, but maximum numbers occurred where beds of submerged vegetation were interspersed with stands of emergent vegetation. He found that Dipteran midges reached greatest abundance in more open habitats somewhat protected from the wind, whereas amphipods were most numerous in dense beds of submerged vegetation. In open water areas, increases in phytoplankton production increases the quantity of food in the form of detritus reaching the bottom. These changes will also lead to a difference in species composition of benthic invertebrates which might be quantitatively related to increased eutrophication (Fruh *et al.* 1966; Schwartz 1980).

Another factor which may influence the evaluation of benthic samples is the method used to collect them. Sampling variability can be high when the organisms encountered have aggregated distributions, and consequently, many sample replicates may be required (Resh and Price 1984). Comparisons of different methods have yielded differing numbers of individuals and species of identical sites (Tsui and Breedlove 1978; Freeman *et al.* 1984).

In stream studies, the nature of the stream location, type, and number of tributaries, and the character of the watershed are all important in determining the makeup of aquatic populations (Gaufin and Tarzwell 1955). The nature of food available is dependent on the nutrient input from the surrounding watershed in a wetland as well, and greatly influences benthic invertebrate occurrence. This might be a more important factor in nutrient limited environments. For example, Cowell and Vodopich (1981), studying distribution and seasonal abundance of benthic macroinvertebrates in a hypereutrophic lake in subtropical Florida, found that the benthic community was not limited by food quantity and that the insect species are multivoltine and capable of continuous growth and development. Apparently, temperature and food abundance do not have as much influence on seasonal abundance in eutrophic, subtropical systems (Cowell and Vodopich 1981). This confirms a general philosophy that it is unwise to make generalizations about aquatic systems based on data exclusively from temperate regions.

Another important factor which can greatly influence the interpretation of results in benthic studies is weather patterns. For example, spring flooding may cause drastic changes in spatial distribution of invertebrates. Density estimates made during these periods may be strongly influenced by water depth and quantity, and decreased density should not automatically be assumed to be the result of mortality (Resh 1979). Sampling during or immediately after extreme events such as drought or hurricanes may give insights into not only the effect of such events on aquatic biota but also on the factors regulating the colonization and distribution of benthic populations or communities (Resh 1979).

Seasonal and yearly variation in larval assemblages are related to insect emergence and life cycle patterns. These cycles can cause even more variation in macroinvertebrate data (Merritt and Cummins 1978).

Effluent-Related Factors

The natural factors that can influence macroinvertebrate distributions have been briefly discussed. Factors that can only be attributed to changes in water quality as a result of wastewater addition can now be examined. Specific changes in water quality include increased nitrogen and phosphorus, leading to increased primary productivity in the form of algal growth, increased suspended solids in the water column due to decaying material, increased chloride concentrations, and changes in biological oxygen demand and dissolved oxygen. In acid water systems, elevation of pH that occurs with wastewater addition may exclude species adapted to acidic waters.

Most forms of stress (pollution) reduce aquatic ecosystem complexity (Cairns and Dickson 1971). Introduction of a pollutant reduces the number of species by eliminating those sensitive to the pollutant until only those organisms that can survive the adverse conditions remain (Cairns and Dickson 1971). Certain groups of organisms have been found to be intolerant of various types of pollution. Cairns and Dickson (1971) provide a good summary of pollution tolerant and pollution sensitive organisms. Pollution tolerant organisms include midge larvae, leeches, and oligochaetes, while pollution sensitive organisms include mayflies, stoneflies, caddisflies, and some dragon flies. Tolerant organisms are found in both clean and polluted situations and their presence does not mean a body of water is polluted. However, a population of tolerant organisms combined with an absence of intolerant organisms is a good indicator of pollution. These ideas should be applied with caution to wetland systems. The limited amount of work done seems to imply that completely unpolluted systems may be dominated by organisms such as midges which were previously thought to be associated with polluted systems (Haack 1984; Brightman 1976).

Functional Feeding Guilds

The use of functional feeding group designations (Merritt and Cummins 1978) has enabled ecologists to group macroinvertebrate taxa into communities based on feeding habit. Feeding guilds include shredders, which use living vascular hydrophyte and decomposing vascular plant tissues. In stream studies, these would be expected to occur in the headwaters of a stream (stream order 1-3), where tree canopy extends over the stream and large size detritus particles are available. Collectors feed on decomposing fine particulate organic matter and would be expected to occur below the headwaters of a stream. Herbivores, or grazers, feeding on periphyton and living plant tissue, predators, feeding on live animal tissue, and filterers, which filter particles through specially adapted anatomical features, are the other feeding guilds. Generalists are a group which can adopt a variety of feeding modes. Traditional ideas of changes in benthic macroinvertebrate community structure with distance from a point source in a polluted stream (Wilhm and Dorris 1968) may not hold true for wetlands, which have naturally low dissolved oxygen levels at times. Haack (1984) found a notable absence of the shredder guild within this wetland site, which has an almost continuous canopy cover at most locations.

Methods

Sampling methods for the benthic macroinvertebrates were identical to those of Haack (1984). A sampler was built with dimensions of (0.3 m x 0.3 m x 0.65 m). During sampling, the sampler enclosed a sediment and water-column for each sample. The volume of water enclosed was swept approximately 10 times with a 0.2 mm mesh net. Net contents were washed into a bottle to which rose bengal stain in a formalin base was added for preservation and staining. All living plant material and all detritus to a depth of approximately 10 cm was collected by hand from within the enclosure and placed in plastic bags. Two replicate samples were taken at each sampling location at specific time intervals (see "Study Site Description: section). Samples were transported on ice to the lab, where they were washed through a U.S. No. 30 sieve (600 μm). Organisms were separated from substrate and preserved in 95% ethanol until identification. Invertebrates were identified at least to family, except in the case of the Chironomidae, which were identified to genus. Keys used included Mason (1973), Merritt and Cummins (1978), Oliver *et`al.* (1978), Pennak (1978), and Lemkuhl (1979).

Samples collected at the wastewater-impacted sites contained numerous Chironomidae, which created a practical problem in terms of identification and counting. After some unsuccessful attempts at subsampling unsorted material, due to clumping of organisms, it was decided to sort all material. However, due to time limitations, a subsample of Chironomidae in samples with large numbers was taken. A limit of 400 randomly sampled Chironomidae per sample were identified to the genus level, and a ratio was established for the genera encountered. All individuals were then counted and the ratio applied. The number 400 was chosen from an experimental count of one large sample, during which a tally of ratios was kept during counting. No significant changes in ratio occurred after 400 individuals.

Dissolved oxygen was measured with a YSI meter model 54. The entire water column was measured, i.e., no attempt was made to determine dissolved oxygen at different water column levels. The pH was measured with an Orion pH meter model 407A. Biological oxygen demand was done using a YSI DO meter (EPA 1973). Analyses of conductivity, total phosphorus, total Kjeldahl nitrogen, and chloride employed standard techniques (APHA 1980).

Diversity values for invertebrates were computed for pre-effluent and post-effluent data according to Odum (1971), substituting number of groups for number of species. Groups were represented by the lowest taxonomic level of identification that was possible. These values are comparable between pre-effluent and post-effluent data since the same level of taxonomic identification was achieved for both sets of data. The Shannon-Weaver index, $H = -\Sigma(p_i) \times (\ln p_i)$, with p_i representing the number of individuals of a particular group divided by the total number of individuals (proportion) in the mean of the two replicate samples taken at each site, is a diversity index commonly used to express community structure. Dominance, defined as the degree to which occurrences are concentrated in one, several, or many species, sums each species' importance in relation to community as a whole (Odum 1971). Dominance was calculated as p_i^2. Evenness, or the apportionment of individuals among the groups found, was calculated as $H/\ln S$, with H being the Shannon-Weaver value and S being the number of groups found in the sample (Odum 1971). Two indices of species richness, $S\text{-}1/\ln N$, and $S/\sqrt{N}$, where N is the total number of individuals in a mean of two samples, were calculated (Odum 1971). Species richness refers to the occurrences of many different species or just a few found within a sample.

Statistical analyses were performed using the Statistical Analyses Package (SAS, Fifth Edition, 1985) at the Northeast Regional Data Center on the University of Florida campus. Two-way ANOVA and Student's T-test were used in interpreting the data.

Results and Discussion

Macroinvertebrate Composition

Study sites were grouped into three categories for the purpose of comparisons (Tables 7a-7c). Control sites included sites U1200, U400, and U25. Sites D25, D200, and D400 were grouped as near-discharge experimental sites. Downstream experimental sites included sites D900 and D2600. Dominant taxa were designated as those comprising greater than 10% percent of individuals noted at astation during the sampling period (Table 8). This time period encompassed 3 mo of pre-effluent sampling in addition to the post-effluent sampling.

Among the control sites, the Chironomidae (Diptera) *Chironomus*, *Procladius*, *Rheotanytarsus*, and *Polypedilum* were dominants. Also dominant was the Crustacean *Asellus*, the Coleopteran *Hydrophilidae*, and Oligochaeta. Twelve families were found at site U1200, 11 at site U400, and 12 at site U25 during sampling from March 1985 to May 1986. This compares with 26 families at site U1200, 12 at site U400, and 13 at site U25 during pre-effluent sampling (Haack 1984).

Dominants occurring at near-discharge sites included the Chironomidae *Chironomus* and *Kiefferulus*, Culicidae, and Ephydridae (all of the order Diptera). Culicidae, which is comprised of the mosquitoes, occurred as a dominant on only one sampling occasion. Increases in mosquito populations have been cause for concern in some wastewater application to wetland studies (Mood 1976), but a number of studies (Davis 1984; Small 1976b) have shown no significant increase in mosquito populations. The Oligochaeta were the only other dominant organism at these sites. Twenty-one families were observed at site D25, 17 at site D200, and 18 at site D400. This compares to 11 families observed at site D25 during the pre-effluent study (Haack 1984). Excluding the 3 mo of sampling during the spring of 1985, when wastewater was not yet entering the wetland, 16 families were observed during the remainder of the sampling period at site D25. This is the same length of time as the pre-effluent study (Haack 1984) with a greater number of families being observed. Families observed during pre-effluent sampling were not all the same as those observed during post-effluent sampling at site D25. In particular, there was an increase in the number of Dipteran families and a loss of all Crustacean families at this site. The remainder of the increase was due to addition of Ephemerellidae (Ephemerella), Gomphidae (Odonata), Coenagriondae (Odonata), and *Physa*, (Castropoda).

Table 7A. Macroinvertebrate occurrences by site (March 1985-May 1986) in a forested wetland near Apalachicola, Florida. UPSTREAM (CONTROL) SITES.

UPSTREAM (CONTROL) SITES		
SITE U1200	SITE U400	SITE U25
Annelida	Annelida	Annelida
*Oligochaeta	Oligochaeta	*Oligochaeta
Coleoptera	Coleoptera	Coleoptera
Dytiscidae	Dytiscidae	Dytiscidae
*Hydrophilidae		Hydrophilidae
	Collembola	
	Isotomidae	
Crustacea	*Isotomurus palustris*	Crustacea
Asellidae		Asellidae
Asellus	Crustacea	*Asellus*
Gammaridae	Asellidae	Gammaridae
Crangonyx	**Asellus*	*Crangonyx*
Cambarinae	*Lirceus*	
Procambarus	Gammaridae	Diptera
	Crangonyx	Ceratopagonidae
Diptera	Cambarinae	Chaeboridae
Ceratopogonidae	*Procombarus*	Chironoidae
Chironomidae		**Chironomus*
**Chironomus*	Diptera	*Cricotopus*
Einfeldia	Ceratopogonidae	*Kiefferulus*
Kiefferulus	Chaeboridae	*Polypedilum*
Labrundinia	Chironomidae	*Tribelos*
Polypedilum	**Chironomus*	*Ephydridae*
**Procladius*	*Dicrotendipes*	*Tabanidae*
**Rheotanytarsus*	*Glyptotendipes*	
Tanypus	*Kiefferulus*	Odonata
	Microtendipes	Gomphidae
Ephemeroptera	**Polypedilum*	
Caenidae	*Procladius*	Trichoptera
Caenis	*Rheotanytarsus*	Leptoceridae
	Stenochironomus	
Hemiptera	Dolichopodidae	
Corixidae		
	Hydracarina	
Megaloptera		
Sialidae		
Sialis		

Table 7A. (continued.)

UPSTREAM (CONTROL) SITES		
SITE U1200	SITE U400	SITE U25
Odonata Libellulidae <u>Pachydiplax longipennis</u>		
(14 samples) (12 families)	(12 samples) (11 families)	(6 samples) (12 families)

*Indicates invertebrate groups comprising greater than 10% of individuals noted at a station.

Table 7B. Macroinvertebrate occurrences by site (March 1985-May 1986) in a forested wetland near Apalachicola, Florida. NEAR-DISCHARGE (EXPERIMENTAL) SITES.

NEAR-DISCHARGE (EXPERIMENTAL) SITES		
SITE D25	SITE D200	SITE D400
Annelida	Annelida	Annelida
Oligochaeta	*Oligochaeta	Oligochaeta
Coleoptera	Coleoptera	Coleoptera
Dytiscidae	Dytiscidae	Dytiscidae
Hydrophilidae	Hydrophilidae	Helodidae
		Hydrophilidae
Collembola	Crustacea	
Sminthuridae	Asellidae	Crustacea
Isotomidae	Asellus	Gammaridae
	Gammaridae	Crangonyx
Crustacea	Crangonyx	
Asellidae		Diptera
Asellus	Diptera	Ceratopogonidae
Gammaridae	Ceratopogonidae	Chironomidae
Crangonyx	Chironomidae	*Chironomus
Cambarinae	*Chironomus	*Kiefferulus
Procambarus	Einfeldia	Polypedilum
	Glyptotendipes	Procladius
Diptera	*Kiefferulus	Pseudochironomus
Ceratopogonidae	Polypedilum	Rheotanytarsus
Chironomidae	Procladius	Tanypus
*Chironomus	Rheotanytarsus	Culicidae
Crypotochironomus	Tanypus	Ephydridae
Dicrotendipes	*Culicidae	Stratiomyidae
Glyptotendipes	Ephydridae	Syrphidae
*Kiefferulus	Stratiomyidae	Eristalis
Parachironomus	Syrphidae	Tabanidae
Procladius	Eristalis	
Rheotanytarsus		Ephemeroptera
Stenochironomus	Gastropoda	Ephemerellidae
Tanypus	Physa	Ephemerella
Dolichopodiadae		
Ephydridae	Hemiptera	Gastropoda
Stratiomyidae	Belostomatidae	Physa
Syrphidae	Mesoveliidae	
Eristalis		Hemiptera
	Odonata	Nepidae
Ephemeroptera	Libellulidae	Belostomatidae
Ephemerellidae	P. longipennis	
Ephemerella	Coenagrionidae	Odonata

Table 7B. (continued.)

NEAR-DISCHARGE (EXPERIMENTAL) SITES		
SITE D25	SITE D200	SITE D400
		Libellulidae
Gastropoda	Trichoptera	*P. longipennis*
Physa	Leptoceridae	Coenagrionidae
Hemiptera		
Corixidae		
Lepidoptera		
Pyralidae		
Odonata		
Coenagrionidae		
Gomphidae		
Libellulidae		
P. longipennis		
(30 samples)	(24 samples)	(20 samples)
(21 families)	(18 families)	(18 families)

*Indicates invertebrate groups comprising greater than 10% of individuals noted at a station.

Table 7C. Macroinvertebrate occurrences by site (March 1985-May 1986) in a forested wetland near Apalachicola, Florida. DOWNSTREAM (EXPERIMENTAL) SITES.

DOWNSTREAM (EXPERIMENTAL) SITES

SITE D900

Annelida
- *Oligochaeta

Coleoptera
- Dytiscidae
- Hydrophilidae

Crustacea
- Asellidae
  - *Asellus*
- Cambarinae
  - *Procambarus*
- Gammaridae
  - *Crangonyx*

Diptera
- Ceratopogonidae
- Chironomidae
  - **Chironomus*
  - *Cricotopus*
  - *Cryptochironomus*
  - *Dicrotendipes*
  - *Glyptotendipes*
  - **Kiefferulus*
  - *Microtendipes*
  - *Polypedilum*
  - *Procladius*
  - *Rheotanytarsus*
  - *Tanypus*
- Dolichopodidae

Hemiptera
- Notonectidae
  - *Buenoa*

Hydracarina

Megaloptera
- Sialidae
  - *Sialis*

Trichoptera
- Molannidae

SITE D2150

Annelida
- Oligochaeta

Coleoptera
- Asellidae
  - **Asellus*
  - *Lirceus*
- Cambarinae
  - *Procambarus*
- Gammaridae
  - *Crangonyx*

Crustacea
- Dytiscidae
- Elmidae
- Gyrinidae
- Hydrophilidae

Collembola
- Isotomidae

Diptera
- Ceratopogonidae
- Chironomidae
  - **Chironomus*
  - *Cricotopus*
  - *Cryptochironomus*
  - *Kiefferulus*
  - *Labrundinia*
  - *Paralauterborniella*
  - *Polypedilum*
  - *Procladius*
  - *Rheotanytarsus*
  - *Tanypus*
  - *Tribelos*
- Dolichopodidae
- Simulidae
- Tabanidae

Hemiptera
- Hebridae

Odonata

Table 7C. (continued.)

DOWNSTREAM (EXPERIMENTAL) SITES	
SITE D900	SITE D2150
Molanna	Cordulegastridae
	Cordulegaster
	Gomphidae
	Libellulidae
	P. longipennis
	Plecoptera
	Trichoptera
	Calamoceratidae
	Leptoceridae
	Mollanidae
	Molanna
(28 samples)	(14 samples)
(13 families)	(19 families)

*Indicates invertebrate groups comprising greater than 10% of individuals noted at a station.

Table 8. Dominant taxa of aquatic macroinvertebrates found at eight sites in a forested wetland near Apalachicola, Florida, from March 1985 to May 1986. The time period included three months of pre-effluent observations. Dominant taxa are those comprising greater than 10% of individuals noted at a station during the sampling period.

	Upstream (Control) Sites			Near-Discharge (Experimental) Sites			Downstream (Experimental Sites)	
	U1200	U400	U25	D25	D200	D400	D900	D2600
Families	12	11	12	21	17	18	13	19
Chironomid genera	8	9	5	10	8	7	11	11
Number of samples	14	12	6	24	24	20	22	6
Dominants	Oligochaeta	*Asellus*	Oligochaeta	*Chironomus*	Oligochaeta	*Chironomus*	Oligochaeta	*Asellus*
	Hydrophilidae	*Chironomus*	*Chironomus*	*Kiefferulus*	*Chironomus*	*Kiefferulus*	*Chironomus*	*Chironomus*
	Chironomus	*Polypedilum*			*Kiefferulus*		*Kiefferulus*	
	Procladius				Culicidae			
	Rheotanytarsus							

Dominants at downstream experimental sites included *Chironomus*, *Kiefferulus*, *Asellus*, and Oligochaeta. Of the downstream sites, 13 families were collected at site D900 and 19 at site D2600. This compares with 12 families observed at site D900 and 22 observed at site D2600 during the pre-effluent study (Haack 1984).

The number of samples taken, as well as the habitat of the site, influenced the number of families observed. Although only a few samples were collected at sites U400 and U25, numbers of families found were not significantly different. These sites are shallow, with slowly moving water. However, greater differences were noted at sites U1200 and D2150, which are sites with deeper, more rapidly flowing water.

The names and numbers of macroinvertebrates found during the 1985-86 sampling period can be found in Appendix A. Each number represents the mean of two replicate samples. Mean values, both pre-effluent and post-effluent, of total invertebrates, taxonomic groups and feeding groups appear in Appendix B. These will be discussed in the following sections.

Total Macroinvertebrate Abundance

An exponential decline in mean total macroinvertebrates with distance from the wastewater outfall was observed (Figure 8). No significant difference between pre-effluent and post-effluent sampling periods was observed for control sites U1200, U400, and U25, taken as a group for total macroinvertebrates (Student's t-test, pre-effluent mean of 5.58, standard error = 0.35 and post-effluent mean of 5.84, standard error = 0.19). Total invertebrates at the control sites, taken as a group, were significantly different from the near-discharge experimental sites D25 (two-way ANOVA, $F_{1,54} = 32.82$, $P < .0001$), D200 (two-way ANOVA, $F_{1,54} = 19.59$, $P < .0001$), D400 (two-way ANOVA, $F_{1,54} = 10.28$, $P < .0028$), and downstream experimental site D2600 (two-way ANOVA, $F_{1,54} = .0473$, $P < .0473$) for the 1985-86 period. Although a similar pattern was observed for 1987, the scale of difference in upstream and downstream locations was not as dramatic. This shift during 1987 may be due, in part, to an increase in total number of predators. No significant difference between pre-effluent and post-effluent total macroinvertebrates was observed for site D900. As previously indicated in the discussion of water chemistry, this site may have been an area not receiving direct flow of wastewater.

In general, exponential increases in total macroinvertebrate abundance occurred at near discharge sites D25, D200, and D400, with a gradual leveling off at lower levels than the initial increase (Figure 8 and 9). Invertebrates may have prematurely increased in numbers at site D200 in August 1985 because of being physically moved by increased water flow due to the first major hurricane during the post-effluent sampling period. The November 1985 observations for site D400 could have been affected by the second hurricane, resulting in a premature, exponential increase. Alternatively, two peaks for all three

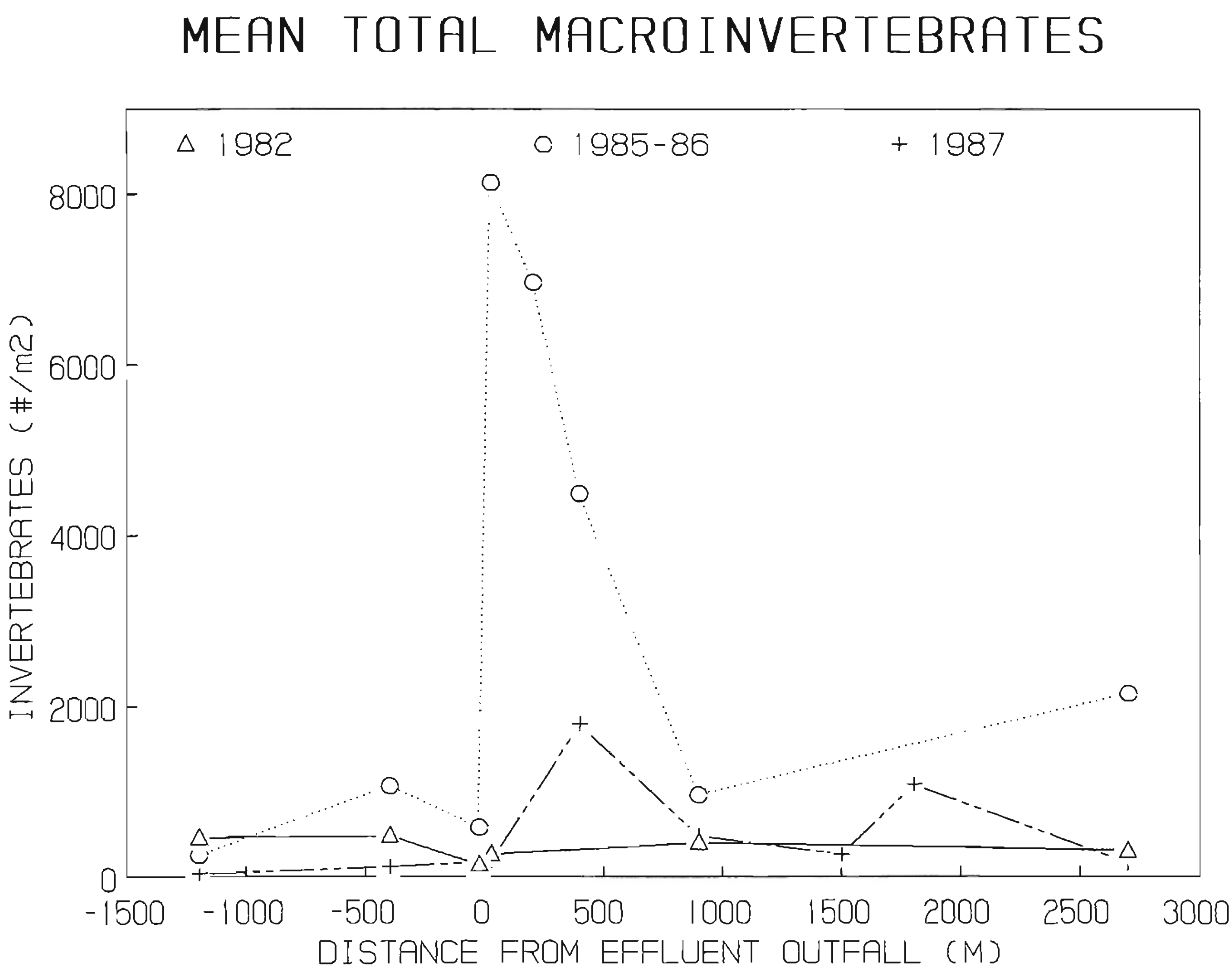


Figure 8. Mean total invertebrates at various distances upstream ("-" distances on figure) and downstream ("+" distances on figure) from point of wastewater discharge to titi-shrub swamp near Apalachicola, Florida. Data are means of rel=plicate monthly values for preapplication (1982), one-year (1985-86) and two-year (1987) postapolication.

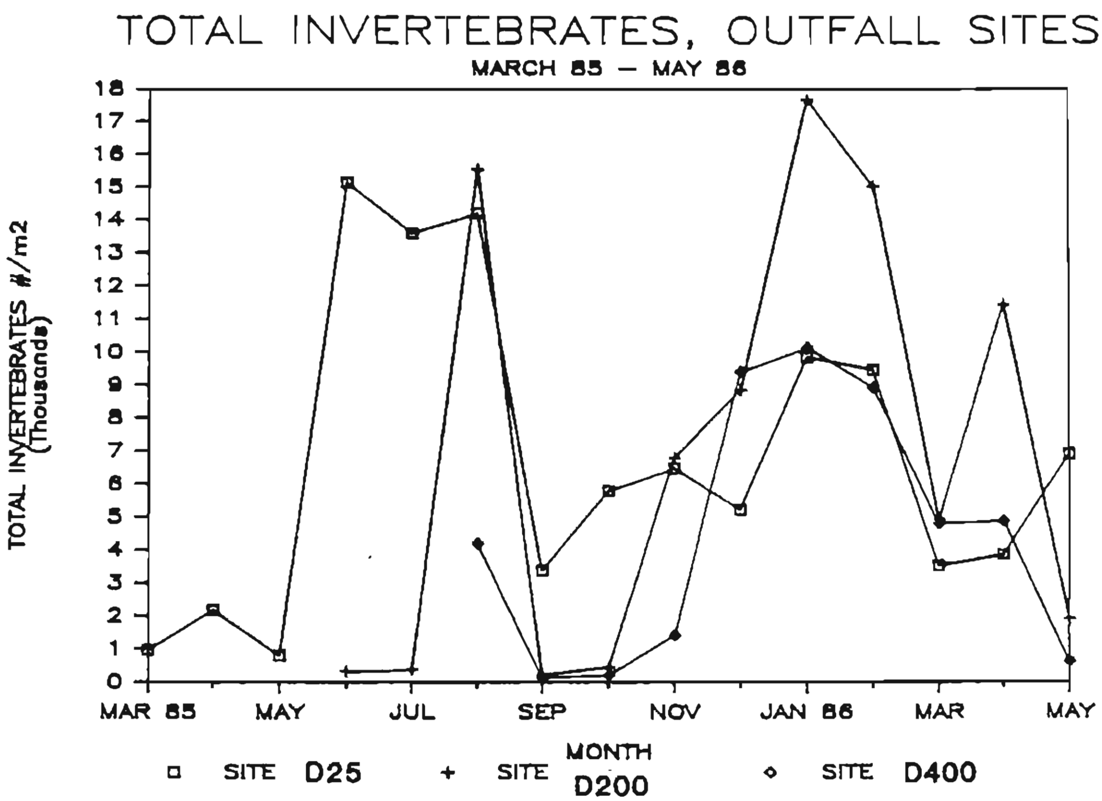
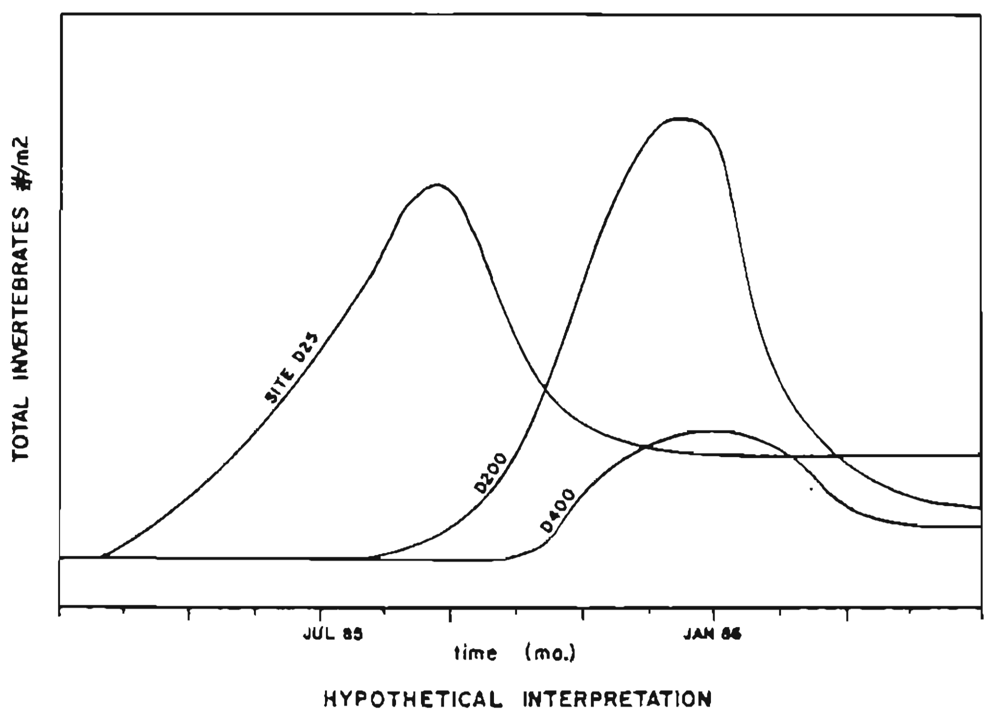
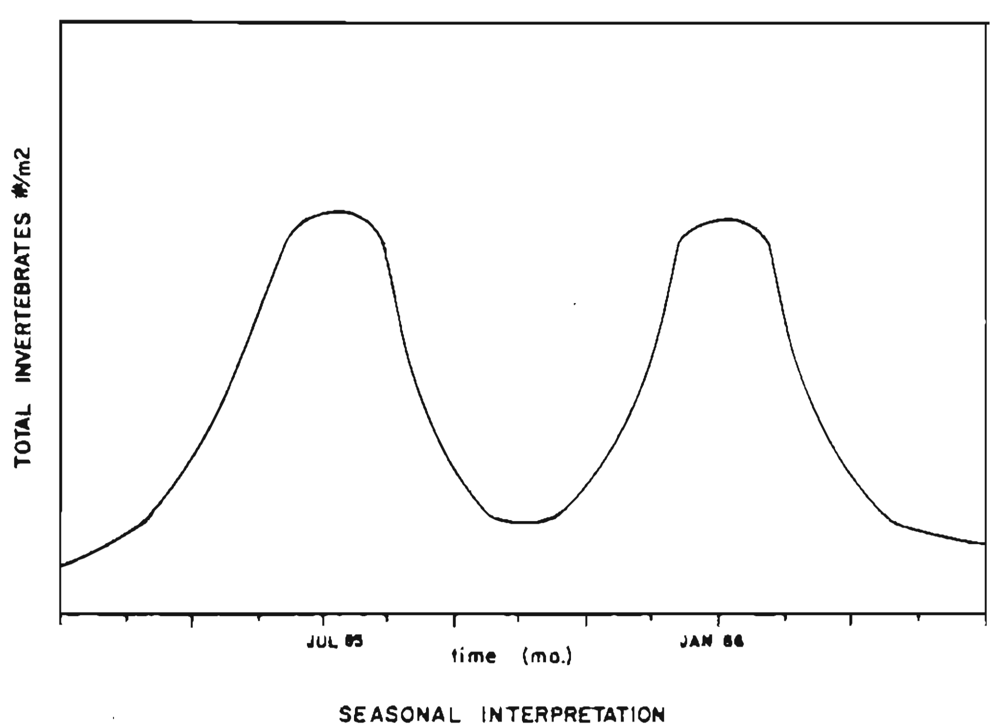


Figure 9. Monthly total invertebrates at outfall sites D25, D200, and D400 after addition of secondarily treated effluent to the wetland. a) Actual data; b) Hypothetical interpretation; c) seasonal interpretation.

(b)


TOTAL INVERTEBRATES #/m2
SITE D25
D200
D400
JUL 85
JAN 86
time (mo.)
HYPOTHETICAL INTERPRETATION

(c)


TOTAL INVERTEBRATES #/m2
JUL 85
JAN 86
time (mo.)
SEASONAL INTERPRETATION

sites, in August and January, could be attributed to seasonal events in macroinvertebrate abundance, such as emergence (Figure 9).

Taxonomic and Feeding Guild Composition

Haack (1984) found no distinct seasonal trends in macroinvertebrate composition at this study site. Therefore, mean taxonomic group abundance and feeding guild abundance were used in this study. The plot for Diptera (Figure 10) mimics the corresponding total invertebrate graph for the post-effluent sampling period (Figure 8). This is an indication of the dominance of Dipterans at the study sites.

The plot for Crustacea (Figure 11) shows a decrease in means for wastewater affected sites. Coleoptera and Oligochaeta do not show a significant increase or decrease (Figure 11).

The orders Ephemeroptera, Trichoptera, and Plecoptera are associated with clean water areas (Gaufin and Tarzwell 1955; Pennak 1978). "Clean water" organisms were grouped together in this study, and included Trichoptera, Plecoptera, Lepidoptera, Ephemeroptera, and Megaloptera. These occurred in low numbers in both pre-effluent and post-effluent sites (Figure 12).

The category "Other" includes Collembola, Odonata, Gastropoda, and Hemiptera. Organisms found in this group were either not likely to be affected by wastewater, or in the case of *Physa*, an indicator of polluted conditions (Pennak 1978). These groups increased at wastewater affected sites D25, D200, and D400 (Figure 13).

Pezeshki (1987) noted that during the first year after wastewater application, feeding guild abundances paralleled taxonomic group abundances for two groups, the Collectors, represented largely by the Dipterans (of which the Chironomidae were most abundant), and the Generalists (Figure 14), represented entirely by the Crustacea. A possible reason for the increase in Collectors is the increase in decaying matter, composed largely of dying *Sphagnum* sp. as well as other plant material, which could have represented an increased food supply for this group. Predators, represented by members of the order Odonata, Coleoptera, and Hemiptera also increased in effluent-affected areas (Figure 15). This was probably due to an increased food supply in the form of Chironomidae. Herbivores and filterers did not show a significant increase or decrease (Figure 14).

Three significant patterns emerged in the 1987 post-application period. First, almost no crustacea were found in 1987 either in upstream (non-wastewater–impacted) nor downstream portions of the wetlands. Second, there was a significant decline back to background levels in collectors from the 1985-86 sampling period to the 1987 sampling period (Figure 15). Third, there was a twofold to threefold increase in total number of predators (Figure 15) from the 1985-86 period to the 1987

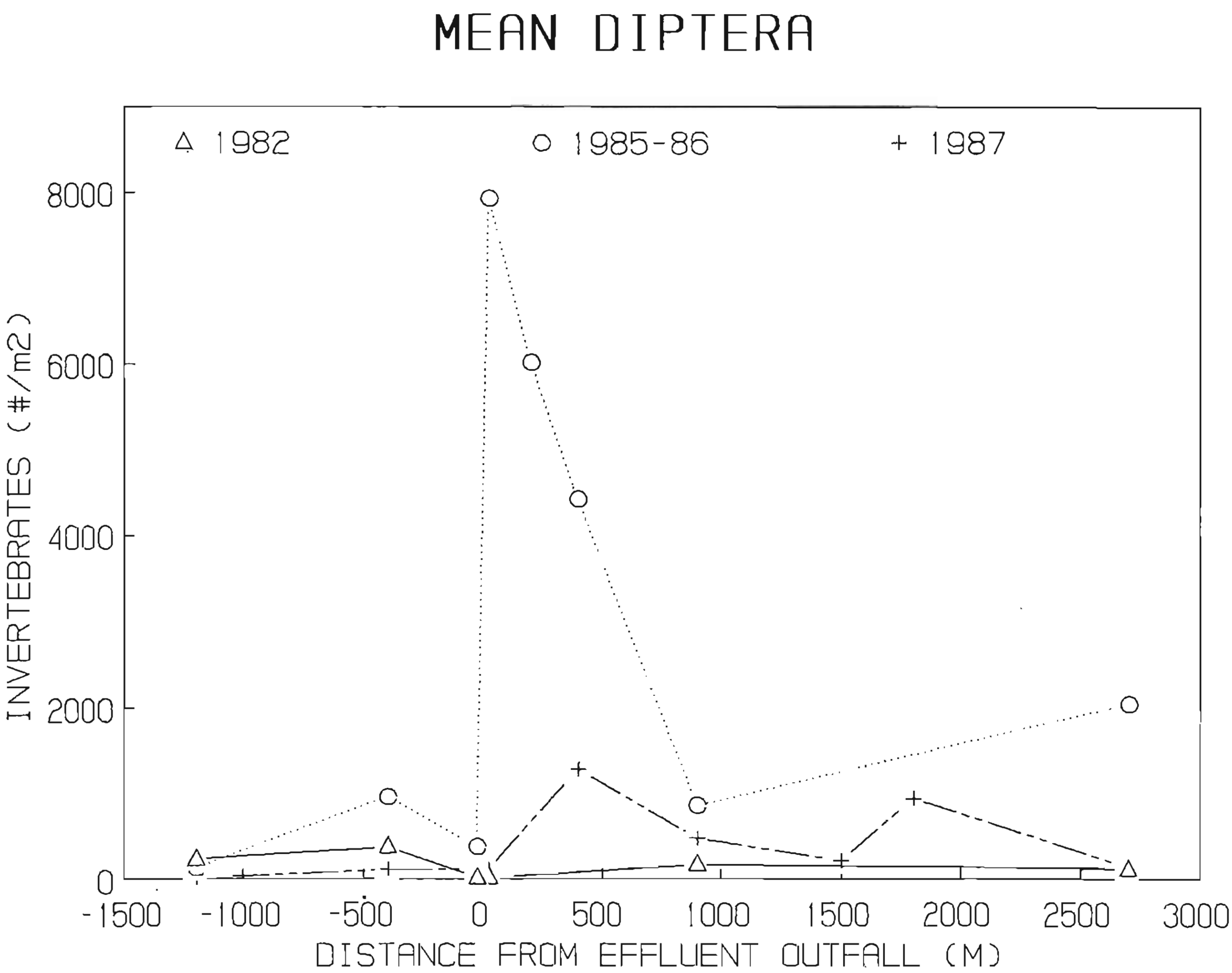


Figure 10. Mean Diptera in the Apalachicola wetland during pre-discharge period (1982; Jan-May 1985) and the post-discharge period (June 1985-May 1986 and 1987).

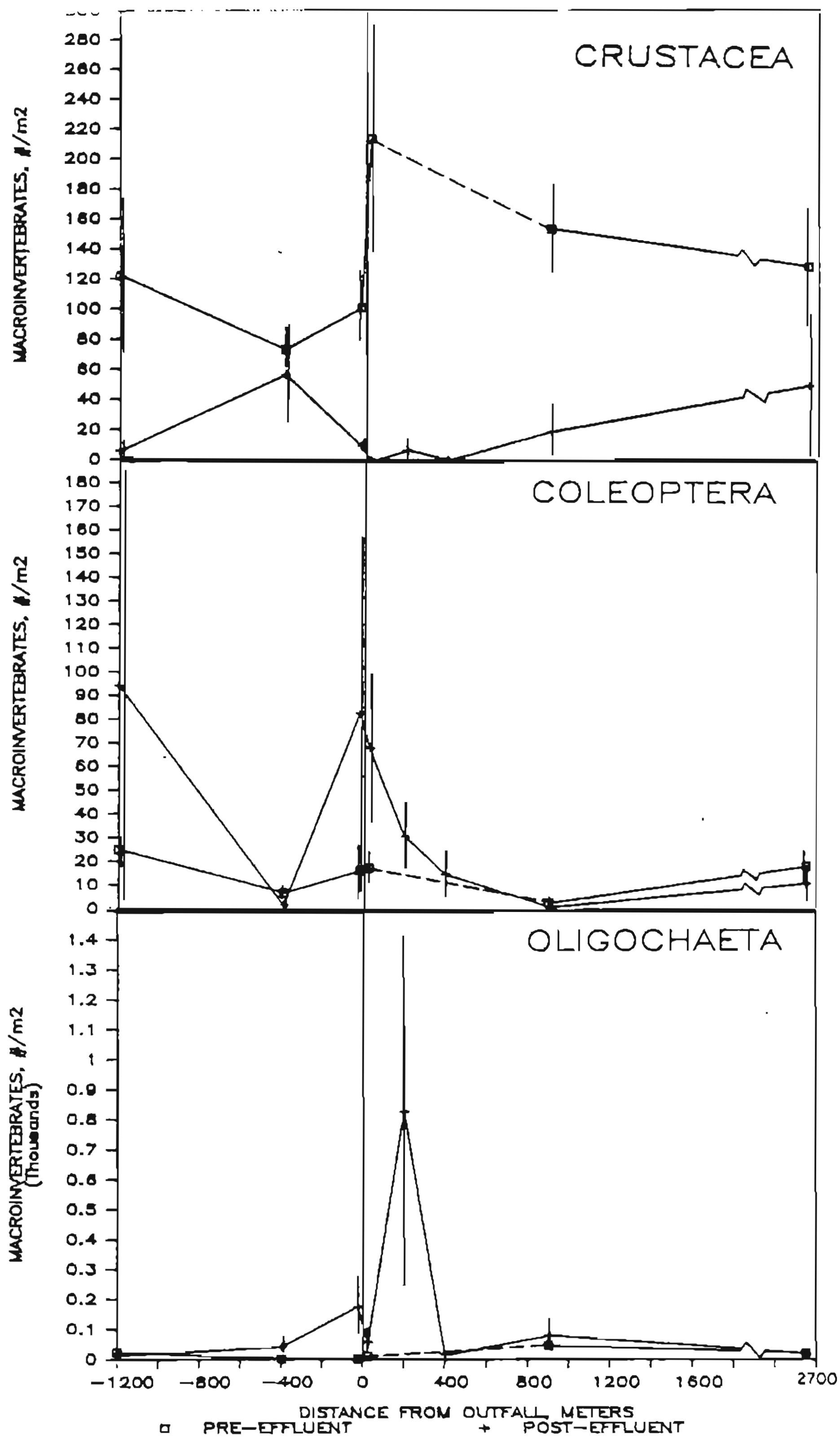


Figure 11. Mean Crustacea, Coleoptera and Oligochaeta in the Apalachicola wetland during pre-discharge period (1982; Jan-May 1985) and the first year post-discharge period (June 1985-May 1986).

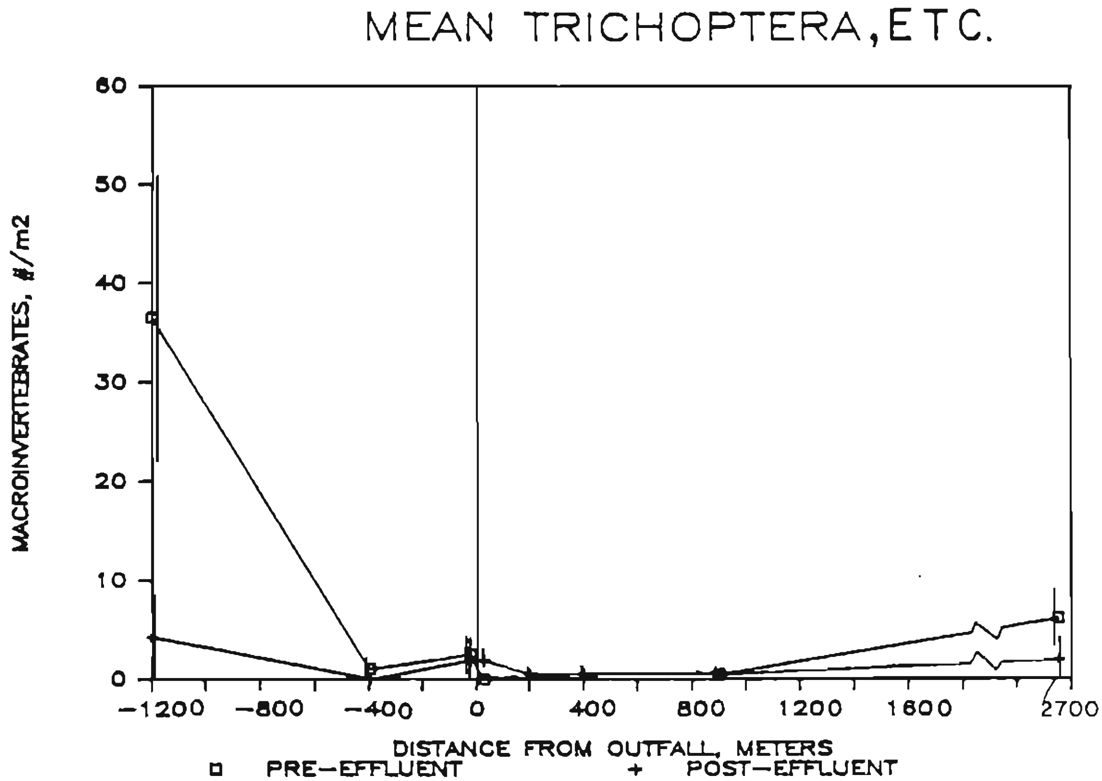


Figure 12. Mean Trichoptera, Ephemeroptera and Plecoptera in the Apalachicola wetland during pre-discharge period (1982; Jan-May 1985) and the first year post-discharge period (June 1985-May 1986).

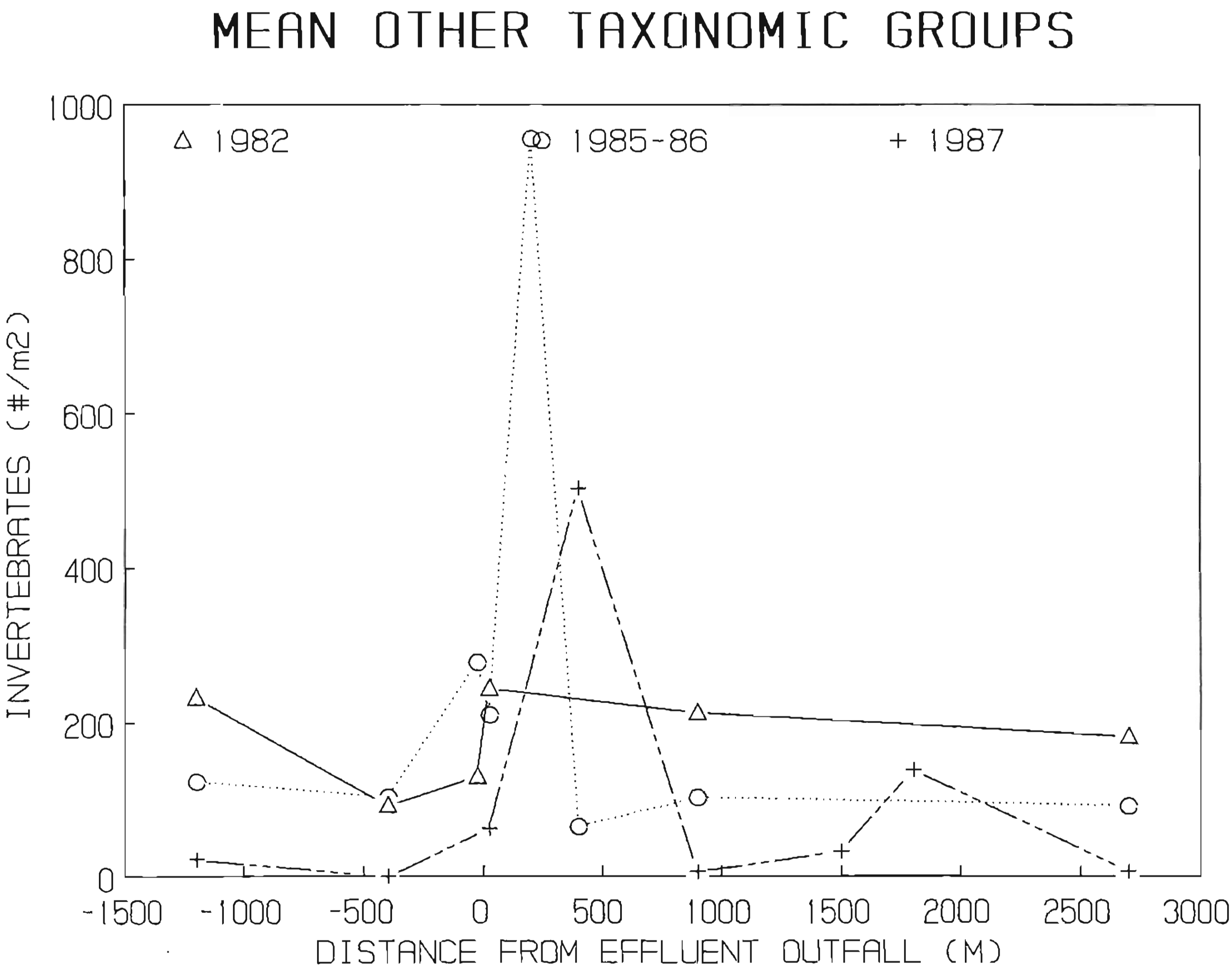
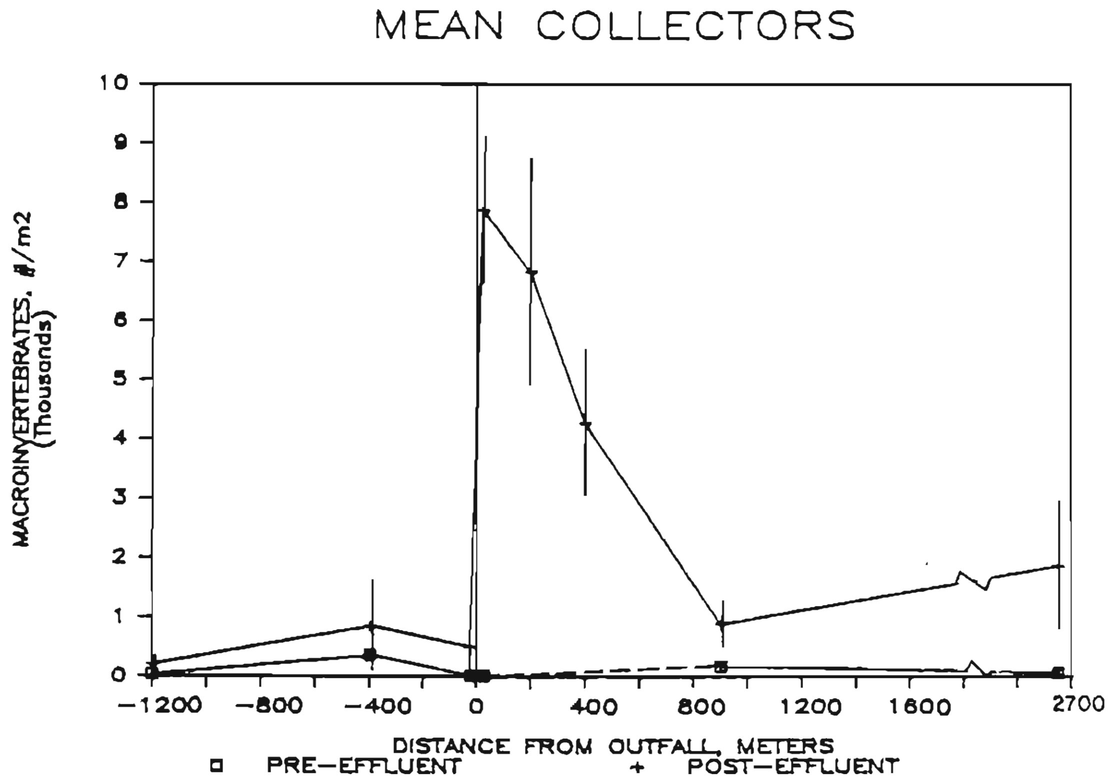


Figure 13. Mean "other" (includes Liepidoptera, Megaloptera, Collembola, Odonata, Gastropoda and Hemiptera) at various sampling sites in the titi shurb swamp near Apalachicola, Florida.



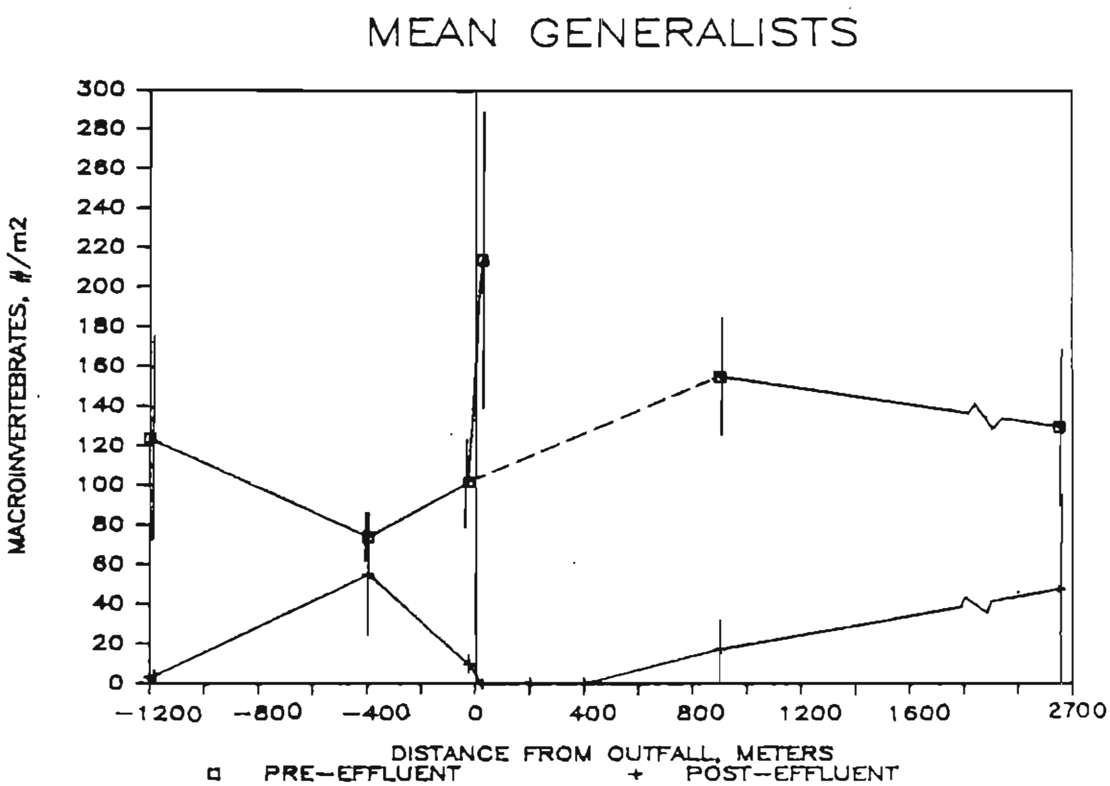
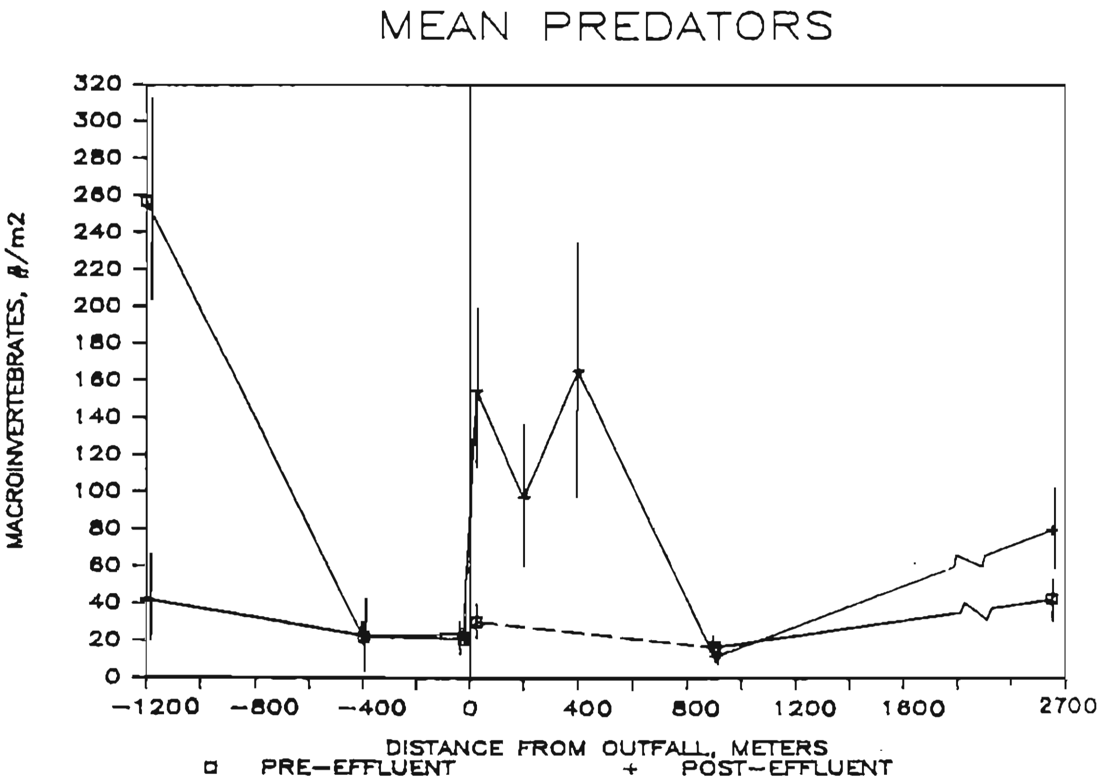


Figure 14a. Mean number of individuals in select feeding guilds in the Apalachicola wetland during pre-discharge period (1982; Jan-May 1985) and the first year post-discharge period (June 1985-May 1986).



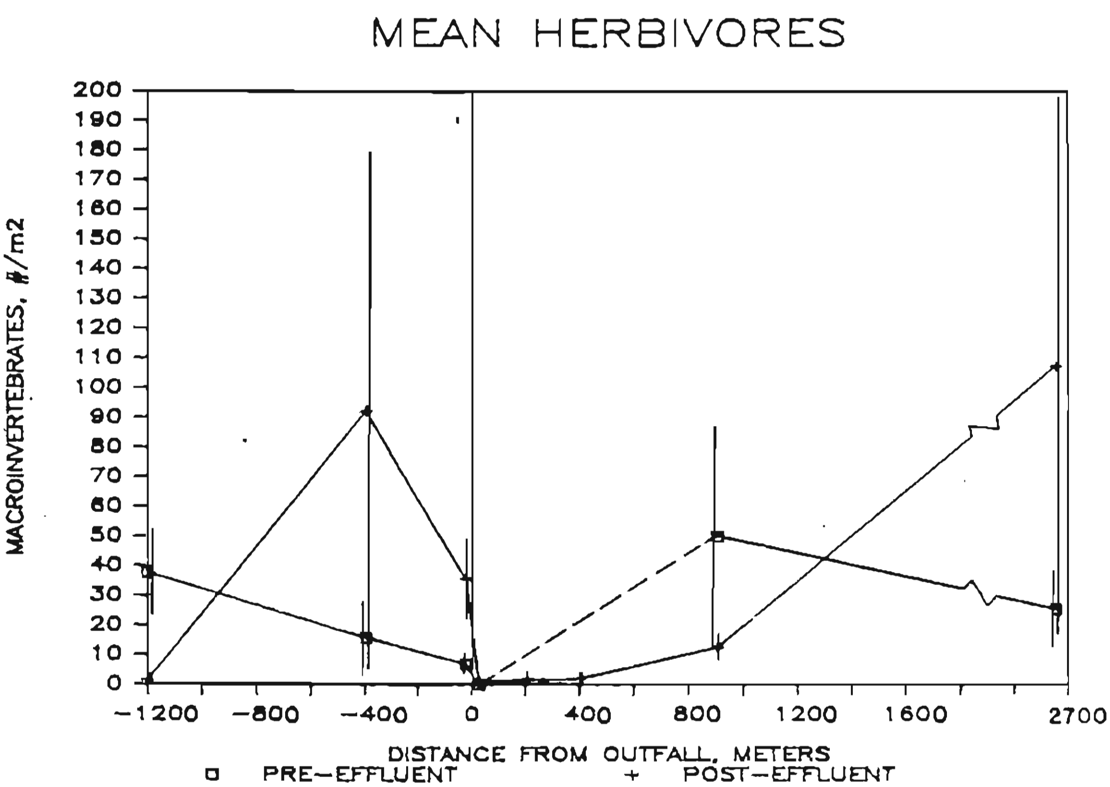


Figure 14b. Mean number of individuals in select feeding guilds in the Apalachicola wetland during pre-discharge period (1982; Jan-May 1985) and the first year post-discharge period (June 1985-May 1986).

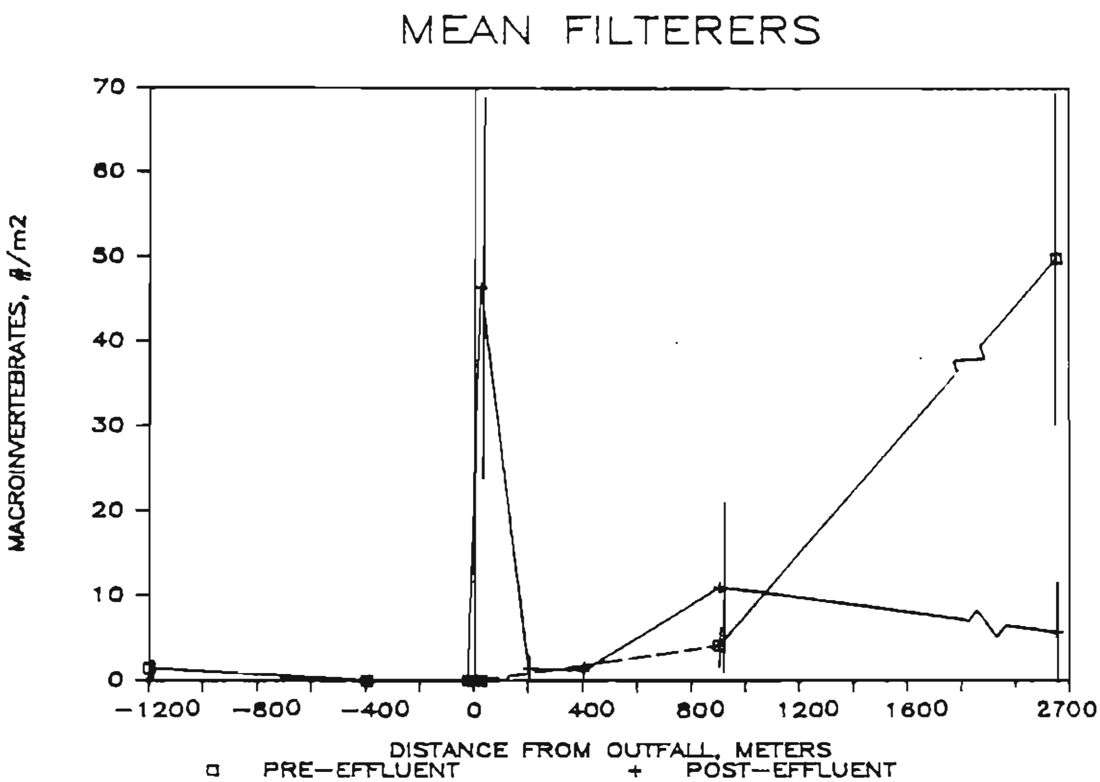


Figure 14c. Mean number of individuals in select feeding guilds in the Apalachicola wetland during pre-discharge period (1982; Jan-May 1985) and the first year post-discharge period (June 1985-May 1986).

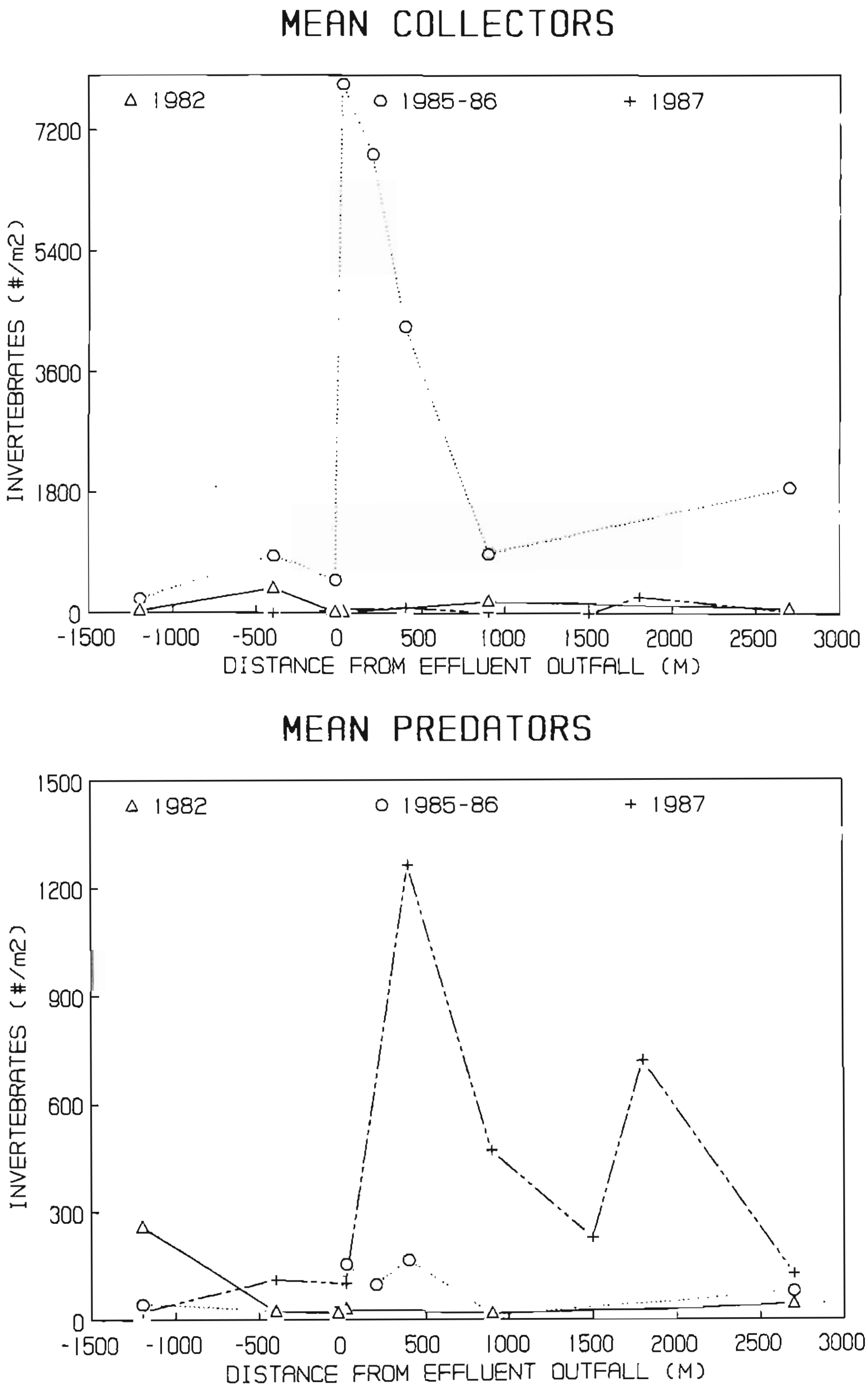


Figure 15. Mean number of collector and predator invertebrate feeding guilds at various sampling sites in titi shrub swamp near Apalachicola, Florida during pre- and post-application study period.

period. Increase in predators may be a function of not only increase in stability and number of prey populations, but also may reflect a stability in "constancy" of water in this wetland.

Diversity

Diversity indices were calculated for each sample and averaged (refer to *Methods* section for an explanation of equations). Then, the mean of these averages was plotted for the sampling periods for each site. A group was defined as the organisms constituting the lowest taxonomic level achieved in this study. An increase in number of groups occurred at sites U25, D25, and D400 during June 85 to May 86 (Figure 16). Mean number of groups at sites U1200, U400, and U25 are significantly different from 1 yr to the next, even though no wastewater effect occurred. For sites U1200 and U400, the probable cause is the infrequency of samples ($n = 4$) for these sites during post-effluent time as opposed to pre-effluent ($n = 12$). Comparisons of means of sites D25 and D900 are more accurate because approximately the same number of samples pre-effluent and post-effluent are being compared.

The mean Shannon-Weaver diversity index showed decreases at wastewater-affected sites D25, D200, and D400 (Figure 16). The decreases in Shannon-Weaver index at sites U1200, U400, and D2600 could be due to the decreased sample size during the post-effluent sampling period; however, site U25 does not show this decrease ($n = 3$). Mean dominance (Figure 17) increased at wastewater-affected sites D25, D200, and D400, Chironomidae being the group contributing most to this value. Correspondingly, mean evenness decreased at all sites during June 85 to May 86 (Figure 17). A significantly greater decrease was observed at site D25.

Plots of means of two values for species richness showed varying results. Species richness, as computed by the equation $D1 = S\text{-}1/\ln T$ (Odum 1971) showed decreased species at the control sites U1200 and U400, increased richness at site U25, and increased values at sites D25, D200, and D400 during June 85 to May 86 (Figure 18). Alternatively, using another method for calculating species richness, $D2 = S/\sqrt{T}$ (Odum 1971), decreased species richness was found at all sites except site U25 during the post-effluent sampling period (Figure 18). No significant change in species richness 2 was observed for site U25.

Deformities in Chironomidae

A number of investigations have shown that aquatic insects may exhibit structural deformities in response to pollution (Cushman 1984). Hamilton and Saether (1971), Hare and Carter (1976), Koehn and Frank (1980) and Warwick (1980) reported a variety of deformities in chironomid (Diptera: Chironomidae) larvae (the genera *Chironomus*, *Micropsectra*, *Procladius*, *Protanypus*, and *Stictochironomus*) living in contaminated habitats (Lake Erie; the Okanagan Lakes, B.C.; Georgian Bay, Ontario; Pasqua Lake, Saskatchewan; Taltowkanal, Germany) (Cushman 1984). These deformities

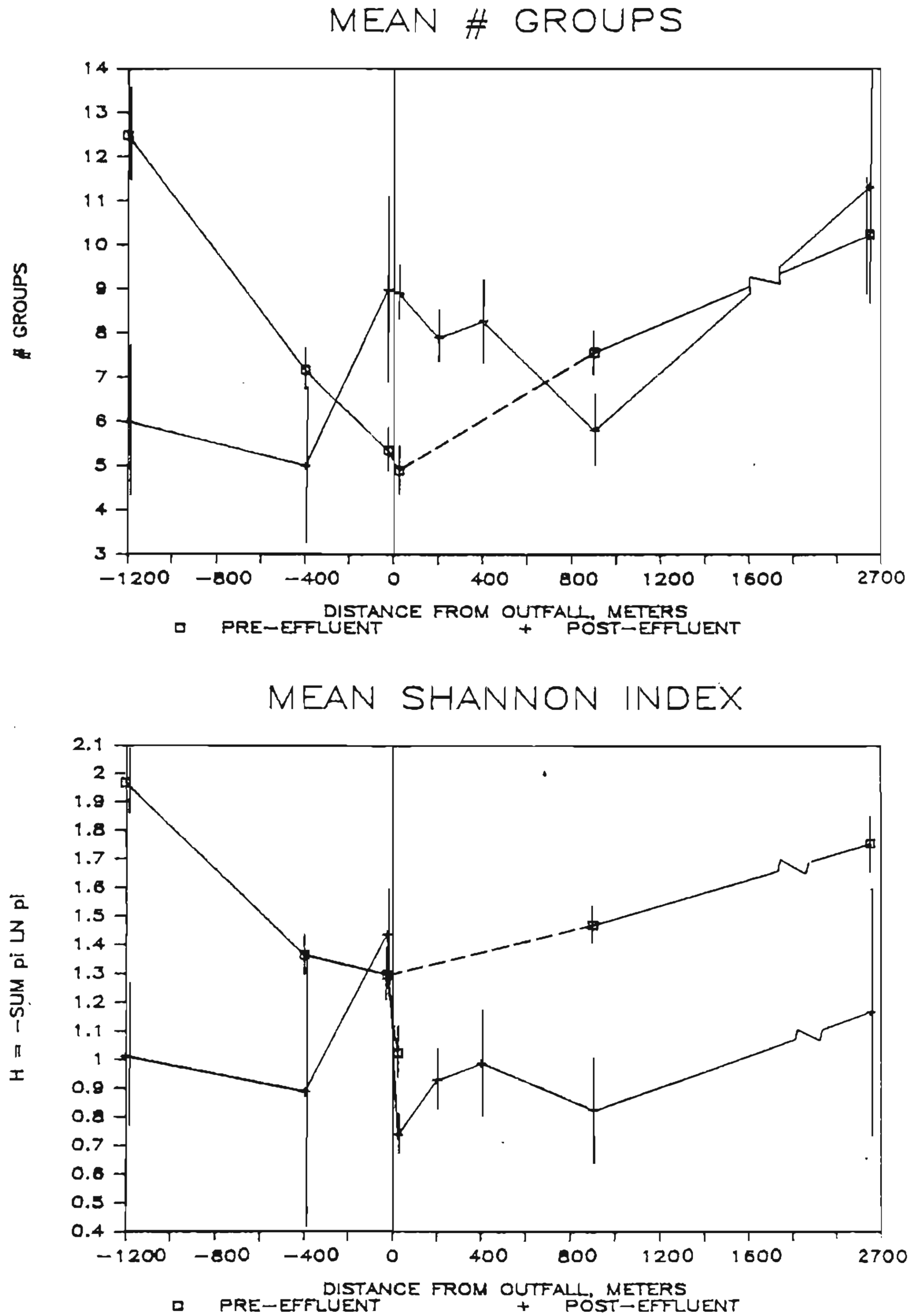


Figure 16. Mean number of toxonomic groups and mean Shannon diversity indices for macroinvertebrates in the Apalachicola wetland during pre-discharge period (1982; Jan-May 1985) and the first year post-discharge period (June 1985-May 1986). (See text for explanation of equations.)

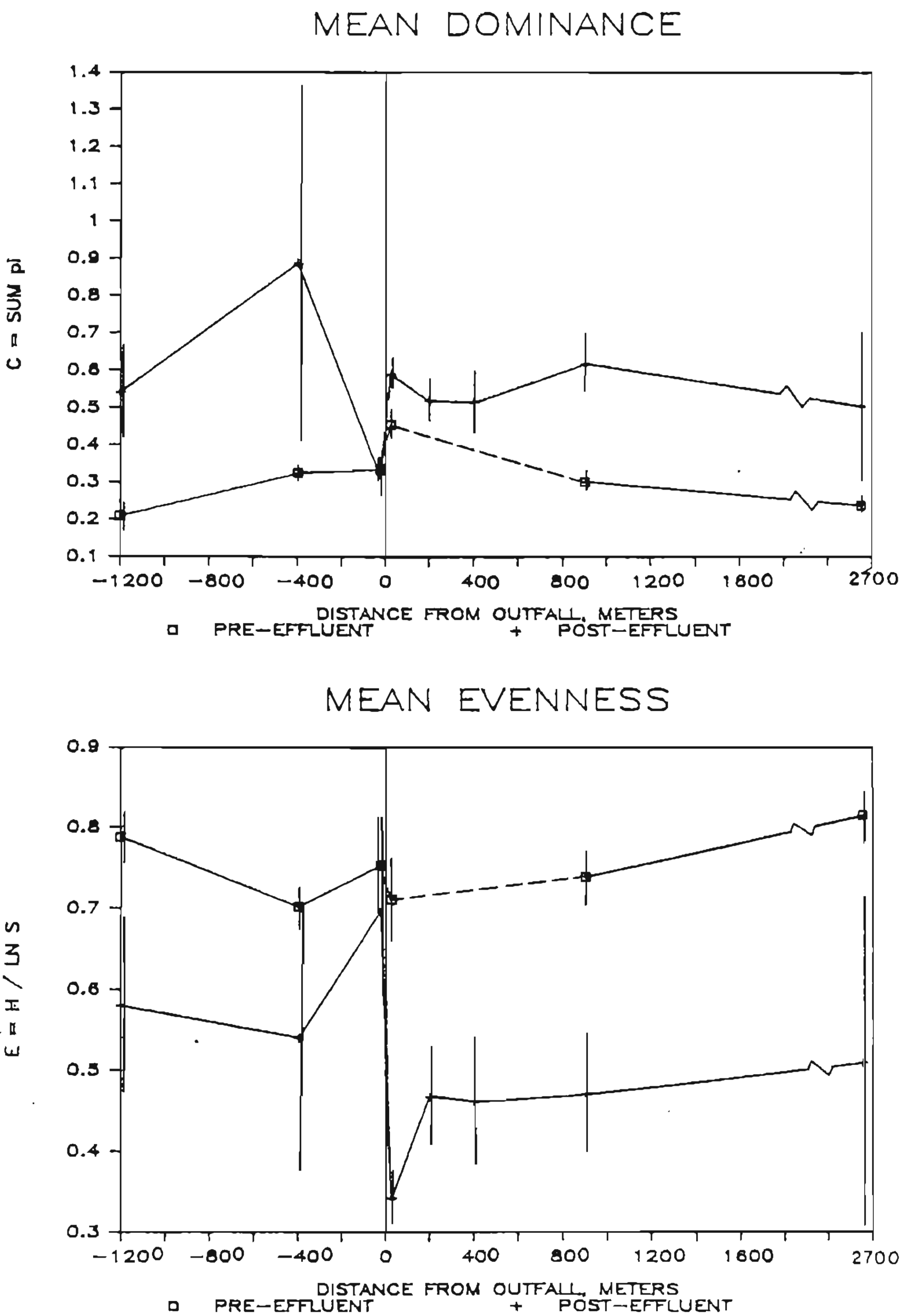


Figure 17. Mean dominance indicies and mean evenness indicies for macroinvertebrates see in the Apalachicola wetland during pre-discharge period (1982; Jan-May 1985) and the first year post-discharge period (June 1985-May 1986).. (See text for explanation of equations.)

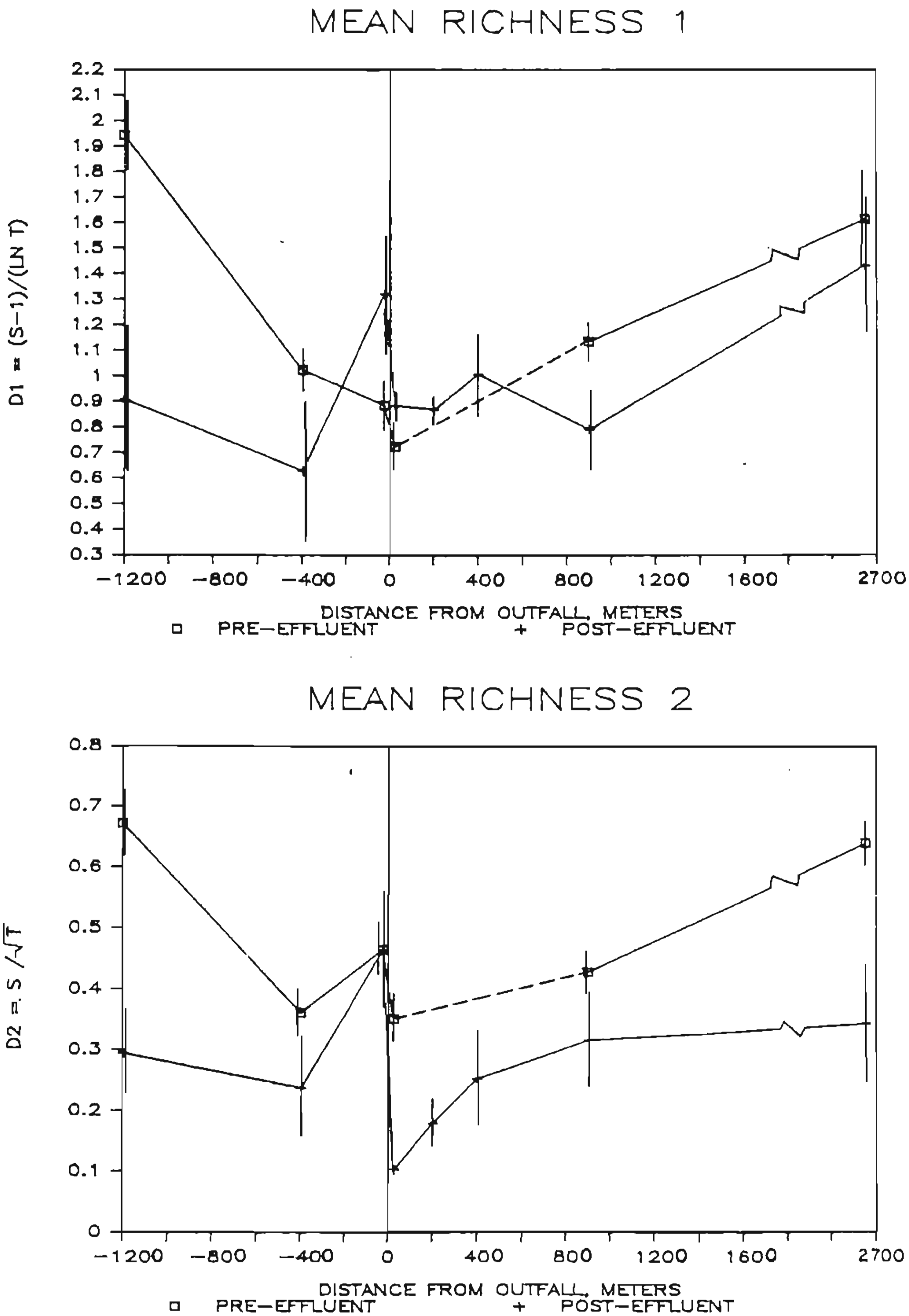


Figure 18. Mean indicies of macroinvertebrate taxonomic richness in the Apalachicola wetland during pre-discharge period (1982; Jan-May 1985) and the first year post-discharge period (June 1985-May 1986). (See text for explanation of equations.) Note: Dl = (S-1)/ln T; D2 = S/ T

included heavy pigmentation of the head capsule, thickening of the head capsule and body wall, and aberrations of the mouthparts (Cushman 1984).

In this study, only the mouth parts of Chironomidae were examined for deformities. The procedures of Cushman (1984) were followed in evaluating whether or not a deformity existed; unusual teeth that could reasonably be attributed to causes such as physical abrasion of teeth during feeding were not considered deformed. Only medial, or middle tooth, deformities were counted; lateral tooth deformities were not considered in this study. The types of deformities encountered were similar to those of Cushman (Figure 19).

The occurrences of deformities for sites D25, D200, D400, D900, and D2600 are presented in Table 9. No deformities were observed for the control sites U1200, U400, and U25. Fewer numbers of individuals were observed at these sites, however. Percent deformities for *Chironomus* sp. ranged from 0% to 11.65% with a mean of from 0.15% at site D400 to 2.57% at site D25, closest to the effluent outfall (Figure 20). Percent deformities for *Kiefferulus* sp. ranged from 0% to 5.88% with a mean of from 0% at sites D200, D900, and D2150 to 0.64% at site D25.

a)

b)

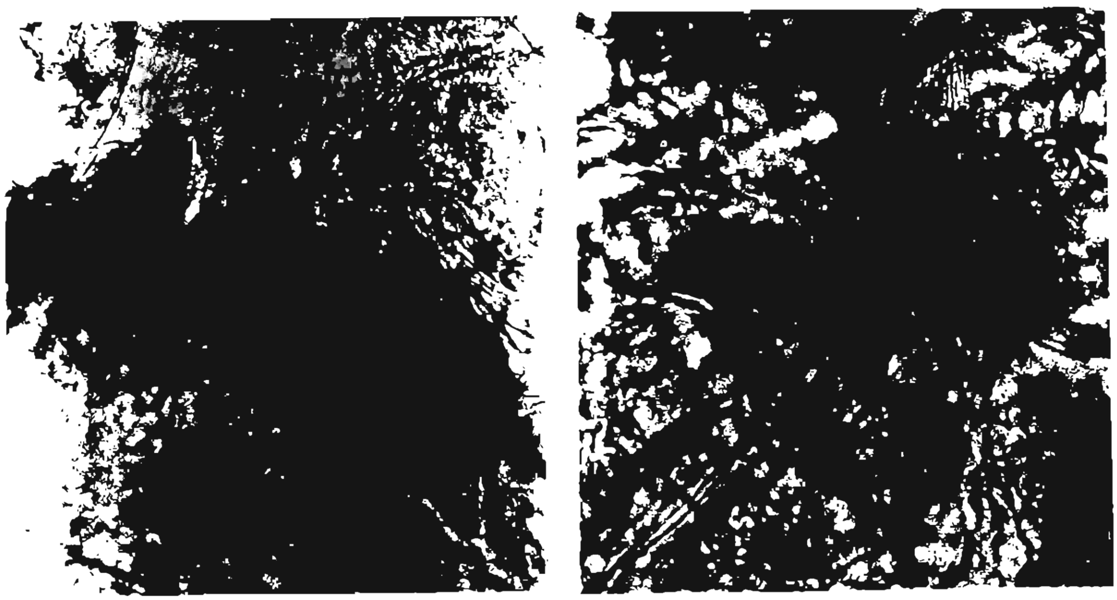


Figure 19. Photographs of mouthparts of *Chironomus* found in a forested wetland to which secondarily treated wastewater was applied. a) Normal *Chironomus* head capsule; b) Typical abnormal *Chironomus*

Table 9. Occurrences of Chironomidae defomity in the wetland near Apalachicola, Florida. C = *Chironomus*, K = *Kiefferulus*

SITE:		D25		D200		D200		D900		D2600	
		C	K	C	K	C	K	C	K	C	K
JUL 85	A:	0	0	0	0	0	0	0	0	0	0
	B:	0	0	0	0	0	0	0	0	0	0
AUG	A:	0	0	0	0	0	0	0	0	0	0
	B:	0	0	6.45	0	0	0	0	0	0	0
SEP	A:	0	0	0	0		0	0	0	0	0
	B:	0	0	0	0	0	0	0	0	0	0
OCT	A:	1.87	0	0	0	0	0	0	0	0	0
	B:	1.93	0	0	0	0	0	0	0	0	0
NOV	A:	0	4.60	0.94	0	0	0	1.39	0	0.60	0
	B:	0	0	0	0	0	0	0	0	0	0
DEC	A:	0	0	0.33	0	0.47	0	5.88	0	0	0
	B:	0	0	0	0	0	0	1.61	0	0	0
JAN 86	A:	0	0	0.83	0	1.44	0	0	0	0	0
	B:	0	0	1.04	0	0	0	0	0	0	0
FEB	A:	11.19	1.82	0.91	0	0.53	0.95	0	0	2.47	0
	B:	10.00	0	0	0	0	0	1.45	0	1.69	0
MAR	A:	5.85	0.47	0	0	0	0	11.43	0	0	0
	B:	4.58	0	0	0	0	0	2.70	0	0	0
APR	A:	4.24	5.88	0.46	0	0	0	0	0	0	0
	B:	11.65	0	0	0	0	0	0	0	0	0
MAY	A:	-	-	0	0	0.74	0	0	0	0	0
	B:	-	-	0	0	0	0	0	0	0	0
MEAN		2.57	0.64	0.50	0.00	0.15	0.04	1.11	0.00	0.22	0.00

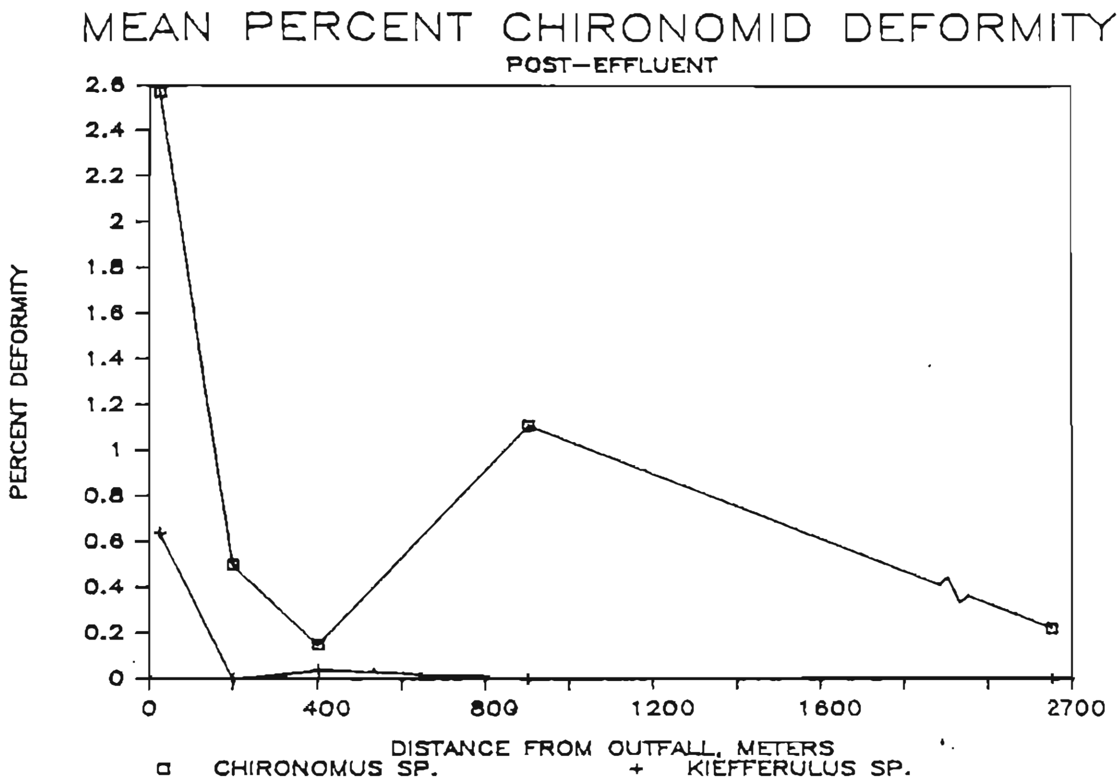


Figure 20. Mean percent Chironomidae deformity at the Apalachicola wetland.

VEGETATION

Purpose

The primary purpose of this portion of the research was to develop a description of the plant community setting of the Apalachicola shrub swamp, complete with vegetation map, prior to wastewater application. This baseline description could then be used to assess changes in plant community structure resulting from wastewater application. In addition, data were collected on baseline nutrient standing stock in wood plants prior to wastewater application.

Literature Review

Vegetation in Titi Shrub Swamps

The vegetation in titi swamps is often undifferentiated into strata (Clewell 1981). Broad-leaved evergreen or semi-deciduous shrubs and small trees are dominant, especially one of three species commonly called titi: *Cliftonia monophylla* (black titi or buckwheat tree), *Cyrilla racemiflora* (red titi or swamp titi) and *Cyrilla parviflora* (little-leaf titi). All three species occur in the same habitats, sometimes individually but often together. Black titi is usually more abundant than the two species of *Cyrilla* and tends to occupy slightly higher sites than red titi (Clewell 1981).

Titi swamps attract infrequent but destructive crown fires that serve in a homeostatic capacity, rejuvenating and perpetuating the community. The vegetation is rarely greater than 25 m in height and the taller the vegetation, the lower the frequency of fire, or at least the longer since the last destructive fire (Clewell 1981).

Titi swamps border pine flatwoods (which frequently burn) and only burn at their fringe serving as a buffer and preventing fires from reaching bay swamps (Clewell 1971). Groundwater seldom fluctuates too far below the surface in titi swamps (Wharton *et al.* 1977). Occasionally titi swamps border pond cypress or black gum swamps preventing these areas from fire as well (Clewell 1971). Irregular fires destroy the aerial portion of the vegetation in titi swamps, and coppicing after fire is very common, leading to multiple trunks (Clewell 1981). In discussing the distribution of the three species of titi Clewell (1981) stated that they do not segregate according to subtle gradients in the habitat. Their distribution appears random, as if once a titi plant, regardless of species, by chance becomes established at a given location, it persists indefinitely surviving fire by coppicing, regenerating the stand without intervening successional stages.

Titi swamps grade imperceptibly into bay swamps (Clewell 1981). Bay swamps occupy those portions of acid swamps that are wetter and less frequently burned than titi swamps (Clewell 1981). The dominants of titi swamps often make up the understory of bay swamps and when a fire does consume a bay swamp, the understory species such as black titi appear to grow faster than do the overstory species such as sweetbay (Clewell 1971). As a result, the site becomes essentially a titi swamp for perhaps 10 to 25 yr, until the overstory of sweetbay trees begin to form (Clewell 1971). He suggested that titi swamps, therefore, seem to be successional to bay swamps. Monk and Brown (1965) suggested that bay swamps are climax communities.

Pond cypress and black gum occupy the most deeply flooded habitats of any forested vegetation in the panhandle and few species are present in any given stand (Clewell 1971). The understory species of pond cypress/black gum swamps are usually the same as those that dominate other communities of acid swamp systems (Clewell 1981). Also, fire is rare. Draining of these swamps has been practiced widely, lowering the water table and allowing invasion of other acid swamp species. These swamps usually occupy peaty acid depressions in the deeper interior sites and bay swamps occupy the shallower exterior sites. Intergradations sometimes occur, particularly between pond cypress swamps and bay swamps. These swamps can also intergrade with bay swamps along the upper reaches of streams. Black gum swamps, which are also referred to as gum ponds, are usually bordered by pond cypress swamps which occupy slightly higher elevations.

Clewell (1971) raised the possibility that black gum is successional to pond cypress or vice versa. Black gum consistently occupies the lowest and wettest sites and these areas are bordered by pond cypress at slightly higher elevations. Monk and Brown (1965) also found that the importance of black gum increased sharply and that of pond cypress decreased sharply with decreasing depth of maximum flooding. In addition, with increasing levels of calcium, the importance of black gum increases sharply and that of pond cypress decreases sharply (Monk and Brown 1965). Initially pond cypress is favored in the lower sites and the subsequent enlargement of these depressions promotes the development of surrounding bay swamps, which when surrounded by titi swamps burn infrequently (Clewell 1971). Peat formation is relatively rapid and calcium released from peat decomposition promotes the establishment of black gum over pond cypress (Clewell 1971).

Much of the Florida panhandle consists of pinelands that are interrupted by swampy depressions. These swamps if deep enough typically contain four communities as zones along an elevational gradient

(Clewell 1981): a titi swamp, at the highest and driest elevations, followed by a bay swamp, a pond cypress swamp, and finally a black gum swamp. One or more of these communities may be missing if the steepness of the gradient compresses the vegetational zones (Clewell 1981). Swamps that contain a large proportion of black gum and particularly pond cypress may represent transitional phases between a pond cypress/black gum system and an acid swamp system or as suggested by Clewell (1981) may be included as a distinct and important part of the acid swamp system.

Biomass in Forested Wetlands

Forested wetlands may be grouped into three categories based on water movement and differences in nutrient inputs: still-water wetlands, slow-flowing water wetlands, and flowing water wetlands (Brown 1981). Still-water wetlands receive nutrients and water predominantly from precipitation. Slow-flowing wetlands receive water and nutrients from groundwater and surface water runoff. Flowing water wetlands receive water and nutrients from flooding streams.

The aboveground biomass of forested wetlands is variable, ranging from 3.6 kg/m^2 for a dwarf cypress forest to 45.2 kg/m^2 for a cypress tupelo alluvial river swamp (Brown 1981; Conner and Day 1982). No definite pattern has been established as high biomass exists in both still-water and flowing water wetlands. Low biomass for the dwarf cypress forest appears to be due to nutrient limitations or other stressors rather than the pattern of water delivery. However, the major source of water is precipitation and floodwaters tend to be stagnant and generally shallow (Brown and Lugo 1982). There is a relationship between productivity and hydrologic and nutrient sources. Biomass production and litterfall (one component of biomass production) are highest in flowing water wetlands, less in slow-flowing wetlands, and lowest in still-water wetlands (Brinson, Lugo and Brown 1981; Brown and Lugo 1982). Any model describing the relationship between structural characteristics of wetland ecosystems must consider the interaction of all major external inputs, including both nutrients and water (Brown 1981).

The leaf litterfall portion of total litterfall has only been reported for a few freshwater forested wetland sites. In the Dismal Swamp the average leaf litterfall for cypress and mixed hardwood species is 492 $g/m^2 \cdot yr$, with the peak in the Autumn (Day 1983). Total litterfall for 2 yr in Austin Cary cypress dome, along with the percent of the total litterfall that was wood were reported by Deghi *et al.* (1980). Therefore the annual average leaf litterfall for the 2 yr can be calculated as 420 $g/m^2 \cdot yr$. The peak litterfall period was in November and December. Leaf litterfall for the floodplain forest of the Apalachicola River was 464 $g/m^2 \cdot yr$ (Elder and Cairns 1982). Considerable seasonal variability in leaf litterfall was observed. A characteristic pattern was evident, with maximum leaf litterfall in November and other high values occurring in Autumn months. A maximum leaf litterfall may be shifted towards a spring peak in association with the development of new leaves (Bray and Gorham

1964). This bimodal seasonal cycle for leaf litterfall (autumn and spring peak) was also found by Post and de la Cruz (1977) in a Mississippi coastal stream.

Methods

Vegetation Analysis

A map of the vegetation of the titi shrub swamp study site was made using both high and low altitude aerial photographs. Quantitative information about the structure and composition of the vegetation at the titi shrub swamp study site was obtained with a variation of the quadrat sampling technique (Smith 1978). Belt transects were laid out in each of four wetland community types delineated on the map. A species area curve was used to determine the minimum number of multiple plots needed for a satisfactory sample (Smith 1978). The identification and diameter at breast height (dbh) of individuals in the tree size class (dbh greater than or equal to 10 cm) were recorded in each of four 10 m x 20 m quadrats within each transect. The identification and dbh of individuals in the shrub size class (dbh less than 10 cm and greater than or equal to 4 cm, and height greater than 1.3 m) were recorded in each of four 5 m x 10 m quadrats (one within each tree size class quadrat). The dbh of the individuals was then converted to basal area. The density, dominance, and frequency values were determined for each species as follows (Cox 1976):

density = number of individuals/area sampled,

dominance = total basal area/area sampled,

frequency = number of plots in which species occurs/total number of plots sampled.

These values were then converted to a hectare basis. For a particular species, these values were then expressed in a relative form, which shows the percentage of that species among all species (Cox 1976):

relative density = (density for a species/total density for all species) x 100,

relative dominance = (dominance for a species/total dominance for all species) x 100,

relative frequency = (frequency for a species/total of frequency values for all species) x 100.

Relative values for density, dominance, and frequency were added together to give a single importance value for each species, which reflects the relative importance of the species in the community. Each importance value was converted to a percentage basis and expressed for both the stratum (size class) and the community.

A line intercept method (Smith 1978) was used along the 80 m permanent transects in each of the four communities to determine the percent ground cover of the vegetation less than 1.3 m in height. This includes herbaceous and woody vegetation. The total linear distance covered by each species (or bare hummock) along the transect was recorded. The percent cover was calculated as the total intercept length of each species (or bare hummock), divided by the total transect length, multiplied by 100.

Biomass and Nutrient Standing Stock Estimates

Biomass of the titi shrub swamp at the study site was estimated using regression equations describing biomass as a function of selected physical dimensions. For three species harvested at the study site, a computer program CURFIT (Spain 1982) for fitting ten basic model equations to a set of x,y data was used to determine the appropriate regression equations. The CURFIT program converts non-linear equations to a linear form. Statistics on the best fitting model equation are provided. Regression equations developed by Brown (1978) were used for species not harvested at the study site.

Ten different sized individuals of three species were felled, black titi, red titi, and *Magnolia virginiana* (sweetbay). The dbh and the height of each individual were recorded. The location (height) and diameter of all primary branches of each individual were recorded. A primary branch was any branch extending from the bole with a diameter less than the bole. The bole ended where the differentiation between the bole and a primary branch could not be made. The diameter of the two primary branches at the end of the bole was also recorded.

The diameter at the base of each individual (BD) and at a location where butt swell no longer occurred (S1D) was recorded. The length of this first section (S1L), with butt swell, was recorded. Beginning at this point and moving towards the end of the bole, the individual was divided into additional sections. The section length (SL) was determined by selecting a section with approximately the same diameter at each end. The length of each section and the diameter of the individual at the top of each section (SD) were recorded. A disc at the top of each section was harvested and each disc length (DL) was recorded. The discs were dried to a constant mass in the laboratory and their dry weight (DW) was determined.

The following formula was used to estimate the dry weight (SW) of all but the first section.

$$SW = SL \times DW / DL.$$

The dry weight of the first section (S1W), with butt swell, was estimated with the following formula.

$$S1W = (S1L \times D1W / D1L) + \{[(S1L \times D1W / D1L) \times (BD - S1D)] / (S1D \times 2)\}.$$

The bole biomass was estimated by summing the estimated dry weights of all sections.

The primary branches of each individual were divided into three size classes (small, medium and large) based on diameter. One primary branch in each size class was randomly selected from each tree and harvested (i.e., 3 per tree). Each primary branch was separated into leaf and branch material. Two hundred leaves were subsampled from each primary branch that was harvested and their area was determined in the laboratory with a Hayashi Denko Company model AAM-5 leaf area meter. Leaf and branch material were dried to a constant mass in the laboratory and their dry weights were determined. The leaf area and the dry weight of the 200 subsampled leaves were used to calculate the leaf biomass to area ratio. A leaf biomass to area ratio was calculated for each species based on tree height for two vertical intervals (9 to 12 m and 3 to 9 m).

The dry weight of branch material and the primary branch diameter of the 30 primary branches sampled for each species were used to predict the dry weight of branch material using primary branch diameter as the independent variable. In the same manner, the dry weight of leaf material and the primary branch diameter of the 30 primary branches sampled for each species were used to predict the dry weight of leaf material using primary branch diameter as the independent variable.

The estimated bole, branch, and leaf biomass for each individual were summed to obtain the estimated aboveground biomass for each individual. The estimated aboveground biomass and the dbh of the ten individuals for each species were used to predict the aboveground biomass using dbh as the independent variable. The regression equation for each species was used for individuals of that species greater than 4 cm dbh sampled in vegetation analysis quadrats to estimate their aboveground biomass on an areal basis. The estimated leaf biomass and the dbh of the ten individuals for each species were used to predict the leaf biomass using dbh as the independent variable. The regression equation for each species was used for individuals of that species greater than 4 cm dbh sampled in vegetation analysis quadrats in the bay swamp community to obtain an estimate of their leaf biomass on an areal basis.

The estimated aboveground biomass and the dbh of the smallest individual for each of the three species were used to predict the aboveground biomass using dbh as the independent variable. The regression equation was fitted through the origin. This regression equation was used for all individuals less than 4 cm dbh sampled in vegetation analysis quadrats to obtain an estimate of their aboveground biomass on an areal basis. Regression equations developed by Brown (1978) were used for black gum, pond cypress and slash pine trees.

Herbaceous biomass and litter were estimated by collecting all the material in five 0.5 m^2 circular plots randomly sampled within each 200 m^2 vegetation analysis quadrat (20 per community type). The material was separated into live (herbaceous) and dead (litter) components. Leaf litterfall samples were collected monthly for 1 yr from three 0.1 m^2 baskets located at 10 m intervals in each community.

A subsample of each disc (bole), leaf and branch material of each harvested primary branch, each herbaceous plot, each litter plot, and triplicate subsamples of the yearly composite of leaf litterfall from each community were ground in a Wiley Mill. A 0.1-g sample of the ground material was digested with a mixture of K_2SO_4, $CuSO_4$ and selenium in a ratio of 100:10:1, and 2 ml of H_2SO_4 (Nelson and Sommers 1972). The samples were heated on a block digester, cooled, and diluted to 50 ml with deionized distilled water, and then analyzed by automated colorimetric analysis for ammonium nitrogen and total phosphorus (USEPA 1980).

An estimate of the bole, branch and leaf biomass was determined as the product of the average percent of the total biomass of these components for the three species intensively studied and the aboveground tree biomass for each community type. The total nitrogen and total phosphorus in the bole, branch and leaf material were determined as the product of the estimated bole, branch and leaf biomass and the average concentration for these components. The total nitrogen and total phosphorus in the herbaceous component for each community type was determined as the product of the herbaceous biomass and the average concentration of this component in each community type. The total nitrogen and total phosphorus of the bole, branch and leaf material, and the herbaceous component of each community type were summed to obtain the total nitrogen and total phosphorus in the aboveground biomass of each community type.

The total nitrogen and total phosphorus in litter for each community type was determined as the product of the dry weight of litter and the average concentration of total nitrogen and total phosphorus in litter in each community type. The total nitrogen and total phosphorus in leaf litterfall for each community type was determined as the product of the dry weight of leaf litterfall and the average concentration of total nitrogen and total phosphorus in leaf litterfall in each community type.

Herbaceous vegetation was measured before and after wastewater addition at four sites in the wetland, representing the four vegetation community types. Vegetation in four 80-m transects was identified and quantified using a line intercept method. The transects were located in a gum pond area (similar to sites U400 and D900), bay head (similar to sites U1200 and D2700), titi shrub swamp area, and mixed shrub swamp area (similar to sites U25, D25, D200, and D400). Details of the methods used can be found in Best *et al.* (1987).

Results and Discussion

Vegetation Analysis

The titi shrub swamp study site (Figure 21) is bordered by flatwoods which have been logged and then planted with slash pine for production. This silvicultural activity included an attempt to drain the wetland communities with a perimeter ditch. Four wetland community types occur at the study site: titi swamp-titi phase, titi swamp-holly phase, bay swamp-mixed swamp phase, and black gum swamp.

Five phases of titi swamps were described by Clewell (1971) in his classification of vegetation types in the Apalachicola National Forest. Two phases, titi and holly, occur at the study site. In the titi phase either red titi or black titi are dominant. Pines are usually absent and the overstory is absent (Clewell 1971). In the titi phase of the titi swamp at the study site, red titi is dominant making up 28% of the community and black titi makes up 16% of the community (Table 10). Together the titi species make up 45% of the community. Slash pine makes up less than 3% and shrub size class individuals make up 83% of the community.

In the holly phase *Ilex myrtifolia* (myrtle-leaf holly) replaces any one species of titi as dominant and an overstory is absent (Clewell 1971). Also, in small swamps at the head of minor drainages little-leaf cyrilla and myrtle-leaf holly tend to grow together (Clewell 1971). In the holly phase of the titi swamp at the study site titi species together make up 35% of the community but for a single species, myrtle-leaf holly is dominant making up 20% of the community (Table 11). Black titi makes up 18% of the community and little-leaf cyrilla makes up 13% of the community. There are three other species that make up at least 10% of the community. There are no tree size class individuals as shrub size class individuals make up 100% of the community.

In the mixed swamp phase of bay swamps dominance is shared between sweetbay and other species (Clewell 1971). The understory is usually undifferentiated from the overstory and is composed of woody species common in titi swamps (Clewell 1981). In the mixed swamp phase of the bay swamp at the study site sweetbay makes up 35% of the community and black titi makes up 58% of the community (Table 12). Shrub size class individuals make up 64% of the community of which 63% are titi species.

In black gum swamps black gum is dominant and pond cypress is usually present. The understory is absent or composed of saplings of overstory species (Clewell 1971). In the black gum swamp at the

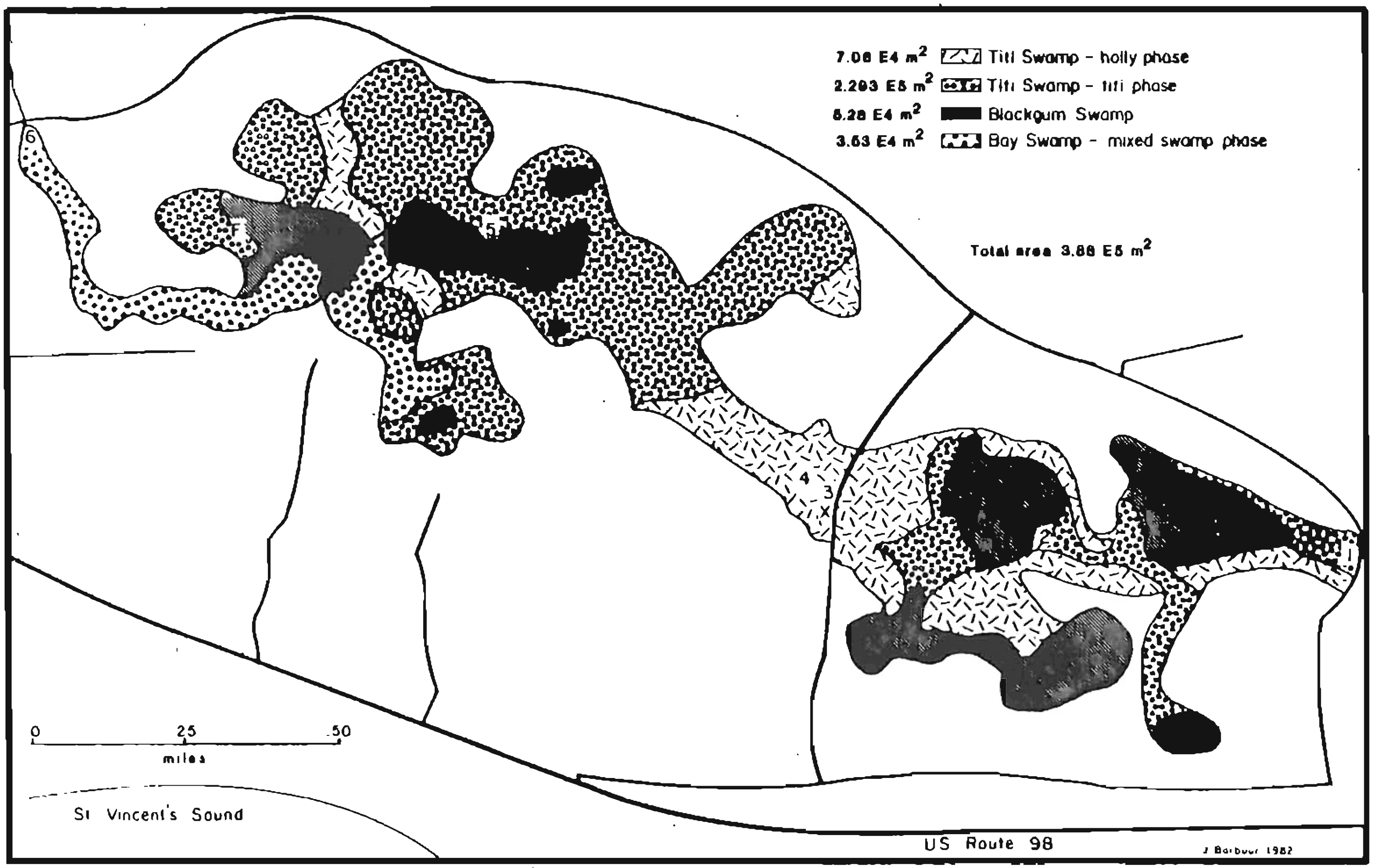


Figure 21. Map of the vegetation of the titi shrub swamp study site in Apalachicola, Florida including surface water sampling stations.

Table 10. Characteristics of woody vegetation (>1.3 m high) in a titi phase of the titi swamp in Apalachicola, Florida. Importance values were derived by summing relative frequency, relative density, and relative dominance and converting to a percentage basis.

Species	Relative Frequency % Stratum	Density: Actual Stems/ha	Density: Relative % Stratum	Dominance: Basal Area m^2/ha	Dominance: Relative % Stratum	Importance Value Basis of 100%: % Stratum	Importance Value Basis of 100%: % Community
TREE SIZE CLASS							
Taxodium ascendens	20.0	12.5	3.3	5.3	51.2	24.8	7.0
Cyrilla racemiflora	20.0	250.0	66.7	2.9	28.1	38.3	4.5
Pinus elliottii	20.0	12.5	3.3	0.9	9.1	10.8	2.2
Magnolia virginiana	20.0	50.0	13.3	0.7	6.5	13.3	1.9
Nyssa biflora	20.0	50.0	13.3	0.5	5.1	12.8	1.8
Tree Total	100.0	375.0	100.0	10.3	100.0	100.0	17.4
SHRUB SIZE CLASS							
Cyrilla racemiflora	16.0	8150.0	22.7	10.9	54.9	31.2	24.0
Lyonia lucida	16.0	9000.0	25.1	1.2	5.4	15.5	13.9
Cliftonia monophylla	12.0	8400.0	23.4	4.6	23.2	19.5	16.2
Clethra alnifolia	16.0	2150.0	6.0	0.2	0.9	7.6	6.6
Ilex coriacea	4.0	3100.0	8.6	0.3	1.4	7.4	6.5
Leucothoe axillaris	4.0	1800.0	5.0	0.5	2.6	3.9	3.3
Nyssa biflora	12.0	2400.0	6.7	1.8	8.8	9.2	7.5
Rhododendron sp.	8.0	700.0	2.0	0.1	0.4	3.5	3.0
Magnolia virginiana	4.0	150.0	0.4	0.5	2.4	2.2	1.8
Shrub Total	100.0	35850.0	100.0	20.1	100.0	100.0	82.8
GRAND TOTAL		36225.0		30.4			100.0

Table 11. Characteristics of woody vegetation (>1.3 m high) in a holly phase the titi swamp in Apalachicola, Florida. Importance values were derived by summing relative frequency, relative density, and relative dominance and converting to a percentage basis.

		Density		Dominance		Importance Value Basis of 100%	
Species	Relative Frequency % Stratum	Actual Stems/ha	Relative % Stratum	Basal Area m^2/ha	Relative % Stratum	% Stratum	% Community
SHRUB SIZE CLASS							
Ilex myrtifolia	7.7	4350.0	21.4	33.6	14.9	19.7	19.7
Cliftonia monophylla	15.4	3000.0	14.8	53.7	23.8	18.0	18.0
Cyrilla parviflora	7.7	2450.0	12.1	42.7	18.9	12.9	12.9
Magnolia virginiana	15.4	3650.0	18.0	10.7	4.8	12.7	12.7
Lyonia lucida	15.4	2350.0	11.6	9.9	4.4	10.4	10.4
Myrica cerifera	15.4	800.0	3.9	26.2	11.6	10.3	10.3
Hypericum reductum	3.9	2050.0	10.1	22.8	10.1	8.0	8.0
Nyssa biflora	7.7	850.0	4.2	6.8	3.0	5.0	5.0
Cyrilla racemiflora	7.7	450.0	2.2	5.3	2.4	4.1	4.1
Persea borbonia	3.8	350.0	1.7	13.9	6.1	3.9	3.9
Shrub Total	100.0	20300.0	100.0	225.5	100.0	100.0	100.0

Table 12. Characteristics of woody vegetation (>1.3 m high) in a mixed swamp phase of the bay swamp in Apalachicola, Florida. Importance values were derived by summing relative frequency, relative density, and relative dominance and converting to a percentage basis.

Species	Relative Frequency % Stratum	Density: Actual Stems/ha	Density: Relative % Stratum	Dominance: Basal Area m^2/ha	Dominance: Relative % Stratum	Importance Value Basis of 100%: % Stratum	Importance Value Basis of 100%: % Community
TREE SIZE CLASS							
Cliftonia monophylla	50.0	1025.0	71.3	16.5	67.1	62.8	21.5
Magnolia virginiana	37.5	400.0	27.8	7.3	29.7	31.7	11.6
Taxodium ascendens	12.5	12.5	0.9	0.8	3.2	5.5	2.4
Tree Total	100.0	1437.5	100.0	24.6	100.0	100.0	35.5
SHRUB SIZE CLASS							
Cliftonia monophylla	40.0	4900.0	63.6	18.4	64.1	55.9	36.8
Magnolia virginiana	40.0	2650.0	34.4	9.7	33.8	36.1	23.1
Cyrilla racemiflora	20.0	150.0	2.0	0.6	2.1	8.0	4.6
Shrub Total	100.0	7700.0	100.0	28.7	100.0	100.0	64.5
GRAND TOTAL		9137.5		53.3			100.0

study site black gum makes up 26% of the community and pond cypress makes up 20% of the community (Table 13). Shrub size class individuals make up 62% of the community.

Tree size class individuals make up 16% of the vegetation at the study site, but shrub size class individuals are the major component, making up 84% (Table 14). Black titi has the highest importance value of any species at the study site (21%) and the titi species together make up 39% of the vegetation at the study site (Table 15).

The ground cover in titi swamps is continuous with the understory and herbaceous species are absent except where these swamps border flatwoods. *Sphagnum* sp. (sphagnum or peat moss) may also be present (Clewell 1971). In the titi phase of the titi swamp at the study site no herbaceous species are present, sphagnum is abundant (63%) (Table 16), and shrub ground cover species are predominantly the same as those in the understory (Table 10). The holly phase of the titi swamp is bordered by flatwoods (Figure 21) which accounts for the presence of herbaceous species such as *Hypericum reductum*, *Xyris* sp. (yellow-eyed grass), and *Lachnanthes tinctoria*. Sphagnum is abundant (56%) and shrub ground cover species are predominantly the same as the vegetation greater than 1.3 m in height (Table 11).

The ground cover in bay swamps is also continuous with the understory or else sparse and patchy. Beds of peat moss are often conspicuous and sedges may be scattered (Clewell 1971). Sphagnum is abundant (46%) in the mixed swamp phase of the bay swamp, yellow-eyed grass, a sedge is present, and the shrub ground cover species are predominantly the same as those in the understory (Table 12). Ground cover in black gum swamps is absent (Clewell 1971). Sphagnum is very abundant in the black gum swamp (94%) and the shrub ground cover species are predominantly the same as those in the understory (Table 13).

Aquatic macrophytes including emergents, floating leaved plants and submergents are not a significant component in the titi shrub swamp. Shading and deep water prevent their growth but *Utricularia* sp. (bladderwort) does occur in the deep water areas. Although not a significant component, aquatic macrophytes do exist in open areas of flatwood depressions and along the margins of deeper swamps. Species found in these areas at the study site include *Hypericum reductum*, *Lachnanthes tinctoria*, yellow-eyed grass, *Rhynchospora* sp., *Scleria* sp., bladderwort, *Eriocaulon* sp., *Sarracenia* sp., and *Drosera* sp.

In general, the communities at this study site are those which occur in what Clewell (1981) described as acid swamp systems. This includes a holly phase of a titi swamp, a titi phase of a titi swamp, a mixed swamp phase of a bay swamp bordered by pond cypress and titi, and a black gum swamp bordered by pond cypress and titi.

Table 13. Characteristics of woody vegetation (>1.3 m high) in the black gum swamp in Apalachicola, Florida. Importance values were derived by summing relative frequency, relative density, and relative dominance and converting to a percentage basis.

		Density		Dominance		Importance Value Basis of 100%	
	Relative Frequency	Actual	Relative	Basal Area	Relative		
Species	% Stratum	Stems/ha	% Stratum	m^2/ha	% Stratum	% Stratum	% Community
TREE SIZE CLASS							
Taxodium ascendens	30.8	487.5	31.5	14.3	47.3	36.5	16.0
Nyssa biflora	30.8	725.0	46.7	10.2	33.9	37.2	13.0
Cyrilla racemiflora	30.8	325.0	21.0	5.0	16.4	22.7	8.0
Pinus elliottii	7.6	12.5	0.8	0.7	2.4	3.6	1.4
Tree Total	100.0	1550.0	100.0	30.2	100.0	100.0	38.4
SHRUB SIZE CLASS							
Nyssa biflora	14.8	3350.0	19.5	3.9	38.2	24.2	12.6
Lyonia lucida	14.8	4250.0	24.8	0.4	4.1	14.6	11.3
Cyrilla racemiflora	14.8	2200.0	12.8	3.0	28.8	18.8	9.7
Clethra alnifolia	14.8	1800.0	10.5	0.4	4.4	9.9	6.9
Ilex coriacea	7.4	2250.0	13.1	0.2	2.3	7.6	5.9
Leucothoe axillaris	11.2	1850.0	10.8	0.2	1.6	7.8	5.9
Taxodium ascendens	7.4	850.0	5.0	1.6	15.7	9.3	4.5
Ilex myrtifolia	7.4	350.0	2.0	0.2	1.8	3.7	2.4
Cliftonia monophylla	3.7	150.0	0.9	0.3	3.1	2.6	1.4
Magnolia virginiana	3.7	100.0	0.6	<0.1	<0.1	1.5	1.0
Shrub Total	100.0	17150.0	100.0	10.2	100.0	100.0	61.6
GRAND TOTAL		18700.0		40.5			100.0

Table 14. Characteristics of woody vegetation (>1.3 m high) in the titi shrub swamp in Apalachicola, Florida. Importance values were derived by summing relative frequency, relative density, and relative dominance and converting to a percentage basis.

Species	Relative Frequency % Stratum	Density: Actual Stems/ha	Density: Relative % Stratum	Dominance: Basal Area m^2/ha	Dominance: Relative % Stratum	Importance Values Basis of 100%: % Stratum	Importance Values Basis of 100%: % Community
TREE SIZE CLASS							
Taxodium ascendens	20.9	512.5	15.2	20.4	31.3	22.5	3.8
Cliftonia monophylla	18.6	1025.0	30.5	16.5	25.3	24.8	3.5
Nyssa biflora	17.5	775.0	23.0	10.8	16.5	19.0	2.8
Magnolia virginiana	18.7	450.0	13.4	7.8	12.3	14.8	2.5
Cyrilla racemiflora	17.5	575.0	17.1	7.8	12.1	15.6	2.4
Pinus elliottii	6.8	25.0	0.8	1.6	2.5	3.3	0.7
Tree Total	100.0	3362.5	100.0	65.3	100.0	100.0	15.7
SHRUB SIZE CLASS							
Cliftonia monophylla	13.7	16450.0	20.4	77.0	27.1	20.3	17.4
Lyonia lucida	13.7	15600.0	19.4	11.4	4.0	12.3	10.7
Cyrilla racemiflora	13.7	10950.0	13.5	19.8	7.0	11.4	9.7
Magnolia virginiana	11.3	6400.0	7.9	20.4	7.2	8.8	7.3
Nyssa sylvatica var. *biflora*	10.2	6600.0	8.2	12.5	4.4	7.6	6.4
Ilex myrtifolia	4.6	4700.0	5.8	33.8	11.9	7.4	6.2
Cyrilla parviflora	2.2	2450.0	3.0	42.7	15.0	6.8	5.6
Myrica cerifera	4.6	800.0	1.0	26.2	9.2	4.9	4.0
Clethra alnifolia	9.1	3950.0	4.9	0.6	0.2	4.8	3.9
Ilex coriacea	5.7	5350.0	6.6	0.5	0.2	4.2	3.6
Hypericum reductum	1.1	2050.0	2.5	22.8	8.0	4.0	3.3
Leucothoe axillaris	4.6	3650.0	4.5	0.7	0.2	3.1	2.7
Persea borbonia	1.1	350.0	0.4	13.9	4.9	2.1	1.7
Taxodium ascendens	2.2	850.0	1.0	1.6	0.6	1.3	1.0
Rhododendron canascens	2.2	700.0	0.9	0.1	0.1	1.0	0.8
Shrub Total	100.0	80850.0	100.0	284.0	100.0	100.0	84.3
GRAND TOTAL		84212.5		349.3			100.0

Table 15. Importance values for woody vegetation species (>1.3 m high) in the titi shrub swamp in Apalachicola, Florida. All species combined regardless of size class.

Species	Importance Value
Cliftonia monophylla	20.9
Cyrilla racemiflora	12.1
Lyonia lucida	10.7
Magnolia virginiana	9.8
Nyssa biflora	9.1
Ilex myrtifolia	6.2
Cyrilla parvifolia	5.6
Taxodium ascendens	4.9
Myrica cerifera	4.0
Clethra alnifolia	3.9
Ilex coriacea	3.6
Hypericum reductum	3.3
Leucothoe axillaris	2.7
Persea borbonia	1.7
Rhododendron sp.	0.8
Pinus elliottii	0.7
TOTAL	100.0

Table 16. Percent ground cover in the four community types in the titi shrub swamp in Apalachicola, Florida.

Titi swamp - titi phase		Titi swamp - holly phase	
Sphagnum sp.	63	*Sphagnum* sp.	56
Lyonia lucida	22	*Lyonia lucida*	20
Bare hummock	13	*Hypericum reductum*	15
Clethra alnifolia	2	*Xyris* sp.	12
Ilex corriacea	2	*Cliftonia monophylla*	8
Rhododendron sp.	2	*Cyrilla parvifolia*	6
Cyrilla racemiflora	1	*Myrica cerifera*	6
Pierus pillyreifolia	1	*Lachnanthes tinctoria*	3
Leucothoe axillaris	<1	*Ilex myrtifolia*	3
Cliftonia monophylla	<1	*Sabal palmetto*	3
Bay swamp - mixed swamp phase		**Black gum swamp**	
Bare hummock	46	*Sphagnum* sp.	94
Lyonia lucida	37	*Bare hummock*	7
Xyris sp.	11	*Ilex corriacea*	7
Sphagnum sp.	4	*Cyrilla racemiflora*	6
Cyrilla racemiflora	1	*Lyonia lucida*	3
Clethra alnifolia	1	*Rhododendron* sp.	<1
Cliftonia monophylla	<1		

Field observations support the hypothesis that black gum is successional to pond cypress in these systems. The greatest density of black gum individuals are located in the deepest and most central portions of the study site and sphagnum is abundant (94%, Table 16). These areas are surrounded by pond cypress and then by either a titi swamp, or a bay swamp at the upper reaches of a stream. The perimeter drainage ditch associated with silvicultural activity may have acted as a fire break and may have contributed to the success of black gum at the study site. Drainage may also have favored the invasion of other acid swamp species. Although black gum is dominant in the black gum swamp at the study site (26%), pond cypress makes up a significant portion of the community (20%). Therefore, the black gum swamp may be transitional in a scenario in which there is initial dominance of pond cypress in the lowest sites followed by the establishment and eventual dominance of black gum through time. This may also explain the presence of understory vegetation composed of other acid swamp species and the high percent cover of sphagnum (94%) found at this site. In addition, the black gum swamp is not a transitional phase to the titi shrub swamp, rather it is a distinct and important component of the system.

Shrub class individuals make up 84% of the vegetation at the study site and titi species make up 38%. Thus, this site can be described as a titi shrub swamp even though the central and deepest portion of the site are dominated by tree size class individuals that are not titi species.

Biomass and Nutrient Standing Stock Estimates

For the three species measured, the bole, branch and leaf biomass make up 69%, 28% and 3% of the total aboveground biomass, respectively (Table 17). The regression equations used to estimate branch and leaf biomass based on primary branch diameter, and aboveground biomass and leaf biomass based on the dbh of individuals in vegetation analysis quadrats are presented in Appendix B.

Aboveground estimates of the woody vegetation (greater than 1.3 m high) in the four community types are presented in Tables 18, 19, 20 and 21. Herbaceous biomass and litter estimates in the four community types are presented in Table 22. A summary of the aboveground biomass estimate of the four community types is presented in Table 23.

The aboveground biomass of the holly phase of the titi swamp is 1.3 kg/m^2 (Table 23). This is the smallest aboveground biomass of the four community types. This is not surprising as no tree size class individuals are present. The herbaceous component makes up 24% of the biomass of this community a large portion of which is sphagnum (Table 16). The aboveground biomass of the titi phase of the titi swamp is 8.4 kg/m^2 (Table 23). Tree size class individuals make up 58% and shrub size class individuals make up 41% of the biomass of this community. The herbaceous component makes up less than 1.5% of the biomass of this community. The aboveground biomass of the black gum swamp is 13.9 kg/m^2 (Table 23). Tree size class individuals make up 76% and shrub size class individuals make

Table 17. The dbh, estimated bole, branch, leaf and above ground biomass for ten individuals each of black titi, red titi, and sweetbay sampled at the study site.

Species	dbh (cm)	height (m)	bole (g)	branch (g)	leaf (g)	above ground biomass (g)
Black titi	18.5	11.4	90522	35448	6492	132462
	16.8	15.0	74348	18821	2909	96078
	15.0	13.1	57524	25388	3733	86645
	12.5	11.4	31940	15643	2125	49708
	8.6	10.2	17284	5403	809	23496
	6.9	8.6	10797	7656	979	19432
	6.5	7.4	6981	3661	445	11087
	4.5	5.6	3011	635	126	3772
	3.9	5.5	1903	554	109	2566
	3.6	4.8	1837	480	93	2410
Total			296147	113689	17820	427656
%			69.2	26.6	4.2	100.0
Red titi	22.0	8.6	40197	52892	3431	96520
	18.2	8.3	24316	17141	733	42190
	14.9	8.7	31508	17921	996	50425
	12.8	7.7	20883	15352	792	37027
	10.4	7.9	16095	12137	422	28654
	7.6	6.9	4716	1342	174	6232
	5.2	6.1	3072	908	94	4074
	5.0	6.8	3071	939	183	4193
	4.0	5.5	1934	254	75	2263
	3.4	5.0	2008	311	74	2393
Total			147800	119197	6974	273971
%			54.0	44.0	2.5	100.0

Table 17. Continued.

Species	dbh (cm)	height	bole (g)	branch (g)	leaf (g)	above ground biomass (g)
Sweetbay	17.6	11.7	61497	16589	2764	80850
	13.9	12.7	49029	3802	1130	53961
	13.5	12.0	44572	5297	1200	51071
	10.5	10.8	23154	3508	710	27372
	9.1	12.0	17357	561	219	18137
	7.4	8.3	9147	679	261	10087
	6.8	9.6	8914	1107	357	10378
	6.2	9.4	7172	746	277	8195
	4.4	7.4	3363	205	78	3646
	4.1	6.2	2867	401	145	3413
Total			227072	32897	7141	267110
%			85.0	12.3	2.7	100.0

Average % for three species	69.4	27.5	3.1	

Table 18. Aboveground biomass estimate of woody vegetation (>1.3m high) in a titi phase of the titi swamp in Apalachicola, Florida.

Species	Size Class (cm dbh)	Regression # in Appendix B	Biomass (g/m^2)
Cyrilla racemiflora	≥10	8	669.2
Cyrilla racemiflora	10< ≥4	8	470.2
Cyrilla racemiflora	<4	10	473.0
			1612.4
Cliftonia monophylla	10< ≥4	7	661.7
Cliftonia monophylla	<4	10	645.1
			1306.8
Magnolia virginiana	≥10	9	209.4
Magnolia virginiana	10< ≥4	9	103.2
			312.6
Nyssa biflora	≥10	11	191.4
Nyssa biflora	10< ≥4	11	145.3
Nyssa biflora	<4	10	303.7
			640.4
Taxodium ascendens	≥10	13	2575.8
Pinus elliottii	≥10	12	1223.0
Leucothoe axillaris	<4	10	146.9
Clethra alnifolia	<4	10	54.2
Ilex coriacea	<4	10	86.4
Lyonia lucida	<4	10	295.7
Rhododendron sp.	<4	10	22.6
Total			8276.8

Table 19. Aboveground biomass estimate of woody vegetation (>1.3m high) in a holly phase of the titi swamp in Apalachicola, Florida.

Species	Size Class (cm dbh)	Regression # in Appendix B	Biomass (g/m^2)
Magnolia virginiana	<4	10	299.1
Cliftonia monophylla	<4	10	147.4
Cyrilla parviflora	<4	10	146.5
Cyrilla racemiflora	<4	10	17.6
Ilex myrtifolia	<4	10	113.2
Myrica cerifera	<4	10	72.0
Lyonia lucida	<4	10	38.8
Nyssa biflora	<4	10	22.0
Persea borbonia	<4	10	37.9
Hypericum reductum	<4	10	70.6
Total			965.1

Table 20. Aboveground biomass estimate of woody vegetation (>1.3m high) in a mixed swamp phase of the bay swamp in Apalachicola, Florida.

Species	Size Class (cm dbh)	Regression # in Appendix B	Biomass (g/m^2)
Magnolia virginiana	≥10	9	2245.2
Magnolia virginiana	10< ≥4	9	2435.0
Magnolia virginiana	<4	10	76.6
			4756.8
Cliftonia monophylla	≥10	7	7277.7
Cliftonia monophylla	10< ≥4	7	6669.0
			13946.7
Taxodium ascendens	≥10	13	301.2
Cyrilla racemiflora	10< ≥4	8	47.8
Cyrilla racemiflora	<4	10	9.9
			57.6
Total			19120.0

Table 21. Aboveground biomass estimate of woody vegetations (>1.3m high) in the black gum swamp in Apalachicola, Florida.

Species	Size Class (cm dbh)	Regression # in Appendix B	Biomass (g/m^2)
Nyssa biflora	≤10	11	4012.8
Nyssa biflora	10< ≥4	11	901.7
Nyssa biflora	<4	10	244.9
			5159.4
Pinus elliottii	≥10	12	791.4
Taxodium ascendens	≥10	13	4509.4
Taxodium ascendens	10< ≥4	13	475.3
Taxodium ascendens	<4	10	72.5
			5057.2
Cyrilla racemiflora	≥10	8	1171.2
Cyrilla racemiflora	10< ≥4	8	459.8
Cyrilla racemiflora	<4	10	156.1
			1787.1
Magnolia virginiana	10< ≥4	9	65.3
Magnolia virginiana	<4	10	10.4
			75.7
Cliftonia monophylla	10< ≥4	7	107.8
Cliftonia monophylla	<4	10	9.3
			117.1
Clethra alnifolia	<4	10	124.7
Ilex myrtifolia	<4	10	44.6
Ilex coriacea	<4	10	92.0
Lyonia lucida	<4	10	132.8
Leucothoe axillaris	<4	10	49.4
Total			13431.4

Table 22. Herbaceous biomass and litter estimates of the four community types in the titi shrub swamp in Apalachicola, Florida.

Community Type	Herbaceous Biomass (g/m^2)	Litter (g/m^2)
Titi swamp - titi phase	123.7	750.4
Titi swamp - holly phase	301.5	511.5
Bay swamp - mixed swamp phase	10.7	878.2
Black gum swamp	459.4	90.7
Average for titi shrub swamp (n=20)	224	558

Table 23. Aboveground biomass estimate of the four community types in the titi shrub swamp in Apalachicola, Florida.

Community Type	Biomass (g/m^2)					
	Tree Size Class	Shrub Size Class		Herbaceous	Total	Total
	≥10 cm dbh	10 ≤4 cm dbh	<4 cm dbh			kg/m^2
Titi swamp - titi phase	4868.8	1380.4	2027.6	123.7	8400.5	8.4
Titi swamp - holly phase	0	0	965.1	301.5	1266.6	1.3
Bay swamp - mixed swamp phase	9824.1	9151.7	86.4	10.7	19072.9	19.1
Black gum swamp	10484.7	2009.9	936.8	459.4	13890.8	13.9

up only 21% of the biomass of this community. The herbaceous component makes up 3% of the biomass of this community and is almost entirely composed of sphagnum (Table 16). Very little litter occurred in the black gum swamp as compared to the other community types. The aboveground biomass of the mixed swamp phase of the bay swamp is 19.1 kg/m^2 (Table 23). This is the greatest aboveground biomass of the four community types. Tree size class individuals make up 52% and shrub size class individuals make up 48% of the biomass of this community. The herbaceous component makes up less than 0.1% of the biomass of this community.

The leaf biomass to area ratios for black titi and sweetbay were greater for the 9 to 12 m vertical interval than for the 3 to 9 m vertical interval (Table 24). Red titi did not occur in the 9 to 12 m vertical interval. The estimated leaf biomass per ground area of the woody vegetation in the bay swamp community is 582 g/m^2 (Table 25).

The titi swamps had the lowest leaf litterfall (283 and 265 $g/m^2 \cdot yr$) and the bay swamp the highest leaf litterfall (584 $g/m^2 \cdot yr$) (Table 26) and this value is similar to the estimated leaf biomass per ground area for the bay swamp community (582 g/m^2) (Table 25). The black gum swamp had an intermediate leaf litterfall value. The average leaf litterfall for the titi shrub swamp is 359.3 $g/m^2 \cdot yr$ (Table 26). Forty-five percent of the annual leaf litterfall occurred in Autumn months from September through December, with the peak occurring in November. The second highest value was in the month of May.

For the three species measured the concentrations of total nitrogen and total phosphorus were as follows: leaf > branch > bole (Table 27). Sweetbay had the greatest concentration of nitrogen and phosphorus of all three species sampled at the study site for all components (Table 27). The holly phase of the titi swamp had the greatest concentration of nitrogen and phosphorus for herbaceous, litter and leaf litterfall (Table 28). Total nitrogen and total phosphorus standing stocks in the aboveground biomass at the study site were as follows: bay swamp > black gum swamp > titi swamp-titi phase > titi swamp-holly phase (Table 29). Total nitrogen and total phosphorus standing stocks in litter and leaf litterfall were greater in the bay swamp than in the other communities at the study site (Table 30).

The aboveground biomass estimate of the holly phase of the titi swamp (1.3 kg/m^2) is much less than values reported by Brown (1981) and Conner and Day (1982) but this community probably experienced a recent fire and is therefore in an early successional stage of development. The aboveground biomass estimates of the titi phase of the titi swamp (8.4 kg/m^2) and the black gum swamp (13.9 kg/m^2) are in the low range of values cited by Brown (1981) and Conner and Day (1982), while the mixed swamp phase of the bay swamp (19.1 kg/m^2) is in the intermediate range of these values.

The titi phase of the titi swamp and the black gum swamp communities can be characterized as still-water wetlands and the aboveground biomass estimate for these communities are low. Fire is an

Table 24. The leaf biomass to area ratio (LBAR) of black titi, red titi, and sweetbay at two vertical intervals (9 to 12 meters, and 3 to 9 meters) at the study site.

	LBAR (g biomass/m^2 leaf area)			
Species	9-12 meters	number of trees	3-9 meters	number of trees
Black titi	140.4	5	126.8	5
Red titi	-	0	142.5	10
Sweetbay	109.4	7	98.4	3

Table 25. Estimated leaf biomass per ground area (LBGA) of the woody vegetation (>1.3m high) in the bay swamp community.

Species	Size Class (cm dbh)	Regression # in Appendix B	Leaf Biomass (g/m^2)
Cliftonia monophylla	≥10	14	255.9
Cliftonia monophylla	10< ≥4	14	178.2
			434.2
Magnolia virginiana	≥10	16	86.9
Magnolia virginiana	10< ≥4	16	55.0
			141.9
Taxodium ascendens	≥10	17	4.1
Cyrilla racemiflora	10< ≥4	15	0.8
Cyrilla racemiflora	<4	15	0.7
Total			581.7

Table 26. Leaf litterfall (g/m^2) in the four communities in the titi shrub swamp in Apalachicola Florida from May 1982 through April 1983. $\bar{x}$ = mean, s = standard deviation, c.v. = coefficient of variation.

	Titi swamp titi phase			Titi swamp holly phase			Bay swamp mixed swamp phase			Black gum swamp		
Month	$\bar{x}$	s	c.v.	$\bar{x}$	s	c.v.	$\bar{x}$	s	c.v.	$\bar{x}$	s	c.v.
1983 Jan	26.9	20.1	74.7	14.0	7.1	50.7	24.6	5.1	20.7	12.0	5.8	48.3
Feb	21.4	21.0	98.1	10.9	2.6	23.8	13.7	6.4	46.7	13.4	5.4	40.3
Mar	15.4	10.2	66.2	15.7	3.8	24.2	34.4	13.8	40.1	18.2	7.2	39.6
Apr	25.7	10.0	38.9	23.6	9.9	41.9	50.3	22.1	43.9	25.9	3.4	13.1
1982 May	50.6	15.1	29.8	30.9	7.4	23.9	111.7	49.8	44.6	12.5	4.5	36.0
Jun	16.9	7.2	42.6	10.2	3.5	34.3	44.3	28.3	63.9	6.3	5.7	90.5
Jul	8.7	1.7	19.5	8.4	5.4	64.3	19.9	13.2	66.3	8.0	1.5	18.8
Aug	12.8	4.0	31.2	12.2	5.2	42.6	59.1	20.1	34.0	18.0	3.1	17.2
Sep	17.5	3.2	18.3	28.1	12.4	44.1	60.1	13.9	23.1	28.2	15.1	53.5
Oct	19.9	2.2	11.1	32.2	4.3	13.3	50.7	32.7	64.5	48.9	29.4	60.1
Nov	49.8	13.5	27.1	45.6	10.8	23.7	76.2	20.8	27.3	78.8	39.4	50.0
Dec	17.7	2.7	15.2	33.6	6.8	20.2	38.8	15.9	41.0	29.5	20.4	69.2
Total	283			265			584			305		

Table 27. Total nitrogen (TN) and total phosphorus (TP) concentrations (mg/g) of the bole, branch and leaf of black titi, red titi and sweetbay sampled at the study site. $\bar{x}$ = mean, s = standard deviation, c.v. = coefficient of variation.

	Black titi				Red titi				Sweetbay				3 species average			
TN	$\bar{x}$	s	c.v.	n	$\bar{x}$	s	c.v.	n	$\bar{x}$	s	c.v.	n	$\bar{x}$	s	c.v.	n
Bole	1.79	0.39	21.8	9	2.91	1.46	50.2	11	3.42	5.04	14.7	10	2.71	1.14	42.1	30
Branch	7.85	3.62	46.1	30	9.09	5.25	57.8	28	12.64	10.06	19.6	27	9.86	6.99	70.9	85
Leaf	8.88	2.32	26.1	29	10.32	3.40	32.9	26	28.57	6.35	22.2	30	15.92	10.15	63.8	85
TP																
Bole	0.08	0.05	62.5	9	0.11	0.07	63.6	11	0.15	0.06	40.0	10	0.12	0.06	50.0	30
Branch	0.20	0.13	65.0	30	0.19	0.14	73.7	28	0.73	0.61	83.6	27	0.37	0.24	64.9	85
Leaf	0.30	0.11	36.7	29	0.30	0.12	40.0	26	1.56	0.42	26.9	30	0.72	0.64	88.9	85

Table 28. Total nitrogen (TN) and total phosphorus (TP) concentrations (mg/g) of the herbaceous component, litter component and leaf litterfall in the four communities in the titi shrub swamp in Apalachicola, Florida. $\bar{x}$ = mean, s = standard deviation, c.v. = coefficient of variation.

	TN												TP											
	herbaceous				litter				leaf litterfall				herbaceous				litter				leaf litterfall			
Community type	$\bar{x}$	s	c.v.	n	$\bar{x}$	s	c.v.	n	$\bar{x}$	s	c.v.	n	$\bar{x}$	s	c.v.	n	$\bar{x}$	s	c.v.	n	$\bar{x}$	s	c.v.	n
Titi swamp - titi phase	7.79	3.47	44.5	14	6.10	1.77	29.0	19	5.74	1.09	19.0	3	0.34	0.20	58.8	14	0.17	0.08	47.1	19	0.22	0.08	36.4	3
Titi swamp - holly phase	9.99	3.89	38.9	14	11.32	3.43	30.3	19	9.10	0.31	3.4	3	0.46	0.23	50.0	14	0.29	0.12	41.4	19	0.36	0.08	22.2	3
Bay swamp - mixed swamp phase	9.20	4.88	53.0	15	8.59	2.63	30.6	18	7.90	0.40	5.1	3	0.36	0.14	38.9	15	0.23	0.09	39.1	18	0.25	0.01	4.0	3
Black gum swamp	7.23	3.28	45.4	17	6.67	2.01	30.1	17	7.02	0.22	3.1	3	0.23	0.11	47.8	17	0.21	0.09	42.9	17	0.18	0.04	22.2	3

Table 29. Total nitrogen (TN) and total phosphorus (TP) in the aboveground biomass in the four communities in the titi shrub swamp in Apalachicola, Florida.

Community Type	aboveground tree biomass g/m^2	bole biomass g/m^2	branch biomass g/m^2	leaf biomass g/m^2	herbaceous biomass g/m^2	Total aboveground standing stock gN/m^2	gP/m^2
Titi swamp- titi phase	8276.8	5744.1	2276.1	256.6	123.7		
	gN/m^2	15.6	22.4	4.1	1.0	43.1	
	gP/m^2	0.7	0.8	0.2	<0.1		1.7
Titi swamp - holly phase	965.1	669.8	265.4	29.9	301.5		
	gN/m^2	1.8	2.6	0.5	3.0	7.9	
	gP/m^2	0.1	0.1	<0.1	0.1		0.3
Bay swamp - mixed swamp phase	19120.0	13269.3	5258.0	592.7	10.7		
	gN/m^2	36.0	51.8	9.4	0.1	97.3	
	gP/m^2	1.6	1.9	0.4	<0.1		3.9
Blackgum swamp	13431.4	9321.4	3693.6	416.4	459.4		
	gN/m^2	25.3	36.4	6.6	3.3	71.6	
	gP/m^2	1.1	1.4	0.3	0.1		2.9

Table 30. Total nitrogen (TN) and total phosphorus (TP) in litter and leaf litterfall in the four communities in the titi shrub swamp in Apalachicola, Florida.

Community type	Litter Total standing stock gN/m^2	gP/m^2	Leaf Litterfall Total standing stock gN/m^2	gP/m^2
Titi swamp - titi phase	4.58	0.13	1.63	0.06
Titi swamp - holly phase	5.79	0.15	2.41	0.10
Bay swamp - mixed swamp phase	7.54	0.20	4.61	0.15
Black gum swamp	0.60	0.02	2.14	0.06

important component limiting biomass accumulation in the titi swamps but it rarely reaches the blackgum swamp. Large trees survive and the biomass is higher in the black gum swamp than in the titi swamp. The bay swamp community may be characterized as a slow-flowing wetland as it is located along the upper reaches of a stream. Bay swamps in general appear to be maintained by seepage from higher terrain, and may receive some nutrient input from these higher areas which may be unavailable to more isolated systems (Wharton *et al.* 1977). Sweetbay had the greatest concentration of nutrients in vegetation sampled at the study site (Table 27) and the bay swamp had the greatest concentration of nutrients of all the communities at the study site (Table 29). The increase in water and nutrient flow to the bay swamp may explain why this community has the greatest aboveground biomass at the study site.

Leaf litterfall provides an indication of biomass production as it is a component of biomass production and the bay swamp community has the greatest leaf litterfall at the study site (Table 26). In addition, the estimated leaf biomass of the woody vegetation in this community was similar to the value reported by Brown (1978) for a floodplain forest (Table 25) supporting the concept that greater water flow and nutrient input may account for higher production and biomass.

Even though there is some rapid water flow during certain times of the year, which may contribute to greater productivity and the subsequent accumulation of biomass in the bay swamp community relative to the other communities at the study site, the biomass is in the intermediate range of values cited above for forested wetlands because of low overall nutrient input to the system. Nutrient input is primarily from precipitation at the study site (Figure 3), as is the case for the dwarf cypress community cited above. Low nutrient input coupled with slow water flow accounts for the overall low biomass in this wetland.

The leaf biomass to area ratios for these three species at all vertical intervals (98.4 to 142.5) are lower than leaf biomass to area ratios reported by Brown (1978) for cypress in the dwarf cypress forest and in Austin Cary cypress dome at all vertical intervals (150 to 403), but they are similar to the value of 135 reported by Brown (1978) for fetterbush (an evergreen species) in Austin Cary cypress dome. The estimated leaf biomass per ground area of the woody vegetation in the bay swamp community (582 g/m^2) is greater than the value reported by Brown (1978) for Austin Cary cypress dome (465 g/m^2), and less than the value reported by Brown (1978) for a floodplain forest (663 g/m^2).

As is the case in the titi shrub swamp, cypress trees in the Okefenokee Swamp (Schlesinger 1978) and species analyzed in cypress domes (Post and Straub 1974) had concentrations of nitrogen and phosphorus as follows: leaf > branch > bole. Total nitrogen concentrations of branch (12.64 mg/g) and leaf (28.59 mg/g) material of sweetbay were similar to values for branch (11.30 mg/g) and leaf (21.0 mg/g) material of this species in a cypress dome (Post and Straub (1974). Total nitrogen (10.32 mg/g) and total phosphorus (0.29 mg/g) concentrations in the leaves of red titi were similar to values for current growth of this species in the Okefenokee Swamp (TN = 12.3 mg/g, TP = 0.48 mg/g) (Schlesinger 1978).

The average (three species) total phosphorus concentrations of the bole (0.11 mg/g), branch (0.38 mg/g) and leaf (0.72 mg/g) material were similar to values for bole (0.06 mg/g), branch (0.40 mg/g) and leaf (0.63 mg/g) material of black gum (a hardwood species) in Austin Cary dome (Deghi 1977). The average (three species) total phosphorus concentration of the leaf (0.72 mg/g) material is intermediate between values for cypress leaf (0.52 mg/g) material in a dwarf cypress community (Brown 1978) and for cypress leaf (0.84 mg/g) material in Austin Cary cypress dome (Straub and Post 1977). Although the average (three species) total phosphorus concentration of the bole (0.11 mg/g) material is greater than the values for cypress bole (0.034 mg/g) material in Austin Cary cypress dome (Straub and Post 1977) and for cypress bole (0.042 mg/g) material in a dwarf cypress community (Brown 1978), this value is similar to values for cypress bole (0.09 mg/g) material in the Okefenokee Swamp (Schlesinger 1978). The average total phosphorus concentration for leaf litterfall (0.25 mg/g) at the study site were similar to values for litterfall at Austin Cary dome (0.36 mg/g) (Deghi 1977).

The standing stock of total phosphorus in aboveground biomass in Florida cypress swamps as summarized by Brown (1981) and in titi shrub swamp communities are presented in Table 31. The standing stock of total nitrogen in aboveground biomass in titi shrub swamp communities are also presented. The standing stock of total phosphorus in aboveground biomass of different systems is positively related to the phosphorus input to those systems (Brown 1981). Total phosphorus inputs increased in the following order: dwarf cypress forest, Austin Cary cypress dome, floodplain forest. There was a corresponding increase in the standing stock of total phosphorus in the aboveground biomass in these systems. A similar relationship exists at the study site due to the coupling of water flow and nutrient inputs as the standing stock of total phosphorus and total nitrogen in the aboveground biomass increases along the gradient of water flow within the system (Figure 21) from the titi swamps to the black gum swamp and ultimately to the bay swamp.

The communities at this study site have increasing biomass and nutrient standing stocks corresponding to increasing water flow and nutrient input, indicating that there is a positive relationship between biomass and nutrient standing stocks, and hydrologic and nutrient inputs in the titi shrub swamp. When comparing forested freshwater wetlands it appears as though there is a positive relationship between productivity and hydrologic and nutrient inputs as well as between biomass and nutrient standing stocks and these inputs.

The annual average leaf litterfall at this titi shrub swamp was 359 $g/m^2/yr$ and is less than the other swamp systems cited above. This is probably due to low overall nutrient input to this system relative to the other swamp systems cited above. The highest leaf litterfall occurred in November but the

Table 31. Standing stock of total phosphorus in aboveground biomass in Florida cypress forests (Brown 1981), and in titi shrub swamp communities in Apalachicola, Florida. The standing stock of total nitrogen in aboveground biomass in titi shrub swamp communities are also presented.

	TP		*TP	*TN
	g/m^2		g/m^2	g/m^2
Dwarf cypress	0.26	Titi swamp	1.0	25.5
Austin Cary dome	2.45	Black gum swamp	2.9	71.6
Floodplain forest	4.78	Bay swamp	3.9	97.3

* source: Table 29

second highest value occurred in May (Table 26). Therefore there is a bimodal seasonal cycle at the study site which may be due to the development of leaves in the spring.

Vegetation characteristics at sites D25, D200, and D400 changed greatly as a result of wastewater discharge. Epiphytic and planktonic algae increased immediately. A short term study of the immediate effects on algal production was done in June 1985. Chlorophyll-*a* concentration of both planktonic and epiphytic algae was determined using the acetone extraction method (EPA 1973). In July, the floating macrophyte duckweed (*Lemna* sp.) appeared and completely covered the outfall area to a distance of 400 m from the outfall. Frog's-bit (*Lymnobium spongium*) appeared in November 1985 and spread rapidly until thick mats completely dominated the water column from the outfall area to about 300 m from the outfall. Cattails (*Typha* sp.) appeared in April 1986 and were sparse with only a few clumps at that time. Sphagnum moss (*Sphagnum* sp.) was not observed alive after August 1985 at sites D25, D220, and D400, and after October 1986 at all downstream sites. Open water areas not dominated by floating macrophytes were characterized by thick mats of epiphytic algae.

After the addition of wastewater, appearance of duckweed and disappearance of sphagnum moss were the most important change observed (Table 32). Woody species such as *Ilex* sp., *Lyonia lucida*, and *Cyrilla racemiflora* were not significantly changed after a year and a half of impact; observations differed due to the sampling method, in which transects were not laid in exactly the same line on the two sampling dates. The appearance of several previously herbaceous species in the mixed shrub swamp (near site D25) tended to increase the diversity of the site's vegetation.

Since periphyton was not measured prior to the addition of wastewater at site D25, comparisons were made to control sites U1200 and D2600 during the post-effluent sampling period. These sites were much lower in chlorophyll-*a* concentration of periphyton than site D25 (Table 33). Plankton algae remained low, suggesting that the major algal component in this system is epiphytic.

At sample locations between site D900 and D2600, up to approximately 1300 m from the effluent outfall, duckweed and thick mats of epiphytic algal were observed. This is in contrast to the persistence of sphagnum moss until approximately January 1986, and the total absence of duckweed and epiphytic algae at site D900. The vegetative response correlates with the observed water chemistry values at these sites.

A Center for Wetlands research study was initiated just prior to the start of wastewater discharge. To correct the ponding and channeling problem created by the unnecessary construction at the site, a long term solution was attempted. Both bald and pond cypress were planted in these bulldozed areas, in the hope that nutrient and water uptake would be enhanced by the presence of these trees.

Table 32. Herbaceous ground cover at the Apalachicola wetland.

	RELATIVE % COVER	
	PRE-EFFLUENT **	POST-EFFLUENT (10-25-86)
MIXED SHRUB SWAMP (Near site D25)		
Andropogon glomeratus	0.00	0.25
Aster subulatus	0.00	0.25
Bare hummock	0.00	1.88
Carex sp.	9.02	0.00
Clethra alnifolia	0.00	0.13
Cliftonia monophylla	5.90	0.00
Cyrilla parviflora	4.86	0.00
Eupatorium capillipes	0.00	0.63
Hypericum sp.	11.46	0.00
Ilex cassine	0.00	0.34
Ilex myrtifolia	2.16	0.00
Ilex sp.	0.00	1.13
Lachnanthes sp.	2.54	0.00
Lemna sp.	0.00	83.84
Ludwigia leptocarpa	0.00	1.13
Lyonia lucida	14.93	1.25
Magnolia virginiana	0.00	0.75
Myrica cerifera	4.79	0.00
Sabal palmetto	2.14	0.00
Salix caroliniana	0.00	0.04
Sphagnum sp.	42.20	0.00
Triadenum sp.	0.00	0.63
Vitus sp.	0.00	0.04
Woodwardia virginica	0.00	7.13
TITI SHRUB SWAMP (Near site D400)		
Bare hummock	12.30	11.13
Clethra alnifolia	2.24	1.29
Cliftonia monophylla	0.13	0.00
Cyrilla racemiflora	1.05	0.04
Ilex cassine	2.02	2.29
Ilex glabra	0.00	1.26
Lemna sp.	0.00	80.51
Leucothoe axillaris	0.26	0.13
Lyonia lucida	20.48	5.38
Pieris phillyreifolia	0.82	0.00
Rhododendron sp.	1.51	0.13
Sacciolepis striata	0.00	0.25
Sphagnum sp.	59.20	0.00

Table 32. (continued.)

	RELATIVE % COVER	
	PRE-EFFLUENT **	POST-EFFLUENT (10-25-86)
GUM POND (Near site D900)		
Bare hummock	6.18	42.97
Clethra alnifolia	0.00	4.07
Cyrilla racemiflora	5.37	11.20
Ilex cassine	5.77	6.11
Ilex glabra	0.00	1.02
Lemna sp.	0.00	5.09
Lyonia lucida	2.64	26.48
Nyssa sylvatica (biflora)	0.00	2.04
Rhododendron sp.	0.53	1.02
Sphagnum sp.	79.50	0.00
BAY HEAD (Near site 02700)		
Bare hummock	45.49	68.68
Carex sp.	11.30	2.39
Clethra alnifolia	0.90	0.30
Cliftonia monophylla	0.44	0.62
Cyrilla racemiflora	1.18	0.25
Lyonia lucida	36.84	27.64
Magnolia virginiana	0.00	0.12
Sphagnum sp.	3.85	0.00

* Woody species less than .5 m tall. All herbaceous species regardless of height.

** Best et al., 1987.

Table 33. Chlorophyll-a concentrations of selected sites in a forested wetland near Apalachicola, Florida.

Source	Date	Site	Concentration mg/m^3
Phytoplankton	May 1982	U400	
(Pre-effluent)		U25	
		D25	<0.02
		D900	
		D2600	
Phytoplankton	June 3, 1985	D25	0.023*
(Post-effluent)		D200	<0.02 *
	June 16, 1985	D25	.029*
		D200	<0.02 *
Periphyton	May 25-June 16, 1985	U1200	<0.02
(Post-effluent)		D25	0.360
		D2600	0.046

*Mean of two replicate samples

BACTERIA

Purpose

The purpose of this portion of the investigation was to determine the existing or potential public health hazard associated with wastewater recycling through the wetland. Comparatively little is known about the fate of bacteria and viruses in wetland systems. Several studies have addressed the removal of bacterial indicators and pathogens after wastewater discharge into wetlands (Boyt 1976; Price 1976; Fox and Allison 1976; Wellings *et al.* 1975; Butner 1983; Bitton *et al.* 1985; Kadlec *et al.* 1979; Vaughn and Landry 1980). In general an improvement in bacteriological water quality of wastewater has been observed following passage of effluent through a wetland system. Less is known about the fate of viruses than of bacteria.

The bacteriological standards for class III waters are set forth in Section 17-3.121 of the Florida Administrative Code. The number of total coliform bacteria cannot exceed 1000/100 ml as a monthly average, nor exceed 2400/100 ml at any time. The number of fecal coliform bacteria cannot exceed 200/100 ml as a monthly average nor exceed 800/100 ml on any one day. Use of coliforms as indicators of contamination by domestic wastewater is well documented, particularly the fecal coliform group.

The ratio of fecal coliforms to fecal streptococci is useful in determining the origin of the pathogens. A high ratio indicates wastes of predominantly human origin.

Methods

Monitoring for Bacterial Indicators and Pathogens

Total coliforms, fecal coliforms and fecal streptococci were enumerated according to membrane filter procedures detailed in *Standard Methods* (APHA 1982). Due to high content of solids in the strand water the enumeration of total coliforms, fecal coliforms and fecal streptococci was changed to the MPN procedure as described in *Standard Methods* after wastewater discharge was begun. Total heterotrophic bacteria were enumerated by spread plating 1.0 ml of an appropriate dilution on Standard Methods Agar.

Enteroviruses

Recovery of enteroviruses from the strand water. Indigenous enteroviruses were monitored in the strand water using the following method. Ten to 80 gal (37 to 300 l) of water were passed through a filterite filter (0.45 or 0.25 μm) after adjusting the pH to 3.5 with 0.1 acetic acid. The filter was then transferred to the lab on ice and further processed for adsorbed virus. The filter was eluted with 800 ml of 0.6 M TCA + 0.1 M lysine, pH 9.0. The pH of the eluate was adjusted to 7.0 with 0.5 M glycine, pH 11.0, and 0.005 M aluminum chloride was added. The pH was then readjusted to 7.0 with 1 m Na_2CO_3. The eluate was mixed for 10 min, and subsequently centrifuged at 4,080 x G for 5 min. The pellet was solubilized in 3 volumes of 5% beef extract + 0.1 M EDTA, pH 9.0, and centrifuged at 16,300 x G for 10 min. The supernatant was then neutralized and dialyzed against PBS at 4°C overnight. The dialyzed sample was concentrated by organic flocculation for the March and April 1983 samples, all subsequent samples were concentrated by Ultracentrifugation.

Indigenous virus assay. Cultures of 8GM cells were inoculated and overlaid with 2% agar and Eagles MEM and scored for plaques for up to 4 weeks.

Results and Discussion

Bacterial Indicators

Table 34 shows the concentration of the bacterial indicators in the strand system prior to the addition of wastewater. It can be seen that the levels of all indicator organisms are generally low. However, it is also seen that the levels fluctuate by as much as 2 log units at each station. This is not unexpected as the water level and input would be dependent upon the weather conditions for the sampling period.

Table 35 shows the indicator levels just prior to, and for 1 mo post sewage discharge into the wetland. The high turbidity of the strand water resulted in the need to change the enumeration procedure from the membrane filter method to the MPN procedure. In general, the levels of all microorganisms monitored increased after the wastewater discharge was started. However, a definite concentration gradient with flow was not seen. In fact, large fluctuations in bacterial concentrations were seen at stations in the direction of flow. This is probably due to the short time since discharge was started that had elapsed when samples were taken, before the system had a chance to stabilize.

Table 34. Total bacterial indicators in titi-shrub swamp near Apalachicola, Florida, prior to the addition of sewage effluent.

Sample date	Sampling Location								
	1	2	4	5	6	Culvert	W4	W7	Pond
Total Coliform (CFU/100ml)									
1/13/85	100	TNTC[1]	130	500	650	620	ND[2]	ND	---[3]
2/20/85	175	1230	---	44	142	60	---	---	55
3/30/85	225	14	---	43.5	580	250	---	---	---
4/04/85	400	1500	60	730	570	825	ND	ND	460[4]
Fecal Coliform (CFU/100 ml)									
1/13/85	9	TNTC[1]	45	3	120	217	ND[2]	ND	---[3]
2/20/85	20	14	---	ND	2	1	---	---	8
3/30/85	116	5	---	6	12.5	ND	---	---	---
4/04/85	420	1000	38	35	390	484	---	---	240[4]
Fecal Streptococci (CFU/100 ml)									
1/13/85	ND[1]	285	1	4	14	5	ND	ND	---[2]
2/20/85	15	2	---	10	4	ND	---	---	2
3/30/85	18	ND	---	2	312	1	---	---	---
4/04/85	82	66	5	4	32	83	---	---	240[3]

Table 35. Total bacterial indicators in titi-shrub swamp near Apalachicola, Florida, after the addition of sewage effluent.

Sample date	Sampling Location PCO	PC25	PC50	PC75	PC100	PC150	PC175	PC200	Holding pond	1	5	Groundwater Well A	B	E
Total Coliform (MPN/100 ml)														
5/19/85	75	1100	2400	460	150	---	---	---	1100	---	---	---	---	---
6/03/85	ND	---	240	---	93	240	230	---	---	---	3.6	---	---	---
6/29/85	4600	---	460	---	2400	4600	---	3.6	---	6.2	---	1100	240	ND
Fecal Coliform (MPN/100 ml)														
5/19/85	75	43	23	3.6	3.6	---	---	---	47	---	---	---	---	---
6/03/85	ND	---	240	---	93	240	230	---	---	---	3.6	---	---	---
6/29/85	4600	---	460	---	2400	4600	---	3.6	---	6.2	---	1100	240	ND
Fecal Streptococi (MPN/100 ml)														
5/19/85	43	1100	460	15	ND	---	---	---	23	---	---	---	---	---
6/03/85	9.1	---	240	---	93	240	3.6	---	---	---	ND	---	---	---
6/29/85	2400	---	460	---	11000	4600	---	ND	---	230	---	ND	ND	ND
Plate Count (MPN/100 ml)														
5/19/85	3.6	7.5	3.9	.83	.18	---	---	---	1.5	---	---	---	---	---
6/03/85	700	---	310	---	130	96	3.3	---	---	---	0.47	---	---	---
6/29/85	490	---	170	---	390	390	---	5.1	---	130	---	360	310	17

1 TNTC - Too numerous to count.
2 ND - None detected
3 No sample (Sample too dirty to filter or no sample taken).
4 MPN - Most probable number per 100 ml

Enterovirus

Attempts to isolate indigenous enterovirus at stations U1200, D2600, and the outfall area prior to discharge revealed no viruses in volumes of up to 80 gal. Attempts after the discharge of wastewater were made at stations up to 100 m from the outfall, in the holding pond (just after chlorination of the effluent), at station U1200, and D2600. These attempts were also negative. These results are not unexpected, considering the low number of samples, short time after discharge, and the season in which these samples were taken. The high temperatures would result in reduced survival of enteroviruses and consequently lower numbers discharged from the treatment plant. In addition, several workers have shown that the peak months to isolate viruses in the environment are early autumn and early winter (Berg and Berman 1980; Bitton 1980; Scheuerman 1984).

Conclusions

The timing of a short time after start of discharge of wastewater and the resulting low number of samples make conclusions about the levels and behavior of indicator bacteria and enteroviruses difficult. To better understand the effects of wastewater discharge on the potential health threats from this practice more data is required. Since many seasonal factors affect the discharge levels and survival of pathogenic bacteria and viruses, at least 1 yr of monthly samples is required to begin understanding the survival and transport patterns of these organisms in the wetland system. In addition, the enumeration of indigenous bacteriophage for comparison to enteroviruses would be beneficial. The use of bacteriophage as an indicator for enteroviruses would reduce the cost and time involved in the enterovirus isolations. However, data supporting the use of bacteriophage is lacking.

SOILS

Purpose

The purpose of this portion of the research was to evaluate the potential of the Apalachicola wetland soils for storage of phosphorus. These data can be used to assess potential for long-term functioning of the wetland soils as a phosphorus sink.

Literature Review

Phosphorus retention by soils may be an advantage of using wetlands as an alternative for wastewater treatment. Therefore, emphasis has been placed on using adsorption isotherms in order to predict soil types that would be amenable to receiving wastewater (Sommers and Sutton 1980).

A phosphorus adsorption isotherm is used to describe the relationship between the amount of phosphorus sorbed and that remaining in solution at constant temperature. Several equations developed for gas-solid systems have been used to interpret the sorption of phosphate on charged surfaces. The adsorption data are fit to isotherms described by the equations. The isotherms can be used to give a relative adsorption maximum, interpreted as a "quantity" factor, indicating the capacity of the soil to adsorb and thus retain phosphorus.

The Langmuir equation is based on the assumptions that adsorption is on a finite number of localized sites, the energy of adsorption is constant, and maximum adsorption corresponds to a complete monolayer. Thus the equation describes a finite limit to adsorption so that a maximum value may be obtained. The Langmuir equation is described as follows:

$$x/m = KCb / 1 + KC$$

where K is a constant related to the adsorption energy, C is the equilibrium phosphorus concentration, and x/m and b are phosphorus adsorbed and maximum phosphorus adsorption per unit weight of soil respectively. In the linear form the equation becomes:

$$Cm/x = C/b + 1/Kb$$

and a plot of Cm/x versus C should give a straight line of slope 1/b from which b, the adsorption maximum can be calculated.

Straight line isotherms have been obtained when results from a limited concentration range are plotted according to the Langmuir equation (Olsen and Wantanabe 1957). Although the Langmuir equation in its linear form has been used frequently in phosphorus adsorption studies the adsorption curves may not be linear over a wide concentration range (Olsen and Wantanabe 1957; Rennie and Mekercher 1959; Gurney 1970; Bache and Williams 1971; Fitter and Sutton 1975). There are many possible explanations for the nonlinearity, but where it does occur the Fruendlich and other equations may be used to fit the adsorption data.

The Fruendlich equation is based on the assumption that the surface consists of sites at which the adsorbate molecules interact laterally, resulting in a continuous distribution of bonding energies that decrease exponentially with increasing saturation of the surface. The Fruendlich equation can be described as follows:

$$x/m = aC^b$$

where x/m and C are as before and a and b are constants that vary between soils. In the linear form the equation becomes:

$$\log x/m = \log a + b \log C$$

and a plot of log x/m versus log C should give a straight line. The Fruendlich equation has been found to give a good fit over a wide range of soils and concentrations (Gurney 1970; Fitter and Sutton 1975; Barrow and Shaw 1975; Barrow 1978).

The Tempkin equation is derived from the Langmuir equation but, like the Fruendlich equation, is based on the assumption of a continuous distribution of bonding energies. In this case the energy of adsorption decreases linearly with increasing surface coverage. The Tempkin equation can be described as follows:

$$xb/m = (RT/B) \log AC$$

where x, b, and C are as before and A and B are constants. A plot of x/m versus log C should give a straight line.

The phosphorus adsorption maxima of soils can be calculated from the slope of the regression lines according to the Langmuir equation. The Fruendlich equation does not have this characteristic and therefore a quadratic regression analysis of the adsorption data developed by Yuan and Lucas (1982) can be used as an alternative to obtain the adsorption maxima. If Y is the phosphorus adsorbed and X the equilibrium phosphorus concentration, then the quadratic equation is as follows:

$$Y = a_0 + a_1X + a_2X^2$$

and the first derivative of this equation is equal to zero when Y reaches the maximum, or

$$dY/dX = a_1 + 2a_2X = 0.$$

Therefore the phosphorus concentration (C) at the adsorption maximum would be

$$C = X = -a_1/2a_2.$$

The adsorption maximum is obtained by substituting $-a_1/2a_2$ for X in the quadratic equation. If this equilibrium phosphorus concentration and the corresponding adsorption maximum derived from the quadratic equation are correct, then substitution of the C values in the other equations should give comparative adsorption maxima (Yuan and Lucas 1982).

There has been a good deal of research on the nature of phosphorus adsorption in soils. There is debate as to whether or not organic matter increases phosphorus adsorption. A number of researchers report a decrease in phosphorus adsorption by soils in the presence of organic matter, the decomposition of which produces organic acids which form stable complexes with aluminum and iron and consequently block phosphorus retention (Singh and Jones 1976). Other workers reported that organic matter increases phosphorus retention by the soil possibly as a result of microbial assimilation. Adsorption and leaching of phosphorus in acid organic soils and high organic matter sand was determined by Fox and Kamprath (1971). These soils in which the colloids are organic had relatively low phosphorus adsorption capacities relative to mineral soils. Phosphorus adsorption by organic matter was negligible because any adsorption that occured was due to the cations associated with organic matter (Wild 1950). Organic soils with only a trace of inorganic minerals have little aluminum or iron to be released for bounding with added phosphorus. Thus although the influence of organic matter on phosphorus adsorption has been debated, organic matter appears to affect phosphorus adsorption in an indirect manner (Berkheiser *et al.* 1980).

Soluble inorganic phosphorus is readily immobilized in soils by adsorption and precipitation reactions with aluminum and iron under acid conditions (Nur and Bates 1979; Nichols 1983). Low phosphorus adsorption has been observed in sandy soils with low clay content and is primarily correlated with low content of extractable iron and aluminum (Ballard and Fiskell 1974; Yuan and Lucas 1982). Layer silicate minerals have low phosphorus fixing potential but amorphous colloids and sesquioxides are effective at fixing phosphorus. The less crystalline the form of the sesquioxides, the greater their capacity to sorb phosphorus. Phosphate ions are thought to be chemically adsorbed onto the surfaces of hydrous oxides of iron and aluminum by ligand exchange, the displacement of water molecules and hydroxyl groups coordinated with the iron and aluminum atoms and the coordination of oxygen atoms in the phosphate ions with the iron and aluminum (Nichols 1983). In addition to this chemical adsorption Ryden and Syers (1977) presented evidence for a more physical type of adsorption that becomes operational as the chemical adsorption sites approach saturation at higher equilibrium concentrations of phosphorus in solution (Nichols 1983).

The chemical and physical adsorption of phosphate onto the surface of soil minerals is a rapid process but slower phosphate fixation does occur and has been attributed to the shift of physically adsorbed phosphorus to chemically adsorbed forms, the diffusion of phosphorus adsorbed on the surface of porous oxides of aluminum and iron to positions inside the soil matrix, and the precipitation of crystalline aluminum and iron phosphates (Nichols 1983). There is uncertainty as to the exact mechanisms involved in phosphorus retention in the soil. There is a continuum of reaction mechanisms and there is little concern for distinguishing between adsorption and precipitation reactions as both phenomena can be considered together as sorption (Berkheiser *et al.* 1980). Adsorption and precipitation of phosphorus by soils is not necessarily a permanent sink for added phosphorus, it is at least partially reversible. A reduction in the phosphorus concentration in the solution in contact with the soil may release some phosphorus into solution (Nichols 1983).

Effort has been directed towards identifying measurable soil parameters that can be related to the phosphorus adsorption capacity of a soil. The active (exchangeable + amorphous) forms of aluminum provide the best single index of phosphorus retention in Coastal Plain forest soils (Ballard and Fiskell 1974). The contribution of active forms of iron to phosphorus retention was at least the equal of aluminum on a per unit weight basis. Poorly crystalline and amorphous oxides and hydroxides of aluminum and iron were postulated to play a primary role in phosphorus retention in flooded soils (Khalid *et al.* 1977). An organic matter aluminum peat complex in acid soils strongly adsorbed orthophosphate ions (Bloom 1981). Phosphorus adsorption was highly correlated with organic matter content and exchangeable aluminum content in a study that evaluated the phosphorus retention capacity of retention-detention wetland soils (Sompongse 1982). She proposed in light of Bloom's (1981) findings, retention through an organic aluminum complex in the soils with high aluminum content. In soils with high iron content, iron seemed to play an important role in phosphorus retention.

Tamm oxalate extractable aluminum and in some cases Tamm oxalate extractable iron have the best correlation with phosphorus sorption in mineral soils (Lopez-Hernandez and Burnham 1974; Ballard and Fiskell 1974). Similar results were found in some wetland organic soils (Richardson 1985). The Tamm oxalate extraction dissolves the amorphous and poorly crystalline oxides of aluminum and iron that have been postulated to play a primary role in phosphorus retention in flooded soils.

Methods

Replicate soil cores were taken with acrylic tubing (4 cm i.d.) at four sampling stations (2, 4, 5 and 6, Figure 2) representing the four wetland community types. Each 20 cm long core was divided into 5 cm increments. Each 5 cm increment was placed in an individual urine cup and then stored on ice during transport to the laboratory in Gainesville. Total organic carbon was determined by the Walkey-Black method and percent carbon was assumed to be 58% of organic matter (Allison 1965).

An Orion Model 399A Ionanalyzer with a glass electrode was used to measure pH in deionized water with a soil liquid ratio of 1:1 (v:v) (Peech 1965). Total nitrogen including nitrate was determined by the semi-micro Kjeldahl method (Bremner 1965). Total phosphorus was determined by the ignition method and 0.1 N HCL extraction (Anderson 1976). This procedure converts all the phosphorus to the orthophosphate form which was determined colorimetrically with the ascorbic acid method (Murphy and Riley 1962).

Phosphorus adsorption was measured for soils sampled at two stations (4 and 5) at two depths (0-5 cm and 15-20 cm). Duplicate 1-gram air-dried samples were shaken for 24 hr at a constant temperature of 22°C with 25 ml of a 0.01M $CaCl_2$ electrolyte solution. One ml of toluene was added to eliminate microbial activity. Varying concentrations of phosphorus were added as follows: 0, 2.5, 5, 7.5, 10, 15, 20, 30, 40, and 50 mg/l as $Ca(H_2PO_4)_2$. The average concentration of phosphorus in secondarily treated wastewater effluent is within this range. The samples were then centrifuged and the supernatant solutions were analyzed for phosphorus by the ascorbic acid method (Murphy and Riley 1962). The amount of phosphorus removed by the soil from the solution was considered adsorbed.

The adsorption data are plotted in four ways: the regular plot (x/m versus C), linear Langmuir plot (Cm/x versus C), linear Fruendlich plot (log x/m versus log C), and Tempkin plot (x/m versus log C), where x/m and C represent the amount of phosphorus adsorbed by unit mass of soil (µg/g) and equilibrium phosphorus concentration in the solution (µg/ml) respectively. Linear regression analysis was performed on the last three plot types to obtain regression lines and coefficients of correlation (R).

The soils evaluated for phosphorus adsorption were also analyzed for extractable phosphorus, extractable iron and extractable aluminum by two methods: A 0.1 N HCL extraction (Mestan 1986), and Tamm oxalate method (Saunders 1965). Phosphorus was measured by the ascorbic acid method (Murphy and Riley 1962), and iron and aluminum were analyzed using flame atomic absorption spectrophotometry (Mestan 1986). The Tempkin equation had the highest correlation when both soil types were considered together. Therefore, the adsorption maxima obtained by substitution of the equilibrium phosphorus concentration derived from the quadratic equation into the Tempkin equation was correlated with measured soil properties. The phosphorus sorption index was computed from a single-point uptake adsorption value (x/m) corresponding to an equilibrium phosphorus concentration of 10 μg/ml. This single-point value was computed from the individual quadratic equations for each soil. This index was chosen on the basis of simplicity and the 10 μg/ml equilibrium phosphorus concentration is within the range of phosphorus concentrations found in secondarily treated wastewater. Linear regression analysis was performed to relate the adsorption maxima and the phosphorus sorption index with measured soil properties.

Results and Discussion

Soils

There are two soil types at the study site (Table 36). The communities dominated by species of titi (titi swamps and bay swamp at the study site) occur on Typic Humaquepts of the Rutlege Series. These Inceptisols (from inceptum or beginning) are nearly black or peaty, strongly acidic, very poorly drained sandy soils (SCS 1975). The black gum swamp occurs on Terric Medisaprists of the Pamlico Series. These Histosols (from histos or tissue) are extremely acidic very poorly drained organic soils (SCS 1975). Both soils occur where the water table is at or near the surface for long periods of the year and ponding is common.

Aquepts (the suborder of Inceptisols at the study site) and Saprists (the suborder of Histosols at the study site) have bulk densities greater than 0.2 g/cm^3 and 0.85 g/cm^3, respectively (SCS 1975). This is the case for these soils at the study site (Table 37). Histosols that are saturated with water contain at least 12 to 18 percent organic carbon depending on the clay content of the mineral fraction and kind of materials (SCS 1975). The soils at the study site dominated by black gum meet this criterion; they have from 24 to 50 percent organic carbon and are therefore organic soils (Table 37). The soils at the study site dominated by the species of titi all have less than 12 percent organic carbon and are therefore mineral soils. Bulk density increased with depth while percent organic matter decreased with depth for all soils at the study site (Table 37). The pH also increased with depth (Table 37).

Table 36. Classification of soils of the titi shrub swamp in Apalachicola, Florida.

Order	Suborder	Great Group	Subgroup	Family	Series	Community Type
Inceptisols	Aquepts	Humaquepts	Typic Humaquepts	Sandy, siliceous, thermic	Rutlege	Titi swamp, titi phase holly phase
						Bay swamp mixed phase
Histosols	Saprists	Medisaprists	Terric Medisaprists	Sandy, siliceous, dysic, thermic	Pamlico	Blackgum swamp

Table 37. Soils analysis of the titi shrub swamp in Apalachicola, Florida.

Community Type	Station	Depth cm	Bulk Density g/cm³	Organic Carbon %	Organic Carbon mg/g	Organic Matter %	pH	TN mg/g	TP mg/g
Titi swamp- titi phase	2	5	0.86	9.7	97.0	16.7	3.8	2.45	90.63
		10	1.00	7.8	77.0	13.4	3.8	2.06	59.00
		15	1.30	5.3	53.2	9.2	3.9	1.40	46.14
		20	1.43	3.5	35.3	6.1	4.0	0.42	45.13
Titi swamp - holly phase	4	5	0.90	7.0	69.7	12.0	4.1	3.08	64.19
		10	1.12	4.0	39.8	6.9	4.2	1.69	42.62
		15	1.17	3.7	36.6	6.3	4.2	1.20	38.75
		20	1.41	3.3	33.4	5.8	4.2	0.95	37.13
Bay swamp - mixed phase	6	5	1.02	4.6	45.5	7.8	4.5	1.16	38.37
		10	1.15	3.6	36.2	6.2	4.6	0.85	27.50
		15	1.53	2.7	27.3	4.7	4.8	0.50	16.87
		20	1.68	2.1	21.0	3.6	4.8	0.36	14.63
Black gum swamp	5	5	0.65	50.0	499.7	86.1	3.7	3.54	180.00
		10	0.99	36.8	368.4	63.5	3.9	2.46	182.25
		15	1.19	29.0	289.6	49.9	4.2	1.90	204.00
		20	1.24	24.0	240.5	41.5	4.2	1.53	224.87

Total nitrogen in the soils is greatest at the surface where organic matter is greatest and decreases with depth as does organic matter (Table 37). Total phosphorus was highest in the surface layers of the mineral soils at the study site. Total phosphorus increased with depth in the organic soils at the study site. Overall, soils in the study site were low in nitrogen and phosphorus.

Phosphorus Adsorption

Adsorption data were fit to isotherms described by the Langmuir, Fruendlich and Tempkin equations and plotted as follows; the regular plot (x/m versus C), linear Langmuir plot (Cm/x versus C), linear Fruendlich plot (log x/m versus log C), and Tempkin plot (x/m versus log C), where x/m and C represent the amount of phosphorus adsorbed by unit mass of soil (μg/g) and equilibrium phosphorus concentration in the solution (μg/ml) respectively.

Plots of the phosphorus adsorption data for the two soils indicate that the amount of phosphorus adsorbed by the soil varies with soil type (Figures 22 through 25). In the regular plot, the amount of phosphorus adsorbed is plotted against the equilibrium phosphorus concentration. The regular plots for the site four mineral soils are curved which suggests a finite limit to adsorption, characteristic of the Langmuir equation. The regular plots for the site five organic soils are straight lines which suggests a decrease in the energy of adsorption with increasing surface coverage, characteristic of the Fruendlich and Tempkin equations.

For each soil type the regular plots are followed by the linear Langmuir, linear Fruendlich, and Tempkin plots. The fit of these equations can be evaluated by comparing the coefficient of correlation of their regression lines (Table 38). The site four mineral soils show high correlation when plotted with the Langmuir equation and lower correlation when plotted with both the Fruendlich and Tempkin equations. The site five organic soils show the highest correlation when plotted with the Fruendlich equation, high correlation when plotted with the Tempkin equation, and no correlation when plotted with the Langmuir equation.

The adsorption maxima and the equilibrium phosphorus concentration are useful in describing the phosphorus retention capacity of the soil. The adsorption maxima obtained directly from the slope of the linear Langmuir equation and the adsorption maxima calculated by substituting the equilibrium phosphorus concentration into the quadratic, Langmuir, Fruendlich, and Tempkin equations are similar (Table 39). The adsorption maxima for the site four mineral soils were less than 230 μg/g soils. The adsorption maxima for the site five organic soils were from 294 to 2888 μg/g soil. The adsorption maxima all increased with depth.

Tamm oxalate extractable phosphorus concentrations were higher than the 0.1 N HCl extractable phosphorus concentrations (Table 40) but less than total phosphorus concentrations. The Tamm oxalate

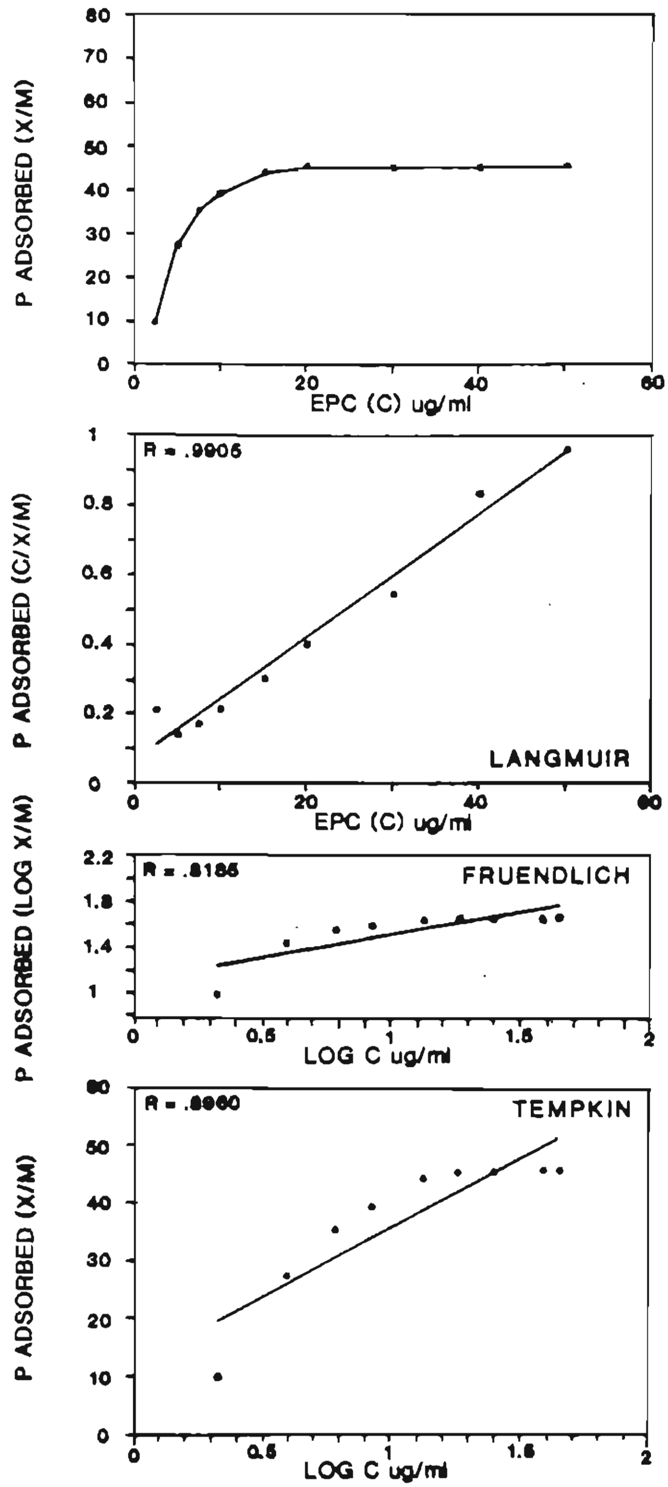


Figure 22. Phosphorus adsorption isotherms for the mineral soil (Rutlege Series) at a depth of 0-5 cm at the Apalachicola wetland. Plots: regular, Langmuir, Fruendlich and Tempkin.

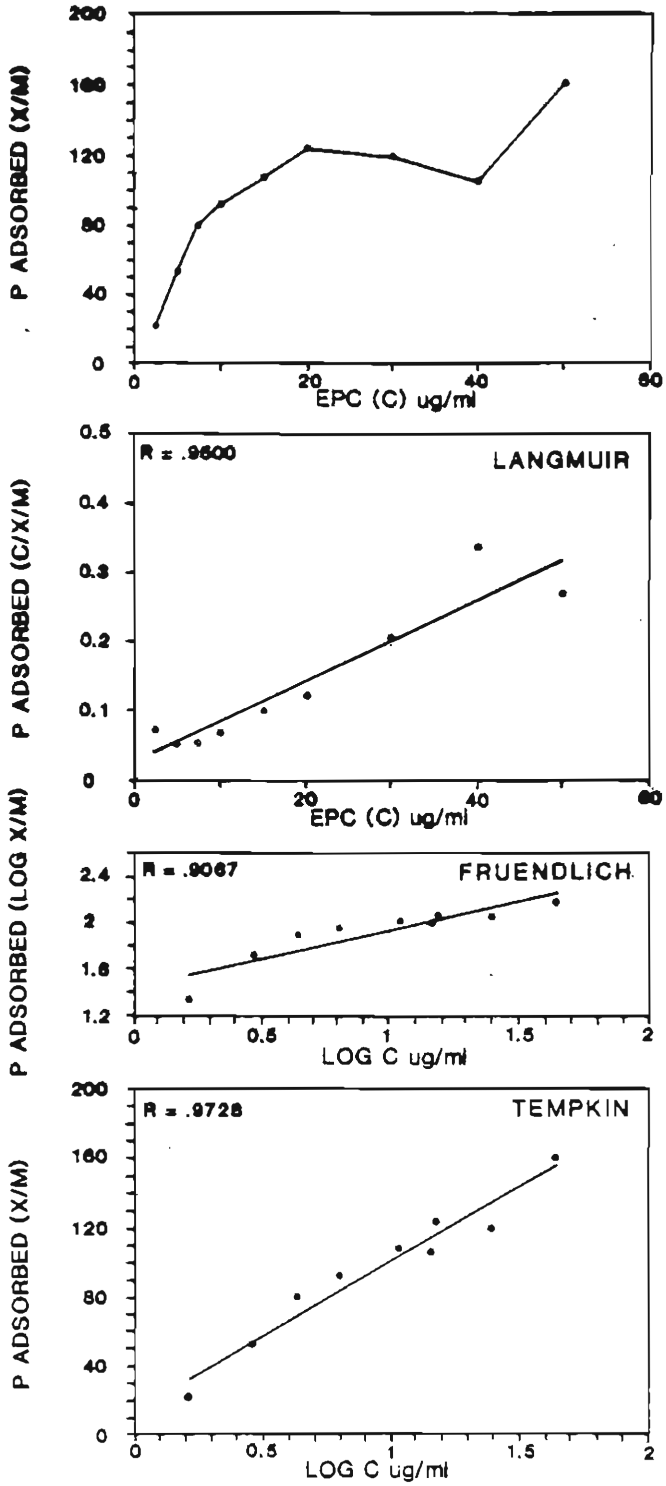


Figure 23. Phosphorus adsorption isotherms for the mineral soil (Rutlege Series) at a depth of 15-20 cm at the Apalachicola wetland. Plots: regular, Langmuir, Fruendlich and Tempkin.

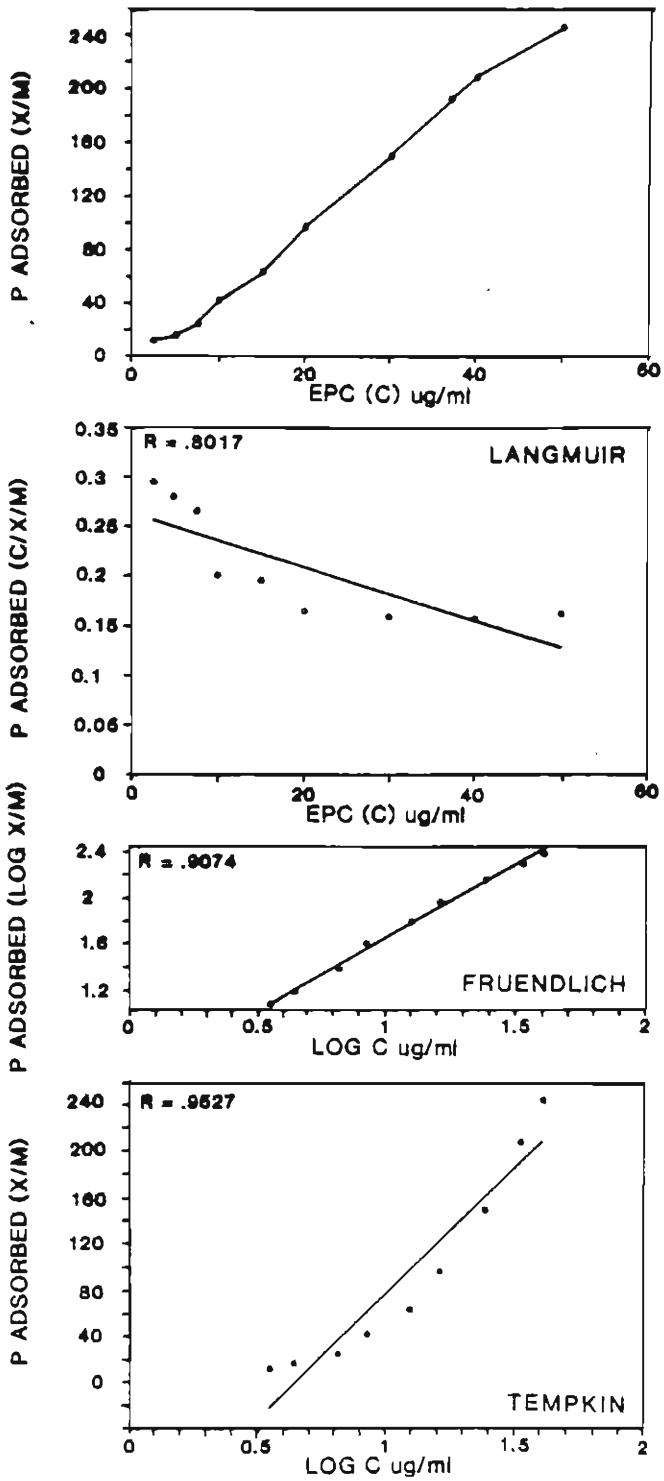


Figure 24. Phosphorus adsorption isotherms for the organic soil (Pamlico Series) at a depth of 0-5 cm at the Apalachicola wetland. Plots: regular, Langmuir, Fruendlich and Tempkin.

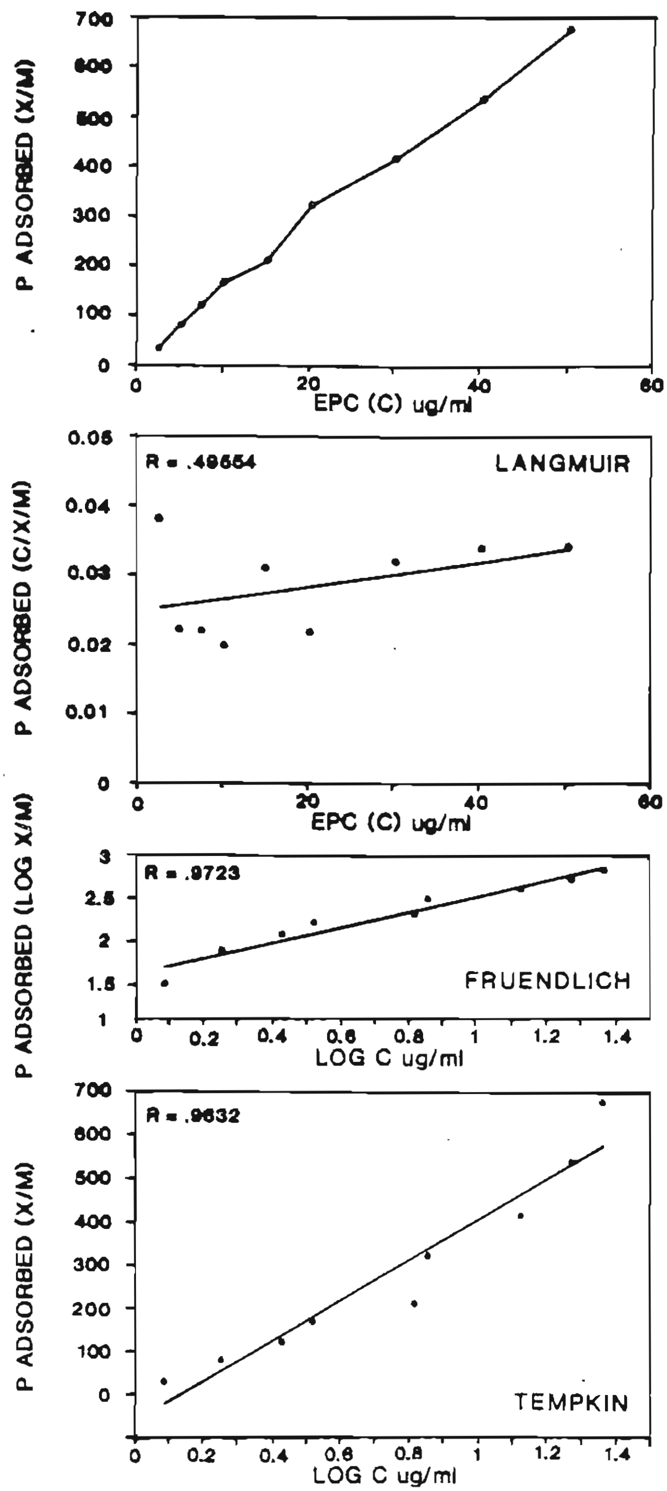


Figure 25. Phosphorus adsorption isotherms for the organic soil (Pamlico Series) at a depth of 15-20 cm at the Apalachicola wetland. Plots: regular, Langmuir, Fruendlich and Tempkin.

Table 38. Coefficients of correlation (R) between the adsorption of added phosphorus by study site soils and the equilibrium phosphorus concentrations (EPC) in solution for the Langmuir, Fruendlich, Tempkin, and quadratic equations.

Soil	depth (cm)	Langmuir	Fruendlich	Tempkin	Quadratic
4	0-5	.9905	.8185	.8960	.8845
	15-20	.9500	.9067	.9728	.8409
5	0-5	.8017	.9074	.9527	.9990
	15-20	.4955	.9723	.9032	.9895

Table 39. Phosphorus adsorption maxima of study site soils calculated by substitution of the equilibrium phosphorus concentration (EPC) derived from quadratic equation into different equations.

			adsorption maxima $\mu g/g$				
Soil	Depth (cm)	EPC ($\mu g/ml$)*	Quadratic	Fruendlich	Tempkin	Langmuir	Langmuir +
4 (mineral)	0-5	29.3	84.6	49.5	47.1	45.2	51.0
	15-20	36.3	229.2	174.4	148.6	132.3	153.8
5 (organic)	0-5	238.8	1715.7	2887.2	380.1	437.2	294.1
	15-20	48.2	1726.1	1442.3	722.0	1093.9	2500.0

* EPC at adsorption maximum obtained from quadratic equation
+ Adsorption maximum obtained from slope of linear Langmuir equation

Table 40. Measured soil properties for the study site soils and the coefficient of correlation (R) between these properties and 1) the adsorption maxima derived by substitution into the Tempkin equation, and 2) the phosphorus sorption index (PSI).

Community type	Site	Depth (cm)	% OM	TP	HCl ex P	Tamm ex P	HCl ex Al ug/g	Tamm ex Al	HCl ex Fe	Tamm ex Fe
Titi swamp - holly phase	4	0-5	11.0	64.2	14.5	34.0	318.5	556.0	44.0	103.0
		15-20	5.8	37.1	2.0	28.0	685.0	925.0	26.0	66.5
Black gum swamp	5	0-5	86.1	180.0	33.5	85.0	516.5	2252.0	217.0	372.0
		15-20	41.5	224.9	36.0	143.0	4035.5	5700.0	46.5	138.0
Adsorption maxima substitution into Tempkin equation			.5197	.9266	.8130	.9825	.9838	.9852	.1715	.2851
Phosphorus sorption index 10 $\mu g/ml$ (quadratic equation)			.0553	.6774	.5192	.8454	.9389	.9469	.3146	.2004

extractable aluminum and iron concentrations were also higher than the 0.1 N HCl extractable aluminum and iron concentrations (Table 40). In addition, the dry weight of aluminum per gram of soil is much greater than the dry weight of phosphorus per gram of soil, and aluminum concentrations in the soil increase with depth (Table 40).

Linear regression of the adsorption maxima, phosphorus sorption index, and the measured soil properties indicate that the highest correlation obtained for both the adsorption maxima and the phosphorus sorption index was with Tamm oxalate extractable aluminum (R = .9852 and .9469 respectively) and 0.1 N HCl extractable aluminum (R = .9838 and .9389 respectively) (Table 40). There was also high correlation between the adsorption maxima and the phosphorus sorption index with Tamm oxalate extractable phosphorus (R = .9825 and .8454 respectively), but only the adsorption maxima had high correlation with total phosphorus (R = .9266) (Table 40). There was low correlation for both the adsorption maxima and the PSI with Tamm oxalate extractable iron (R = .2851 and .2004 respectively), 0.1 N HCl extractable iron (R = .1715 and .3146 respectively), and percent organic matter (R = .5197 and .0553 respectively) (Table 40).

The adsorption maxima are useful in describing the soil phosphorus retention capacity although at higher phosphorus concentrations these values may be exceeded by the actual adsorption (Yuan and Lucas 1982). This parameter is indicative of the phosphorus retention potential but may overestimate the actual field adsorption maximum because channelized water movement will reduce contact with a large portion of the soil matrix (Richardson 1985).

The adsorption maxima for the site four mineral soils are generally low (less than 230 μg/g soil) suggesting a limited capacity for phosphorus adsorption. The site five organic soils have higher adsorption maxima (294-2888 μg/g soil) suggesting a greater capacity for phosphorus adsorption. Yuan and Lucas (1982) reported adsorption maxima calculated in the same manner from 82 to 1148 μg/g for 30 Florida sandy mineral soils including a Humaquept. Krottje *et al.* (1982) reported adsorption maxima in a range from 62 to 775 μg/g for four Florida soils with less than 19% organic matter. They also reported an adsorption maximum of 2000 μg/g for a Florida soil with 67.2% organic matter. The adsorption maxima of these five soils were calculated as the slope of the Langmuir equation. Phosphorus adsorption was highly correlated with organic matter and exchangeable aluminum content (Krottje *et al.* 1982).

The 0.1 N HCl extractable phosphorus may be considered surface active phosphorus or the labile soil phosphorus (Bache and Williams 1971). The Tamm oxalate extract is a stronger reagent, disturbing more than just the surface active (labile) phosphorus. Therefore Tamm oxalate extractable phosphorus concentrations were higher than the 0.1 N HCl extractable phosphorus concentrations but less than the total phosphorus concentrations which include more tightly bound forms of phosphorus.

The adsorption maxima and a phosphorus sorption index were correlated with measured soil properties in order to indicate which soil factors are best related to phosphorus adsorption in the soil.

The relatively low phosphorus adsorption capacity of the site four mineral soils and the much higher phosphorus adsorption capacity of the site five organic soils are related to the content and availability of aluminum in these soils rather than the amount of organic matter present. The dry weight of aluminum per gram of soil is much greater than the dry weight of phosphorus per gram of soil, and aluminum concentrations in the soil as well as the phosphorus adsorption maxima increased with depth. The high correlation of extractable aluminum with both the adsorption maxima and the phosphorus adsorption maxima as well as the increase in the adsorption maxima and aluminum concentrations with depth indicate the importance of aluminum in phosphorus adsorption in the study site soils. The high correlation of the adsorption maxima and the phosphorus sorption index with Tamm oxalate extractable aluminum indicates that phosphorus adsorption is related to the amorphous and poorly crystalline oxides of aluminum.

Pre-treatment to remove phosphorus from wastewater entering flow through wetlands in particular will be necessary unless site-specific information is available to indicate a capacity for its retention in the system. Determination of the phosphorus adsorption maxima for soils at this titi shrub swamp indicate the potential for phosphorus retention in the soil. The site five organic soils have a high capacity for phosphorus adsorption. The capacity for phosphorus adsorption is much lower in the site four mineral soils. The finding by Fox and Kamprath (1970) that soils which have low capacity to adsorb phosphorus require very high concentrations of phosphorus in solution to compensate for a lack of total available phosphorus suggests that although a soil may have a low phosphorus adsorption capacity it may indirectly contribute to a net uptake of phosphorus in the system through plant immobilization. Therefore, whether through adsorption or immobilization this study site appears to have potential for phosphorus treatment of added wastewater.

HYDROLOGY

Purpose

The purpose of this task was to estimate the water budget in the wetland both prior to and after wastewater discharge. The general hydrologic equation for determining the water budget in a wetland is Inflow = Outflow $\pm \Delta S$ (change in storage). The components of a wetland water budget are further described by Carter *et al.* (1979) as $P + SWI + GWI = ET + SWO(R) + GWO + \Delta S$, where P is precipitation; SWI is surface water inflow (including overland runoff); GWI is groundwater inflows; ET is evapotranspiration; SWO is surface water outflow (runoff); GWO is groundwater outflow from the basin; and ΔS is the change in storage.

Several assumptions were made to simplify the estimation of the water budget. The basin storage is assumed to be constant over a long period of time and, therefore, the change in storage is neglected. The water budget estimation is made on a depth rather than on a volume basis; therefore, overland runoff and groundwater flow are assumed to be outflow components and are eliminated from the equation. The simplified water budget equation becomes $P = R + GWO + ET$.

Despite the recognized importance of the hydrologic regime to the structure and function of wetlands, it is often the component of wetland ecosystem research which is not thoroughly investigated (LaBaugh 1986). In order to properly assess impacts to a wetland, a complete water budget must be prepared. Only when the water budget is combined with measurements of nutrient concentrations can a complete interpretation of wetland treatment system performance be made (Hammer and Kadlec 1983). A water budget for the wetland study site was determined for use in the simulation model of the titi shrub swamp receiving wastewater.

Literature Review

Evaporation is the conversion of water from the liquid state into vapor, and its diffusion into the atmosphere. Transpiration is the return of water to the atmosphere by plants. Evapotranspiration then is the evaporation from all moist surfaces to the atmosphere. Evapotranspiration includes several processes that are separately difficult to quantify; therefore, potential evapotranspiration is usually estimated. Potential evapotranspiration is defined as the evaporative flux that will not exceed the available energy from both radiant and convective sources (Saxton and McGuinness 1982). In determining potential evapotranspiration, atmospheric variables are considered separately from plant and soil effects. Often water is not freely available and actual evapotranspiration is less than potential evapotranspiration. Therefore, potential evapotranspiration is estimated first, based on meteorological factors, and the amount of that potential used by the actual evapotranspiration processes is then estimated.

Watersheds dominated by flat slopes and long-term seasonal precipitation and flooding are depressional watersheds (Bedient 1975). These watersheds are dominated by lateral rather than vertical soil water movement and the lateral movement is difficult to measure due to poorly defined drainage paths. The titi shrub swamp is in a depressional watershed and its analysis requires an emphasis on soil storages and evapotranspiration changes over long periods of time as well as some quantification of lateral water movement. The water balance technique of Thornthwaite and Mather (1957) for determining potential evapotranspiration is ideal for analyzing depressional watersheds (Bedient 1975).

A water budget for the Okefenokee Swamp was developed by Rykiel (1977). Evapotranspiration was estimated as a residual term. An independent estimate of potential evapotranspiration was made with the Thornthwaite method for comparison with the residual estimate. Potential evapotranspiration was found to underestimate evapotranspiration and therefore should be used as a minimum value with normal rainfall (Rykiel 1977).

Estimates of potential evapotranspiration were compared to field measurements (groundwater level fluctuation) of evapotranspiration in a cypress strand (Carter *et al.* 1973). Evapotranspiration measured in this manner was higher than estimated potential evapotranspiration except when the groundwater level was well below the land surface and water was unavailable to plants. Evapotranspiration values measured in the same manner in these cypress strands were reported by Burns (1978). When the groundwater level was high, field evapotranspiration approached pan evaporation. These studies suggest that estimates of potential evapotranspiration may underestimate evapotranspiration when water availability is high.

In order for evapotranspiration to occur a source of energy and a vapor pressure gradient between the evaporating surface and the atmosphere must exist. Solar energy is the main source of energy and advection of energy from outside an area may increase evapotranspiration (oasis effect). Evapotranspiration is influenced by a number of factors including solar radiation, air temperature, vapor pressure gradient, wind and air turbulence. In addition to the meteorological factors the nature of the evapotranspiring surface and availability of water are important. For example, the height and roughness of vegetation influence air turbulence and transpiration can at times exceed open water evaporation (Linacre 1976).

On the other hand, the sheltering effect and high albedo of vegetation as well as the resistance to water movement in dry periods could decrease the rate of water loss during dry periods (Linacre 1976). The presence of vegetation in a wooded swamp in southern Ontario decreased evapotranspiration (Munro 1979). Swamp vegetation was efficient in converting net radiation into turbulent energy exchange, thus minimizing water loss. It appears as though wetlands may evapotranspire at a low rate when water is limiting and at a higher rate when water is readily available.

Evapotranspiration in three cypress swamps in Withlacoochee State Forest was measured by Ewel (1985) by determining changes in water levels. Daytime reductions in water level, due to evapotranspiration and infiltration, could be distinguished from nighttime reductions in water level, due to infiltration only. Evapotranspiration rates were calculated as the difference between the daytime and nighttime water level changes converted to a volume basis. Average annual evapotranspiration was estimated to be 31 in. during the 3 yr for which data were available. Average annual precipitation for the 3-yr period was 59 in. Therefore, evapotranspiration was 52% of precipitation at these sites. Evapotranspiration in slash pine flatwoods in north Florida was estimated over the same 3-yr period to be 41 in./yr, or 74% of precipitation. The evapotranspiration rate estimated by Ewel (1985) was 77% of this rate. This comparison confirmed earlier reports of low evapotranspiration rates for certain cypress swamps.

A decrease in the rate of water loss would be a water conservation mechanism and any discussion of water conservation by wetland vegetation should include reference to xeromorphy. Xeromorphy in plants may be a water conservation mechanism. Plants of acid habitats are often structurally adapted to conserve water, as are plants from xeric habitats (Clewell 1981). Such plants in acid habitats are called physiological xerophytes (Clewell 1981). Xeromorphic characteristics in plants include thick cuticles, deeply sunken stomata, and highly reflective surfaces. These are the characteristics of evergreen sclerophyllous leaves such as those of titi and sweetbay. These characteristics have evolved in desert plants in response to drought but some xeromorphic species have a "bimodal" distribution, i.e., they are found in both wet and dry habitats but not in intermediate habitats (Larsen 1982). A species could undergo selection for characteristics that adapt it more effectively to both wet and dry habitats than for the habitats between these extremes. In the process of evolving characteristics permitting survival in wet areas, the plants could have acquired characteristics fitting them for survival in dry areas.

These characteristics may be in response to low fertility and potential water deficiency, but water loss is the key factor (Brunig 1971). On the other hand, xeromorphy in plants may be an adaptation to low fertility and water conservation features may be fortuitous (Larsen 1982). If xeromorphy is an adaptation to dry conditions the reduction of transpiration losses could be a necessary adaptation for survival during dry periods. Low transpiration rates for cypress domes may likewise be an adaptation for survival when water becomes limiting during dry periods (Brown 1981).

Evapotranspiration can be determined by various direct measures such as the measurement of the increase in water vapor in air flowing through gas exchange chambers (Odum *et al.* 1970; Odum and Jordan 1970; Cowles 1975; Brown 1978; Burns 1978). The metabolism and transpiration of some plants in a tropical rain forest were measured by Odum *et al.* (1970), and the effects of air velocity on leaf metabolism was evaluated. Air velocity in low ranges limited metabolism of living forest components. Graphs of metabolism and air velocity had the form of limiting-factor hyperbolae. In addition, transpiration increased asymptotically with airflow over leaf surfaces (Odum *et al.* 1970). Therefore, flow rates in chambers should not minimize metabolism or enhance transpiration. Air flow rates were adjusted by Brown (1978) so as to not limit metabolism or enhance transpiration, and to insure that the maximum difference in temperature between the ambient and exhaust air never exceeded 3°C.

Methods

Precipitation data from a NOAA weather station approximately 2 km east of the study site was collected for the pre- and post-effluent sampling periods. A United States Geological Survey stream water level recorder was installed at the point of discharge of the wetland into Whortleberry Creek during the 1985-86 sampling period.

Hydrology is the primary determinant of wetland ecosystems and the most important factor influencing wetland biogeochemistry (Gosselink and Turner 1978). A water budget defines the balance between inflows and outflows of water within system boundaries for a certain period of time. The general hydrologic equation for determining the water budget in a wetland is

$$\text{Inflow} = \text{Outflow} \pm \Delta S \text{ (change in storage).}$$

The specific components of a wetland water budget have been further described by Carter *et al.* (1979) as:

$$P + SWI + GWI = ET + SWO + GWO + \Delta S$$

where P is precipitation, SWI is surface water inflow (including overland runoff), GWI is groundwater inflow, ET is evapotranspiration, SWO is surface water outflow, GWO is groundwater outflow (discharge through aquifers, seepage), and ΔS is the change in storage.

Determination of individual water budget components may not be a simple matter (Carter *et al.* 1979). Several assumptions were made to simplify estimation of the water budget due to the difficulty in determining water budget components. The basin storage was assumed to be constant over the period of time for which the budget was calculated; therefore, the change in storage (ΔS) was assumed to be zero. There are no tributaries providing surface water inflow to the site. The linear nature of shrub swamps in the panhandle precludes any significant watershed interception of precipitation beyond that falling directly on the system (Wharton *et al.* 1982). Therefore, overland runoff is assumed to be zero. SWI is thus eliminated from the equation. The water budget was estimated on a depth basis rather than on a volume basis; therefore, wetland area is not taken into account. Overland runoff and groundwater flow are assumed to be outflow components. GWI is thus eliminated from the equation. The simplified water budget equation for this study site is $P = R + G + ET$, where R is runoff and G is groundwater flow. An annual water budget for the study site was calculated using data for the 5-yr period from 1982 through 1986 and for October 1985 through September 1986 when transpiration measurements were made, hereinafter referred to as the water budget year. All water data are reported in English System Units (in.) as is common in the hydrology field.

Precipitation and Runoff

Daily precipitation records are kept at the Apalachicola weather station approximately 1 km east of the study site (NOAA 1982-1986). Runoff was calculated from daily precipitation using the SCS curve number method presented in section 4 of the National Engineering Handbook (SCS 1972). Chow (1973) described the method that uses the following equation:

$$Q = [(P - 0.2\ S)^2]/(P + 0.8\ S),$$

where Q is the runoff in inches, P is the storm precipitation in inches, and S is the potential infiltration in inches which is determined as follows:

$$S = (1000\ /\ Cn) - 10$$

where Cn is the curve number previously determined by the SCS (1972) for hydrologic soil-cover complexes that are a combination of soil type and cover. The curve number can be determined for antecedent moisture condition (AMC) classes based on total antecedent precipitation. Konyha *et al.* (1982) described five modifications for determining potential infiltration in accordance with the SCS curve number method in order to predict runoff in flat high water table watersheds in Florida. Two of the methods (AMC II and AMC III) were used to estimate runoff from precipitation at the study site for 1982 through 1986. In addition these two methods were used to estimate runoff from precipitation at the study site during the water budget year.

Groundwater

Water levels in shallow groundwater wells (see map, Figure 2) were measured monthly for 1 yr to construct maps of the potentiometric surface of the study site for high and low water periods. Water depth at stations 2, 3 and 5 (see map, Figure 2) within the wetland were concurrently measured.

Groundwater flow is composed of two components at the study site infiltration through a semi-impermeable organic layer and surface sands, and deep seepage through clayey sands. Infiltration through the semi-impermeable organic layer and more permeable surface sands is highly variable and no data were collected to estimate this flow. Therefore, an estimate was made of the maximum groundwater flow that could occur from the upper surface sands to the lower clayey sands. The upper limit of this deep seepage flow was estimated using a simplification of Darcy's law:

$$v = K\ \Delta h/\Delta z$$

where v is the velocity of the water passing from the surface sand zone to the clayey sand zone, K is the hydraulic conductivity of the clayey sand zone, and $\Delta h/\Delta z$ is the hydraulic gradient between the two zones. Δh is the change in piezometric head between the two zones and Δz is the thickness of the clayey sand zone.

Evapotranspiration

Evapotranspiration was determined with an empirical formula relating climatic variables causing evapotranspiration (Thornthwaite and Mather 1957). The Thornthwaite method uses mean monthly air temperatures to determine an annual heat index. Mean monthly air temperatures for the Apalachicola weather station were reported by NOAA (1982-1986). Unadjusted monthly potential evapotranspiration is determined from the mean monthly air temperature based on the annual heat index. Adjusted monthly potential evapotranspiration is determined by multiplying the unadjusted values by the monthly duration of sunlight (12 hr basis) at the station's latitude.

When precipitation was greater than potential evapotranspiration, actual evapotranspiration was taken to be equal to potential evapotranspiration. In months when precipitation was less than potential evapotranspiration, water was lost from the soil. The actual water loss varies with the amount of moisture in the soil. This monthly soil water loss was determined as the difference between the maximum soil moisture storage and the monthly soil moisture retained for the accumulated monthly water loss. The maximum soil moisture was determined using the following formula:

$$\text{maximum soil moisture} = (1000\ /\ Cn) - 10$$

where Cn is the curve number previously determined by the SCS (1972) for hydrologic soil-cover complexes that are a combination of soil type and cover. The monthly soil moisture retained for the accumulated monthly water loss was determined using soil moisture depletion curves presented by Bedient (1975). The soil moisture depletion curves indicate the relationship between the accumulated water loss and soil moisture retained for different maximum soil moisture storages. The monthly soil water loss was added to the monthly precipitation to obtain an estimate of monthly actual evapotranspiration for those months when potential evapotranspiration was greater than precipitation. Pan evaporation can also be used to estimate potential evapotranspiration. Pan evaporation values from U.S. Weather Bureau Class A Land Pans are measured at selected NOAA weather stations. The closest NOAA weather station measuring pan evaporation is in Milton, Florida, 160 km northwest of the study site. Pan evaporation for the Milton weather station were reported by NOAA (1982-1986).

Brown (1978) measured surface evaporation for soil and water surfaces as well as transpiration from plants in several wetlands in Florida. The surface evaporation values measured by Brown (1978) were used in this study to estimate the portion of evapotranspired water loss due to surface evaporation. The direct measurement of transpiration in the study site was determined using the gas exchange chamber method.

Transpiration

In five studies where chambers were used to measure metabolism and transpiration, airflows were selected so as to not limit metabolism or enhance transpiration. The volume and the number of turnovers per minute of six chambers of wide ranging size used in these five studies are given in Table 41. The computer program CURFIT (Spain 1982) was used to fit the data on chamber volume and turnover time. The best fitting model equation for these data of chamber volume and turnover time was a power function ($y = Ax^n$) where y = number of turnovers per minute, and x = chamber volume. This model equation was used to determine the number of turnovers per minute and thus the airflow in the chambers used in this study that would not limit metabolism or enhance transpiration.

Three chambers were used in this study. The dimension, volume, turnover time (calculated with the model equation), and the airflow that would not limit metabolism or enhance transpiration are given in Table 42. All three chambers' volumes were within the range of the chamber volumes used to develop the model equation. In the field, airflow in the chamber was set at the level calculated to not limit metabolism or enhance transpiration. The airflow was increased when the temperature inside the chamber increased above the ambient temperature outside the chamber.

The flow-through cylindrical chambers were constructed with wooden hoops and polyethylene plastic. Flow through the chamber was provided by a fan with variable speed adjustment mounted at one end. The other end was left open. A canopy branch was inserted into the chamber. Transpiration rates were

Table 41. Chamber volume and number of turnovers per minute in four studies where metabolism and transpiration were measured.

Chamber Volume (m^3)	# of Turnovers (per minute)	Source
0.0002	73.0	Odum et al. 1970
0.0004	30.0	Brown 1978
0.052	8.0	Brown 1978
0.898	5.0	Burns 1978
8.0	1.35	Cowles 1975
4000.0	0.19	Odum and Jordan 1970

Table 42. The dimension, volume, turnover time (calculated with model equation) and the airflow that would not limit metabolism or enhance transpiration, for the three chambers used in this study.

Chamber	I	II	III
diameter (m)	0.58	0.58	0.58
radius (m)	0.29	0.29	0.29
length (m)	2.00	1.00	1.40
volume (m^3)	0.528	0.264	0.370
# turnovers (per min)*	3.97	5.01	4.47
minimum airflow (m/sec)	1.75	1.10	1.38

*calculated with model equation $y = Ax^n$

$A = 326.5$

$n = -.3344$

determined by monitoring water vapor changes, with a dew point hygrometer, of the air passing through the chamber. Every attempt was made to maintain similar conditions inside and outside the chamber.

Transpiration of sweetbay and black titi was measured in a bay swamp from a 9 m tower constructed with collapsible scaffolding. The chambers were suspended from the tower with an adjustable boom and pulley system. In situ measurements were recorded every 15 min for a time period from before transpiration began in the morning to after transpiration ended in the evening. This time period represented one run. The temperature inside the chamber and the ambient temperature outside the chamber were measured with mercury thermometers. An electric pump pulled either an intake or exhaust air sample from the chamber through the dew point hygrometer. The intake dew point temperature and the exhaust dew point temperature were measured with a EG and G Model 880 dew point hygrometer. The flow through the chamber was measured with a battery operated Weather Measure Model W141A hot wire anemometer and the solar input was measured with a Matrix Mark VI solar radiometer. Electricity to operate the fan, pump and dew point hygrometer was supplied by a portable generator located at the base of the tower. At the end of each run the branch inside the chamber was harvested and the leaf biomass was determined. These measurements were the basis for the following calculations performed with a computer spread sheet program.

The ambient temperature (TAM) and the intake and exhaust dew point temperatures (DTIN and DTEX) were converted to saturation vapor pressure (ESTAM, ESDTIN, ESDTEX) with the Clausius-Clapeyron relationship, as follows:

$$\text{saturation vapor pressure (mb)} = 6.841\ \text{EXP}(0.0608) \times \text{T}\ (^{\circ}\text{C}).$$

The saturation vapor pressure (ESTAM, ESDTIN, ESDTEX) was converted to absolute humidity (AHTAM, AHIN, AHEX) with a manipulated form of the gas law, as follows:

$$\text{absolute humidity (g/m}^3\text{)} =$$

$$\text{saturation vapor pressure} \times \text{MW} \times (10^3\ \text{erg}\cdot\text{cm}^{-3}\cdot\text{mb}^{-1}\cdot 10^6\ \text{cm}^3\cdot\text{m}^{-3})/\text{R} \times \text{TK}$$

where MW = molecular weight of water (18 g/mole)

R = gas constant (8.31×10^7 erg/°K · mole), and

TK = ambient temperature in °K.

Relative humidity was calculated as follows:

$$\text{relative humidity} = \text{AHTAM/AHIN} \times 100.$$

The saturation deficit was calculated as follows:

$$\text{saturation deficit} = (\text{ESTAM})\ (1 - \text{relative humidity}).$$

The flow rate (FR) was calculated as the product of the cross-sectional area of the fan duct and the measured flow.

The rate water was released or the transpiration rate was calculated as follows:

$$\text{transpiration rate in g } H_2O/\text{hr} = (\text{FR})\ (\text{AHEX} - \text{AHIN}).$$

These hourly transpiration rates were integrated with a computer program using trapezoidal integration to obtain the total daily transpiration rate. The leaf area was determined as the product of the leaf biomass measured at the end of the run and the leaf biomass to area ratio for the vertical interval where transpiration was measured (9 to 12 m). The leaf biomass to area ratio for the vertical interval from 9 to 12 m was determined in the tree dimension analysis for each species. The transpiration rate per leaf area was calculated as the dividend of the transpiration rate and the leaf area. The total daily transpiration rate per leaf area was calculated as the dividend of the total daily transpiration rate and the leaf area. The transpiration rate per biomass was calculated as the dividend of the transpiration rate per leaf area and the leaf biomass to area for the vertical interval where transpiration was measured (9 to 12 m).

In order to extrapolate the transpiration measurements to the ecosystem level the total daily transpiration rate per ground area was determined. The total daily transpiration rate per ground area was calculated as the product of total daily transpiration rate per leaf area and the leaf area index.

In order to calculate the leaf area index the vertical distribution of leaf biomass was determined with the plumb-bob method similar to the method used by Benedict (1975) and Brown (1978). A marked line was lowered through the vegetation from the tower where transpiration was measured, with a three part extension pole. The number of leaves hitting the line and their species and location along the line were recorded. This was performed at 16 compass points and at three pole extension distances for a total of 48 samples. The number of leaves of each species hitting the line at a given vertical interval (9 to 12 m or 3 to 9 m) in terms of the percent of 48 samples was multiplied by the leaf biomass to area ratio for a given vertical interval for that species. The percent of the total leaf biomass for each species at each given vertical interval was the vertical distribution of leaf biomass for that species. The leaf area index for each species was calculated as the ratio of the estimated leaf biomass per ground area for that species and the leaf biomass to area ratio for a given vertical interval for that species, multiplied by the vertical distribution of leaf biomass for that species. The estimated leaf biomass per ground area was determined with dimension analysis of 10 trees of each species applied to trees sampled in community analysis quadrats.

Results and Discussion

The climate in the locality of the Apalachicola weather station is typical of that experienced on the northern Gulf of Mexico (NOAA 1982-1986). Because of the moderating effect of the surrounding Gulf, temperatures are usually mild and subtropical in nature. Average annual precipitation is about 57 inches, but actual monthly and yearly totals vary widely. Thunderstorms occur in all months and about three-fourths of the average annual number occur during the summer months. The average annual precipitation for the 5-yr period from 1982 through 1986 was 66 in. (Table 43). This was almost 10 in. greater than the preceding 74-yr average of 56 in. for this weather station (Kennedy 1982). Precipitation for the water budget year was similar (65 in.) to values for the 5-yr period from 1982 through 1986.

Precipitation data from a NOAA weather station near the experimental wetland was plotted for the pre- and post-effluent sampling periods (Figure 26). Peaks represent weather events such as tropical storms and hurricanes. Since the experimental wetland is located less than 2 mi from the Gulf of Mexico, severe weather events were common. During the 1985-86 study period, two major hurricanes occurred (September and October 1985).

Surface water flows in a generally east-west direction in the wetland, culminating in a small stream, Whortleberry Creek (see map, Figure 1). This stream actually flows north until discharging into the Apalachicola River, which flows into the Gulf of Mexico. Treatment plant discharge represented the other major source of water entering the wetland during the 1985-86 sampling period (Figure 26). A comparison of Figures 26a and 26b reveals a contribution to wetland discharge by both precipitation and treatment plant discharge. Major precipitation events occurred from July to September during both sampling periods (Figure 26). Treatment plant discharge began in May 1985 and gradually increased from 500 million gal/day to slightly in excess of 1 million gal/day in November 1985 (Figure 26). Stream discharge did not always correlate with peak precipitation, for example, during February 1986, a recorded stream discharge of almost 4 million gal/day was not the result of increased precipitation. This indicates a contribution to stream discharge of water from other sources, possibly a neighboring wetland or other surface runoff or discharge from groundwater. Best *et al.* (1983) give a more complete discussion of hydrologic patterns at this research site.

Water depths at the sample sites varied seasonally during the 1985-86 sampling period. Depths were recorded at sites of benthic macroinvertebrate collection only. At sites D25, D200, and D400, levels varied from 5 to 40 cm. Water levels at benthic macroinvertebrate collection locations at site D900

Table 43. Average annual precipitation (P), estimated runoff (R), pan evaporation (PE), potential evapotranspiration (PET), actual evapotranspiration (AET) and water budget residual (RES) for the study site from 1982 through 1986 and for the water budget year (WBY).

Year		P (in)	R (in)	PE (in)	PET (in)	AET (in)	R+AET (in)	R+PET (in)	RES I (in)	RES II (in)
1982		71.96	23.60	56.91	40.83	38.67	62.27	64.43	9.69	7.53
1983		64.38	16.49	57.84	40.05	34.42	50.91	56.64	13.47	7.74
1984		56.50	16.82	62.14	39.84	32.90	49.72	56.66	6.78	-0.16
1985		68.57	20.92	54.91	41.27	39.05	59.97	62.19	8.60	6.38
1986		66.81	21.35	57.93	43.42	37.35	58.70	64.77	8.11	2.04
Average	(in)	65.64	19.84	57.95	41.08	36.48	56.31	60.94	0.33	4.72
	(m)	1.67	0.50	1.47	1.04	0.93	1.43	1.54	0.24	0.13
	(%)	100.0	30.2	88.3	62.6	55.6	85.8	92.8	14.2	7.2
WBY	(in)	64.71	21.16	58.00	43.20	37.14	58.30	64.36	6.41	0.36
	(%)	100.0	32.7	89.6	66.8	57.4	90.1	99.5	9.9	0.5

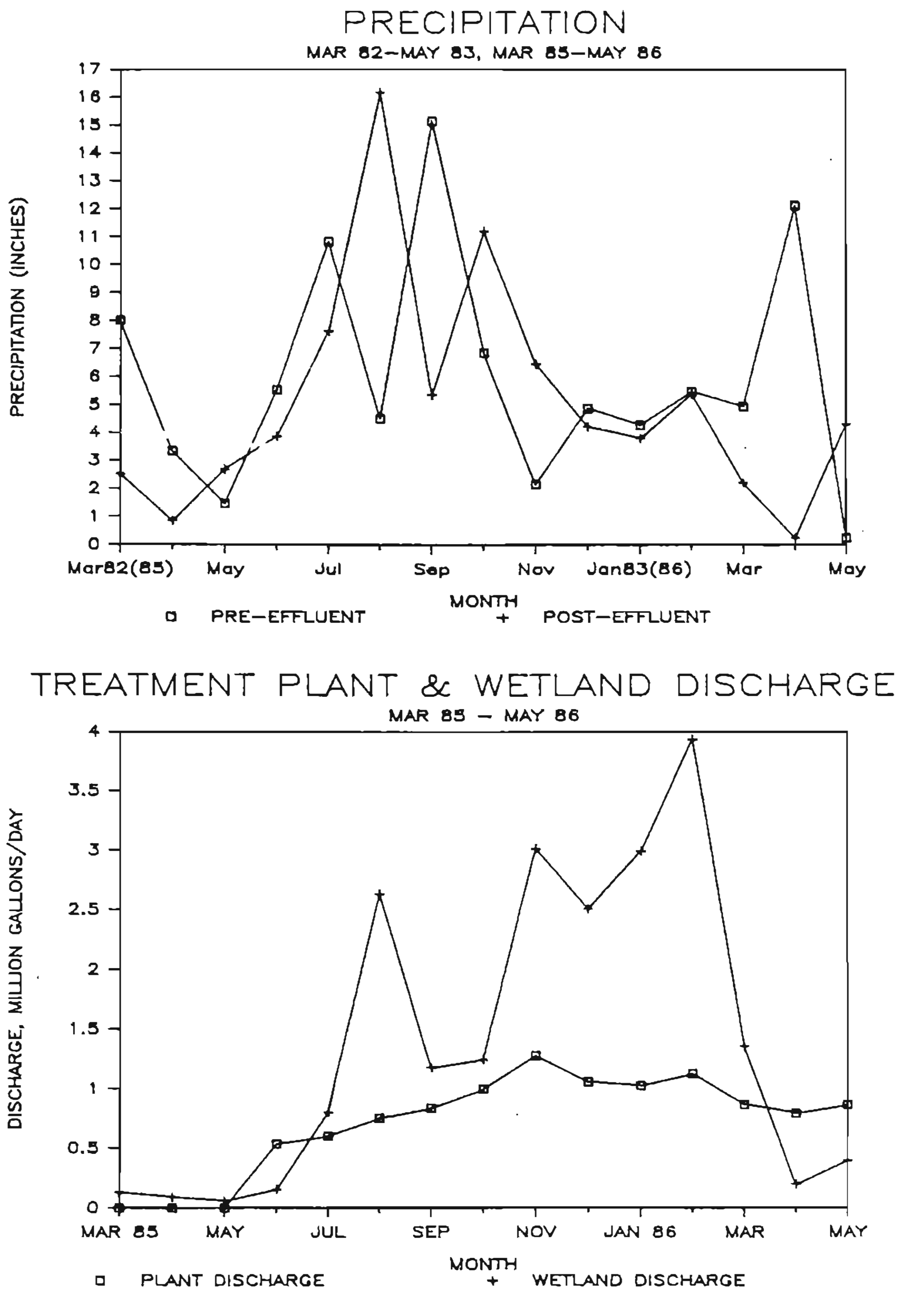


Figure 26. Hydrologic inputs at the Apalachicola wetland. (a) Precipitation, (b) Treatment plant and wetland discharge.

varied from 23 to 50 cm. The water level in the swamp is a function of precipitation and, close to the effluent outfall, of added wastewater, with periodic drying a natural process of the system.

Runoff

Prior to wastewater addition, runoff was estimated from daily precipitation using the SCS curve number method described in Section 4 of the *National Engineering Handbook* (USDA/SCS 1972). Konyha *et al.* (1982) evaluated five methods of predicting runoff in flat, high-water–table watersheds. All five methods use the SCS runoff equation and differ from one another in how they determine potential infiltration. Two of these methods were applied to estimate runoff. The first method, AMC II (AMC standing for antecedent moisture class) assumes watershed conditions during periods of average precipitation and tends to underpredict large runoff events. The second method, AMC III, assumes wet watershed conditions and predicts acceptably for large runoff events, overpredicting, however, most small runoff events.

Soils at the study site are in the Rutlege and Pamlico Series, which are Group D type soils in the SCS (1972) hydrologic soil group classification system. Soils in this group are characterized by high runoff potential, very slow water transmission rates and a high permanent water table. The curve numbers (Cn) for these soils assuming average (AMC II) and wet (AMC III) watershed conditions were 77 and 89.5 respectively. Assuming average (AMC II) watershed conditions underpredicts large runoff events and assuming wet (AMC III) watershed conditions overpredicts small runoff events (Konyha *et al.* 1982). The estimate of runoff assuming wet (AMC III) watershed conditions was at least 45% greater than the estimate of runoff assuming average (AMC II) watershed conditions. The average of the two methods was therefore selected. The average annual estimated runoff for the 5-yr period from 1982 through 1986 was 20 in. (Table 43). This was 30% of the average annual precipitation at the study site for this period. Estimated runoff for the water budget year was similar (21 in.) to values for the 5-yr period from 1982 through 1986.

Estimates of runoff using 5 day antecedent precipitation to determine antecedent moisture class (AMC) are not recommended for use in Florida (Konyha *et al.* 1982). The method predicts erratically, sometimes predicting with reasonable accuracy and sometimes under predicting runoff by over 3 in. The method which assumes average (AMC II) watershed conditions also underpredicts large runoff events. The underpredictions occur because for these events the water table is high and the average watershed assumption is inappropriate. When wet (AMC III) watershed conditions are assumed the predictions are acceptable for the large runoff events, but most small runoff events are overpredicted. It may have been best to assume average (AMC II) watershed conditions for small runoff events and wet (AMC III) watershed conditions for large runoff events, but this would have required selecting a cutoff point for runoff events in terms of precipitation, for which there is no basis.

Groundwater Outflow

Groundwater outflow was estimated using Darcy's law, a potentiometric map of the wetland, and natural gamma well logs describing the thickness of the surficial aquifer (Best *et al.* 1983). An average value of 37.0 in./yr was used in the water budget estimation.

The potentiometric surface for high and low groundwater periods are presented in Figures 27 and 28 respectively. The difference between potentiometric contours of the wetland is 4 ft for both the high and low periods (16' to 12' and 14' to 10' respectively). During low water flow was in the southwest direction, shifting towards the northwest during high water flow. The average water depth and fluctuation for all three stations within the wetland were 0.51 m and 0.93 m respectively (Table 44).

The natural gamma log for a well located 2 km northeast of the study site indicates that there is a zone of surface sands extending to a depth of 42 ft overlying a zone of clayey sands extending to a depth of 178 ft (Figure 29).

Hydraulic conductivities for similar zonation (a semi-impermeable organic layer and surface sands underlain by clayey sands) beneath cypress domes in Alachua County ranged from 0.01 to 0.1 ft/wk for the semi-impermeable organic layer, from 1 to 10 ft/wk for the surface sands and from 0.001 to 0.01 ft/wk for the clayey sands (Cutright 1974). In estimating the upper limit of deep seepage through the clayey sand zone, the high end of the hydraulic conductivity range measured by Cutright (1974) for clayey sands was used ($K = 0.01$ ft/wk). The change in the piezometric head Δh between the two zones is 178 ft and the thickness of the clayey sand zone Δz is 136 ft (178-42). Therefore, the maximum velocity ($v = K \Delta h / \Delta z$) of the water passing from the surface sand zone to the clayey sand zone is 0.013 ft/wk; at most 0.676 ft or 8 in. pass through the clayey sand zone per year.

Evapotranspiration

Evapotranspiration was estimated as the balance of the equation, and, therefore, includes all of the error associated with the estimation of the other components and the simplifying assumptions.

Estimates of surface water runoff obtained using the two runoff prediction methods were compared with actual discharge measurements determined from stage records and a rating curve (Table 45). The method more precisely predicted runoff as measured by the USGS stage recorder. Stage records were provided by the USGS from a water level recorder and flow meter which were installed at the outflow of the wetland. Estimates of evapotranspiration were compared with field measurements of transpiration during the study period (Table 46).

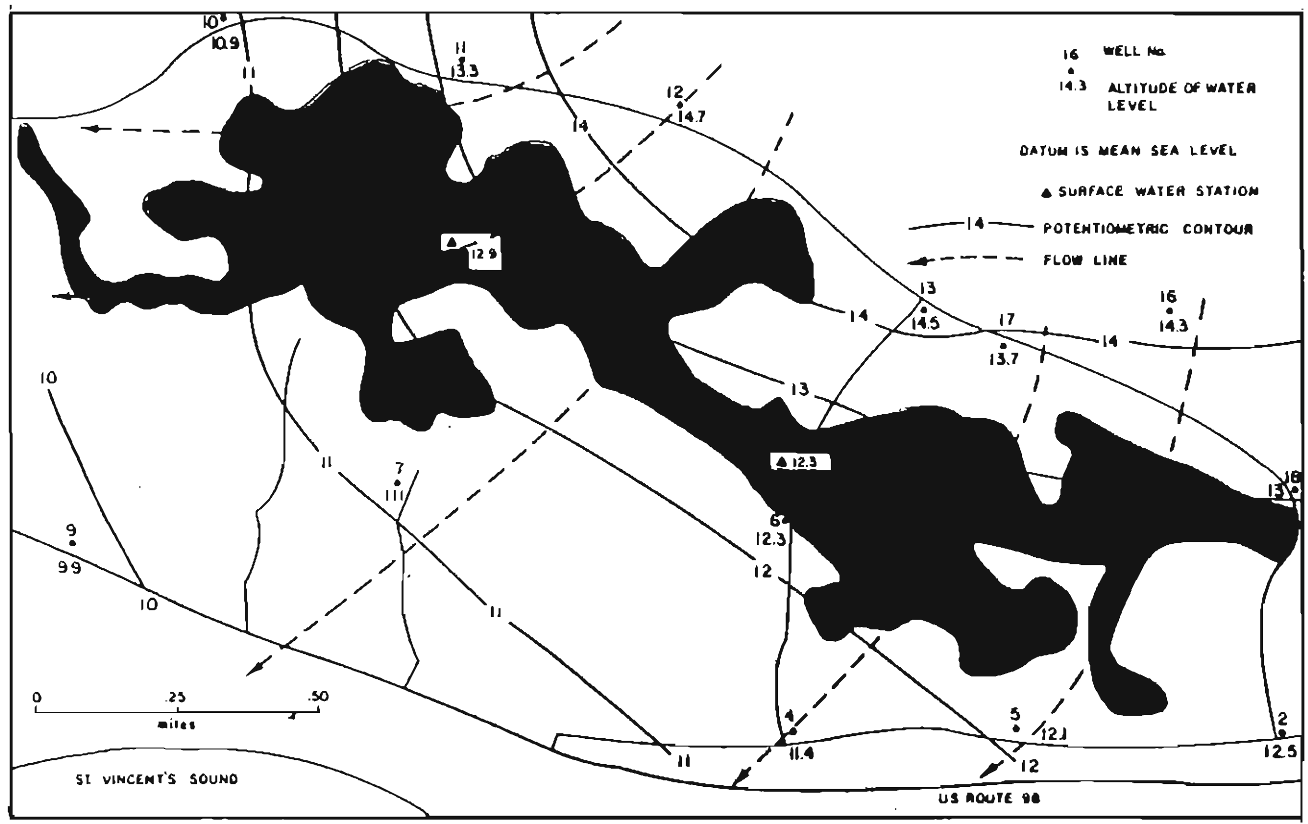


Figure 27. Potentiometric surface of the surfical aquifer at the titi shrub swamp study site in Apalachicola, Florida; 30 July 1982 (high groundwater).

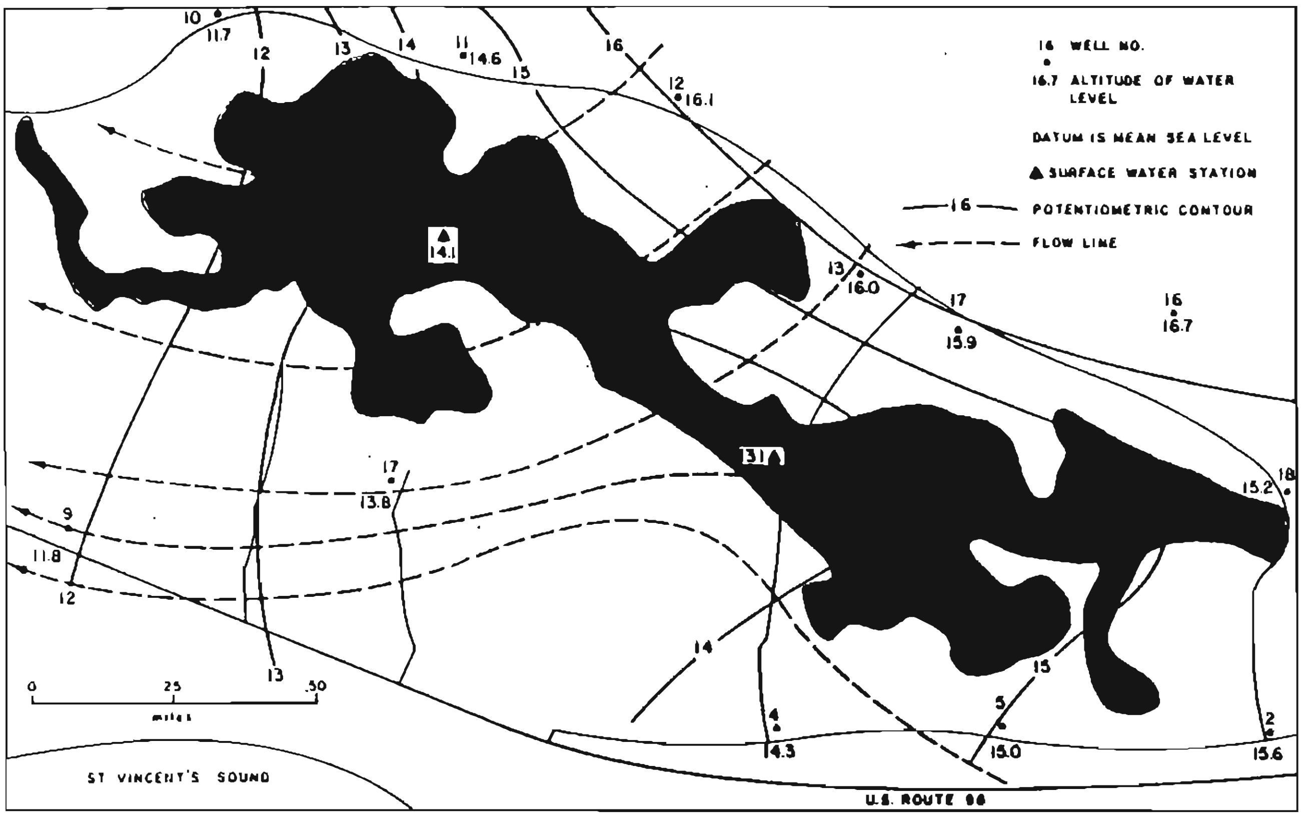


Figure 28. Potentiometric surface of the surfical aquifer at the titi shrub swamp study site in Apalachicola, Florida; 30 May 1982 (low groundwater).

Table 44. Water depth (m) for three stations with the titi shrub swamp in Apalachicola, Florida.

	Depth (m)		
Date	Station 2	Station 3	Station 5
Mar 30, 1982	0.62	0.43	0.73
Apr 29, 1982	0.54	0.35	0.62
May 30, 1982	0.15	0.03	0.39
Jun 29, 1982	0.20	0.23	0.66
Jul 29, 1982	0.95	0.76	1.53
Aug 29, 1982	0.71	0.55	0.80
Sep 29, 1982	0.62	0.38	0.69
Oct 29, 1982	0.60	0.38	0.31
Nov 28, 1982	0.43	0.08	0.26
Dec 30, 1982	0.51	0.32	0.62
Jan 30, 1983	0.47	0.41	0.54
Feb 28, 1983	0.55	0.46	0.67
Average	0.53	0.36	0.65
Range	0.15-0.95	0.03-0.76	0.26-1.53
Fluctuation	0.80	0.73	1.27

Average water depth for all sites = 0.51 meters
Average fluctuation for all sites = 0.93 meters
Average high water for all sites = 1.08 meters

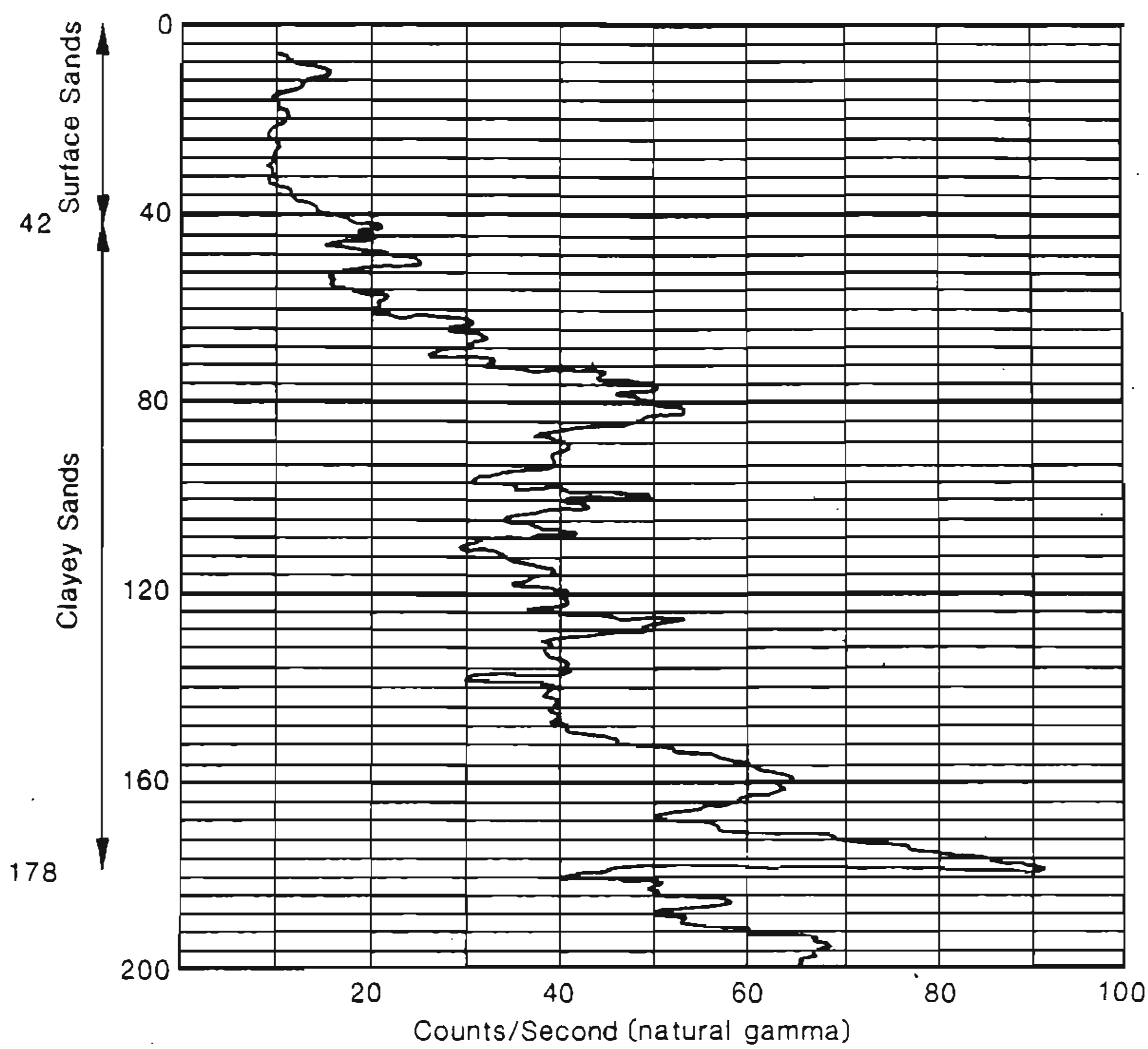


Figure 29. Natural gamma log of a well located 2.0 miles northeast of the titi shrub swamp study site in Apalachicola, Florida.

Table 45. Monthly precipitation (P) and estimated runoff in inches for 1982-1984.

	1982	Runoff		1983	Runoff		1984	Runoff	
Month	P	AMC II	AMC III	P	AMC II	AMC III	P	AMC II	AMC III
Jan	2.64	.07	0.38	4.30	0.22	1.09	4.73	0.40	1.45
Feb	6.21	1.33	2.75	5.49	0.57	1.89	3.93	0.54	1.45
Mar	8.02	2.76	4.71	4.97	0.05	0.81	6.08	0.80	2.49
Apr	3.34	0.57	1.51	12.14	3.93	7.11	9.18	4.15	6.18
May	1.48	0.01	0.19	0.25	0.00	0.00	0.32	0.00	0.00
Jun	5.56	1.03	2.39	8.03	0.97	3.01	3.37	0.05	0.51
Jul	10.84	1.89	4.43	2.24	0 00	0.15	18.07	3.67	8.26
Aug	4.54	0.06	0.66	6.37	0.50	1.91	4.72	0.67	1.69
Sep	15.37	6.03	9.30	6.89	0.68	2.35	1.25	0.07	0.35
Oct	6.88	1.56	3.41	2.05	0.03	0.36	1.78	0.10	0.44
Nov	2.18	0.07	0.51	6.69	1.44	3.17	2.16	0.00	0.27
Dec	4.90	0.23	1.35	5.96	0.83	2.28	0.91	0.00	0.10
		15.61	31.59		9.22	24.13		10.45	23.19
SUM	71.96	23.60		65.38	16.67		56.50	16.82	
%	100	32.80		100	25.50			29.76	

Table 46. The water budget for the titi shrub swamp in Apalachicola, Florida, for 1982-1984.

Year	Inflow		Outflow	
	Precipitation inches (%)	Runoff inches (%)	Groundwater inches (%)	Evapotranspiration inches (%)
1982	71.96 (100)	23.60 (32.80)	0.37 (0.51)	47.99 (66.69)
1983	65.38 (100)	16.67 (25.50)	0.37 (0.57)	48.34 (73.94)
1984	56.50 (100)	16.82 (29.76)	0.37 (0.65)	39.31 (69.57)
Average	64.61 (100)	19.03 (29.35)	0.37 (0.58)	45.21 (70.07)

The average water budget for the 3-yr pre-effluent period indicated that for a total of 64.61 in. of rainfall entering the system, 19.03 in. or 29.35% left through runoff, 0.37 in. or 0.58% left through groundwater outflow, and 45.21 in. or 70.07% left through evapotranspiration.

The maximum soil moisture used to calculate the soil water loss and the actual evapotranspiration, was 2.01 in. This was determined from an average curve number (Cn) of 83.25 (AMC II Cn=77, AMC III Cn=89.5) for the soils at the study site. The average curve number (Cn) was used because annual estimated runoff was determined based on the average of the two runoff estimate methods.

During months when precipitation was greater than potential evapotranspiration, actual evapotranspiration was taken to be equal to potential evapotranspiration. The difference between precipitation and potential evapotranspiration is termed the water surplus. A water surplus occurred in the months from November through March except for December 1984 when very low precipitation (0.91 in.) was recorded (74-yr average for December for this weather station = 3.79 in., Kennedy 1982). These are the coldest months when transpiration is low. During months when precipitation was less than potential evapotranspiration, water was lost from the soil. The difference between the potential evapotranspiration and actual evapotranspiration is termed the water deficit. A water deficit always occurred in May during the 5-yr study period and in June for 4 of the 5 yr that were analyzed. This water deficit coincides with the spring burst of productivity. Water deficits also occurred at various times from 1982 through 1986 in April, July, August, September and October.

For the 5-yr period from 1982 through 1986 the annual average potential evapotranspiration was 41 in. or 63% of precipitation and the annual average actual evapotranspiration was 36 in. or 56% of precipitation (Table 43). The potential evapotranspiration and the actual evapotranspiration for the water budget year were similar (43 in. and 37 in.) to annual average values for the 5-yr period from 1982 through 1986.

Annual average pan evaporation for the Milton weather station for the 5-yr period from 1982 through 1986 was 58 in. Annual average potential evapotranspiration for the Apalachicola weather station for the 5-yr period from 1982 through 1986 was 41 in. Therefore potential evapotranspiration was 0.71 of pan evaporation.

Water Budget

The simplified water budget equation for the study site is $P = R + G + ET$. The water budget for the water budget year is similar to the water budget for the 5-yr period from 1982 through 1986. All values presented below are the average for the 5-yr period.

Precipitation was 66 in. Estimated runoff was 20 in. or 30% of precipitation reaching the study site. Estimated actual evapotranspiration was 36 in. or 56% of precipitation reaching the study site. Estimated potential evapotranspiration was 41 in. or 63% of precipitation reaching the study site. If runoff and estimated actual evapotranspiration are added together (56 in.) they account for 86% of precipitation reaching the study site. The difference between these components and precipitation reaching the study site is water budget residual I (RES-I), which was 10 in. or 15% of precipitation reaching the study site. If runoff and estimated potential evapotranspiration are added together (61 in.) they account for 93% of precipitation reaching the study site. The difference between these components and precipitation reaching the study site is water budget residual II (RES-II), which was 5 in. or 8% of precipitation reaching the study site. This value is used as the estimate of groundwater flow.

Infiltration in this wetland is low due to the semi-impermeable organic humate layer. Water is held above this layer and leaves the wetland predominantly through runoff and evapotranspiration. Runoff and evapotranspiration account for from 86 to 93% of precipitation reaching the study site depending on whether actual or potential evapotranspiration is used in the calculation. The annual upper limit of deep seepage groundwater outflow from the study site was calculated to be 8 in. This was less than water budget residual I (RES I = 10 in.). The difference between water budget residual I (RES-I) and the upper limit of deep seepage groundwater outflow (2 in.) could either be due to an increase in storage within the study site or an underestimate of evapotranspiration. Some increase in storage within the study site might be expected because the annual average precipitation for the 5-yr period analyzed was almost 10 in. greater than the 74-yr average for the Apalachicola weather station. The water budget residual II (RES II = 5 in.) was less than the upper limit of deep seepage groundwater outflow. Therefore, if evapotranspiration is greater than estimated actual evapotranspiration in the simplified water budget and approaches the estimated potential evapotranspiration value then the water budget residual can be accounted for through deep seepage groundwater flow. A value for deep seepage groundwater flow less than the estimated upper limit is consistent with low infiltration in the system.

Transpiration

Transpiration was measured on 11 days. There was standing water present and it never rained when transpiration was measured. Chamber I was used on October 21, 1984, and November 2, 1984. Chamber II, a shorter chamber, was used on April 21, 1985, and was destroyed by a hurricane in August 1985. Chamber III was used for the remaining runs, between October 5, 1985, and September 14, 1986. Transpiration of black titi was measured on November 2, 1984. Transpiration of sweetbay was measured on the other 10 days.

Each day transpiration was measured represents one run and a spread sheet for each run was developed. Each spread sheet indicates the measurements made in the field every 15 min for ambient temperature outside the chamber, intake dew point temperature, exhaust dew point temperature, measured flow and solar input. The leaf biomass measured at the end of each run is also presented. The spread sheet also indicates calculated values for ambient saturation vapor pressure, intake saturation vapor pressure, exhaust saturation vapor pressure, ambient absolute humidity, intake absolute humidity at saturation, exhaust absolute humidity at saturation, relative humidity, saturation deficit, flow rate, transpiration rate, leaf area, transpiration rate per leaf area, transpiration rate per biomass and the leaf biomass to area ratio at the vertical interval where transpiration was measured (9 to 12 m).

Plots of ambient temperature outside the chamber, solar input, relative humidity, saturation deficit, transpiration rate and transpiration rate per leaf area for each run are presented in Figures 30 through 40. Daily transpiration rates per leaf area are also indicated. Transpiration never began before 8:45 AM and always ended by 6:15 PM. The shortest period of transpiration was 7 hr on December 14, 1985, and the longest periods of transpiration were 9 hr on June 28, 1986, and on August 20, 1986. In general, for all runs, as solar input, ambient temperature and saturation deficit increased, transpiration increased. Conversely, as solar input, ambient temperature and saturation deficit decreased, transpiration decreased. An inverse relationship existed between relative humidity and transpiration.

The vertical distribution of leaf biomass for black titi and sweetbay in the bay swamp community was greatest in the vertical interval from 9 to 12 meters (Table 47). Most of the leaf biomass was concentrated in the canopy where transpiration was measured.

Daily transpiration rates ranged from 412 to 1924 g H_2O/day (Table 48). There was a definite seasonality in the results as the lowest values occurred in December and March, and the highest values occurred in May and June. The average daily transpiration rate for the ten sweetbay runs was 1154 g H_2O/day. The daily transpiration rate for the black titi run was 1073 g H_2O/day. Daily transpiration rates per leaf area ranged from 651 to 3124 g H_2O/m^2 leaf area · day (Table 48). The average daily transpiration rate per leaf area for the ten sweetbay runs was 1593 g H_2O/m^2 leaf area · day. The daily transpiration rate per leaf area for the black titi run was 801 g H_2O/m^2 leaf area · day.

Daily transpiration rates per ground area ranged from 866 to 4155 g H_2O/m^2 ground area · day (Table 48). The average daily transpiration rate per ground area for the ten sweetbay runs was 2118 g H_2O/m^2 ground area · day. The daily transpiration rate per ground area for the black titi run was 2523 g H_2O/m^2 ground area · day. Therefore, the daily transpiration rate per ground area for the bay swamp community was 4641 g H_2O/m^2 ground area · day.

Daily transpiration rates per ground area for a species in each run were calculated as the product of the daily transpiration rate per leaf area and the leaf area index for that species throughout its entire vertical distribution (Table 47). Daily transpiration rates per ground area for a species in each run can also be calculated as the product of the daily transpiration rate per leaf area and the leaf area index for that species in the vertical interval where transpiration was measured (9 to 12 m) (Table 47). This seems

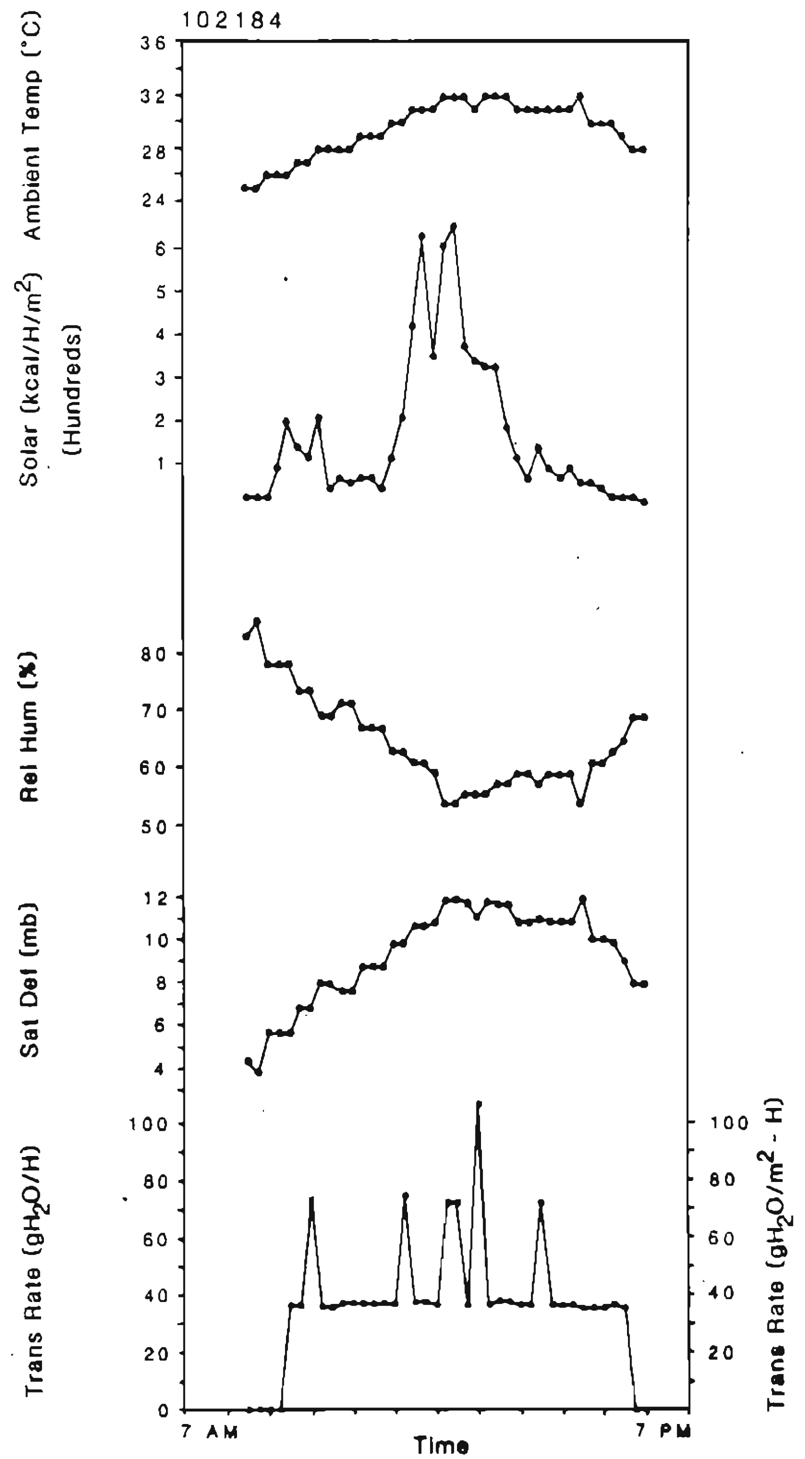


Figure 30. Transpiration run on sweetbay at Apalachicola wetland, 21 October 1984. DTRLA = 1459 g H_2O/m^2 leaf area-day.

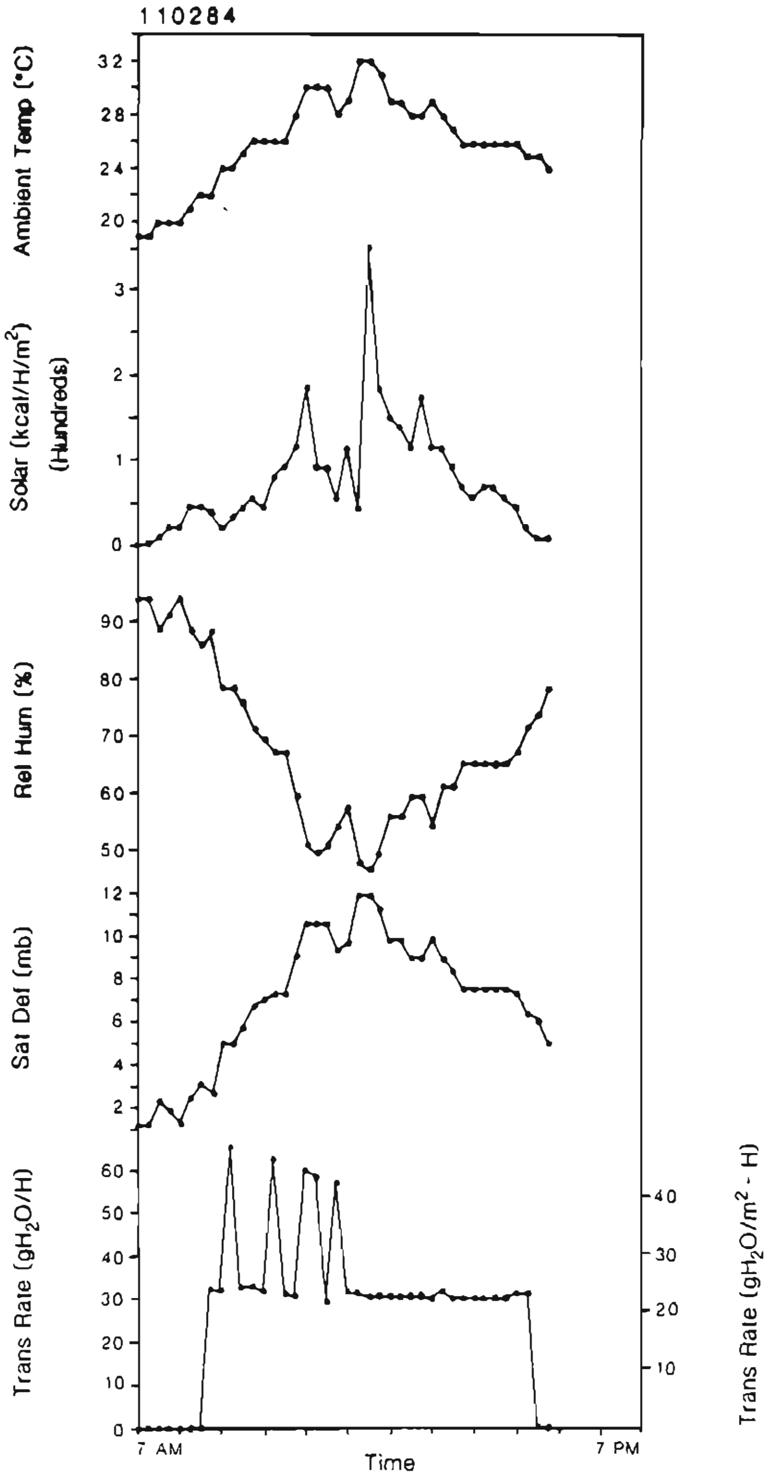


Figure 31. Transpiration run on black titi at Apalachicola wetland, 2 November 1984. DTRLA = 801 g H_2O/m^2 leaf area-day.

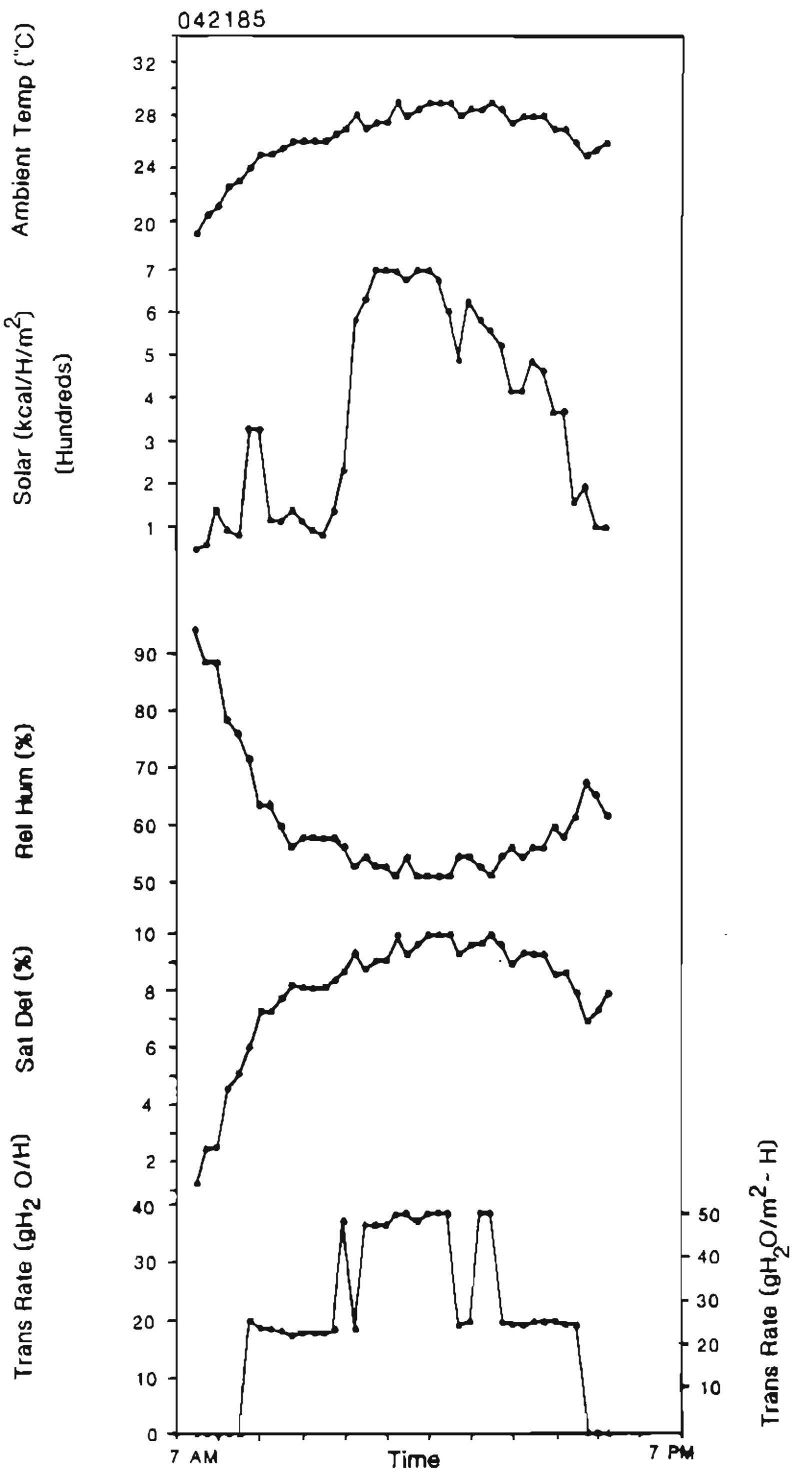


Figure 32. Transpiration run on sweetbay at Apalachicola wetland, 21 April 1985. DTRLA = 1000 g H_2O/m^2 leaf area-day.

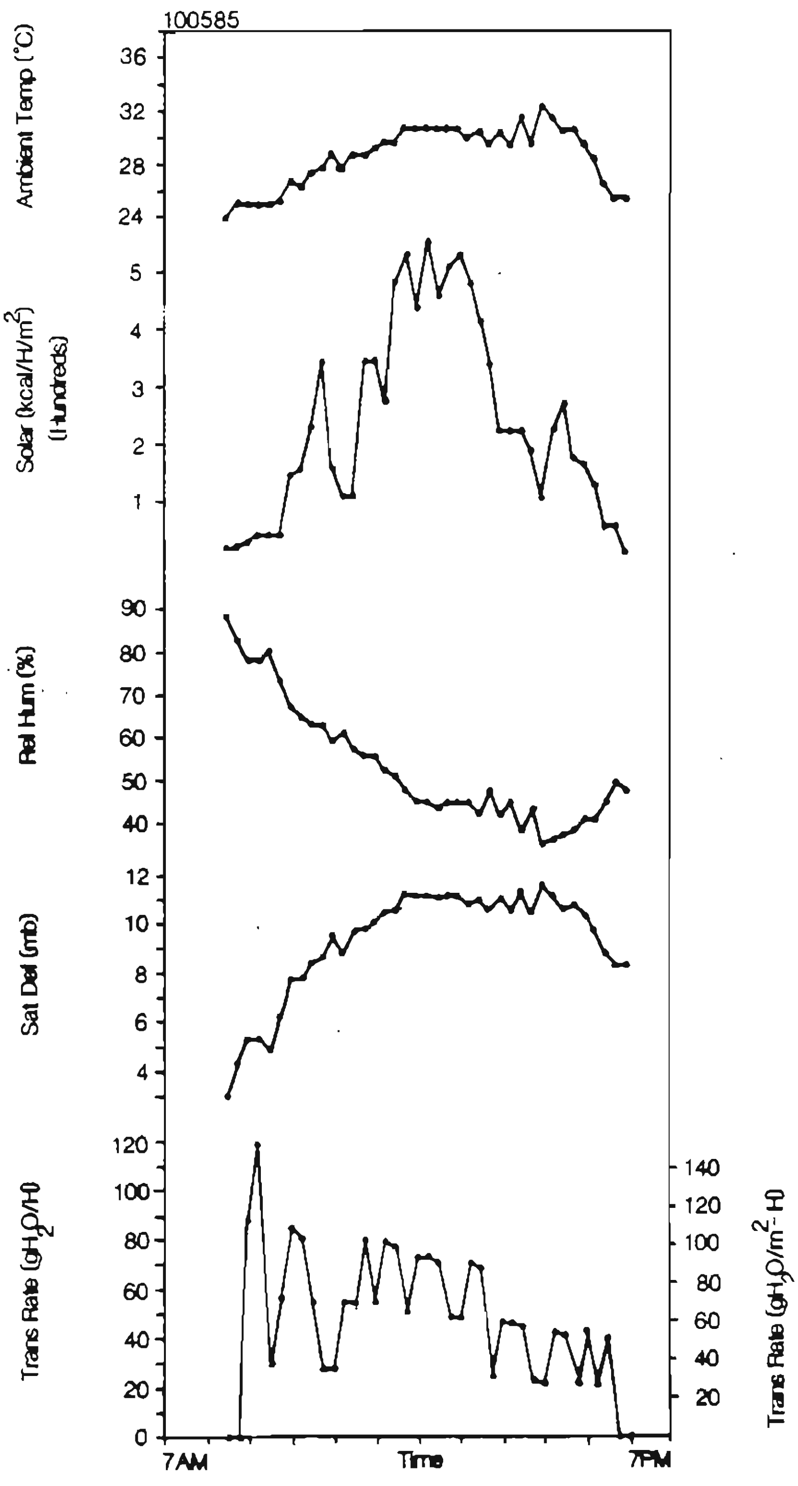


Figure 33. Transpiration run on sweetbay at Apalachicola wetland, 5 October 1985. DTRLA = 2436 g H_2O/m^2 leaf area-day.

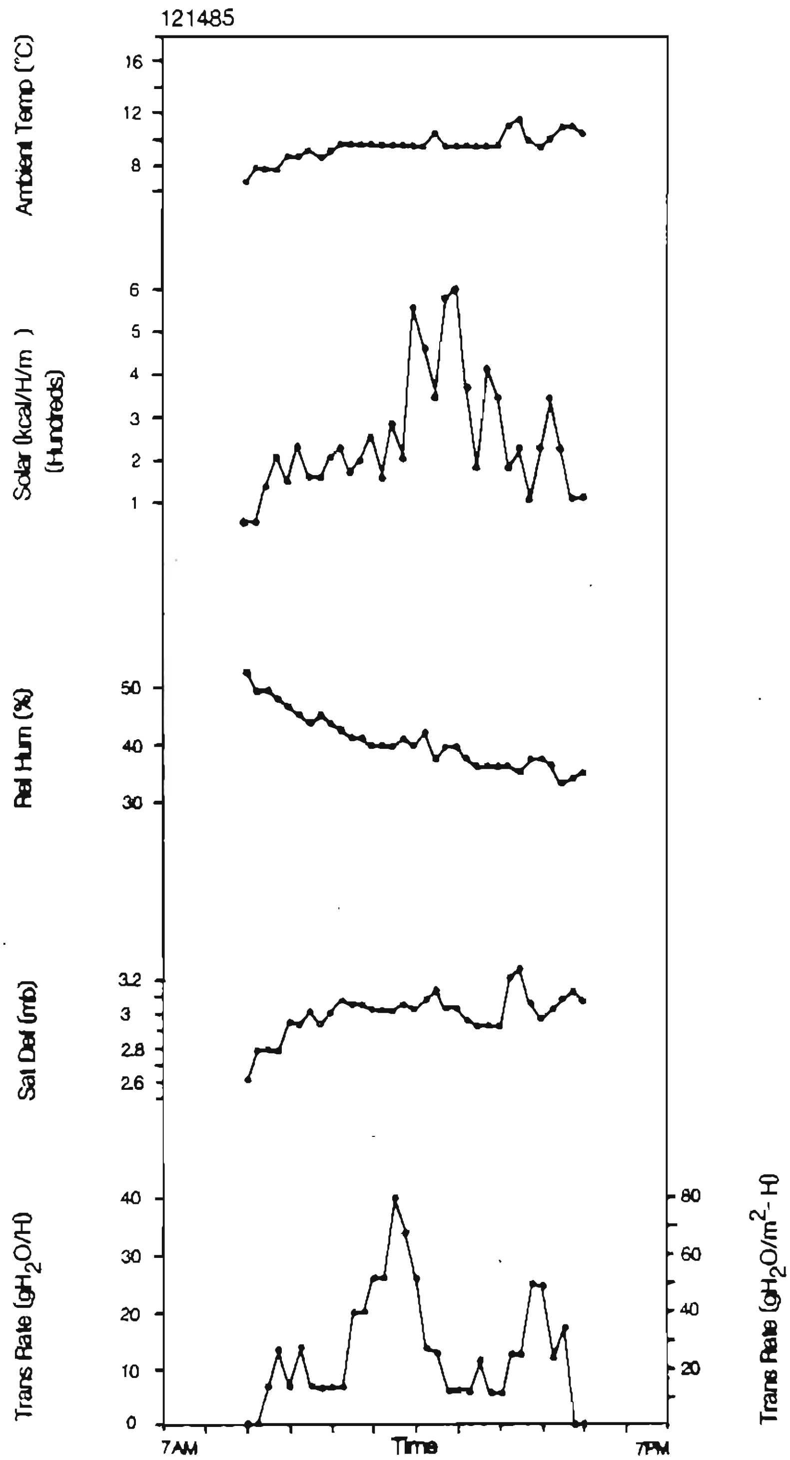


Figure 34. Transpiration run on sweetbay at Apalachicola wetland, 14 December 1985. DTRLA = 1144 g H_2O/m^2 leaf area-day.

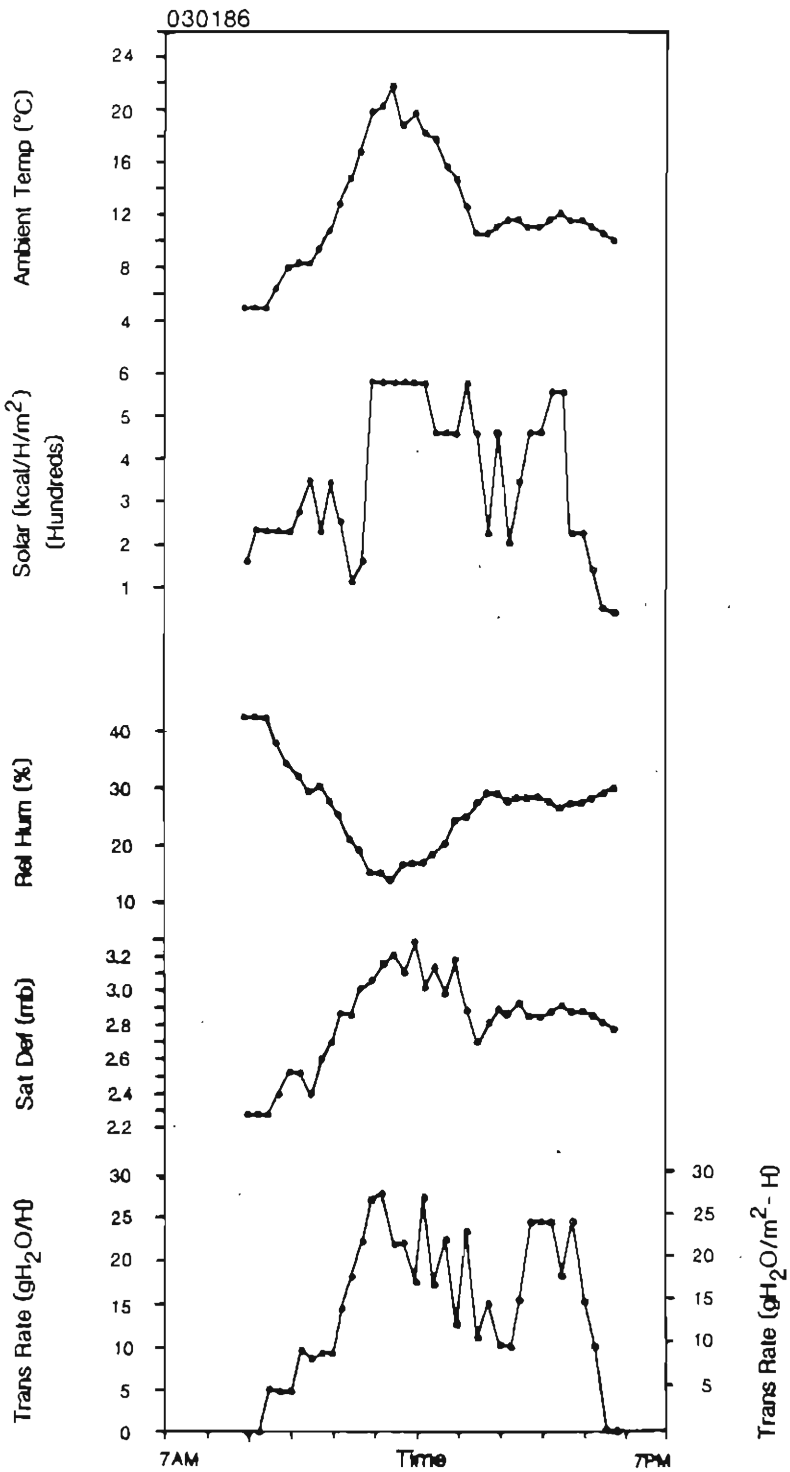


Figure 35. Transpiration run on sweetbay at Apalachicola wetland, 3 March 1986. DTRLA = 651 g H_2O/m^2 leaf area-day.

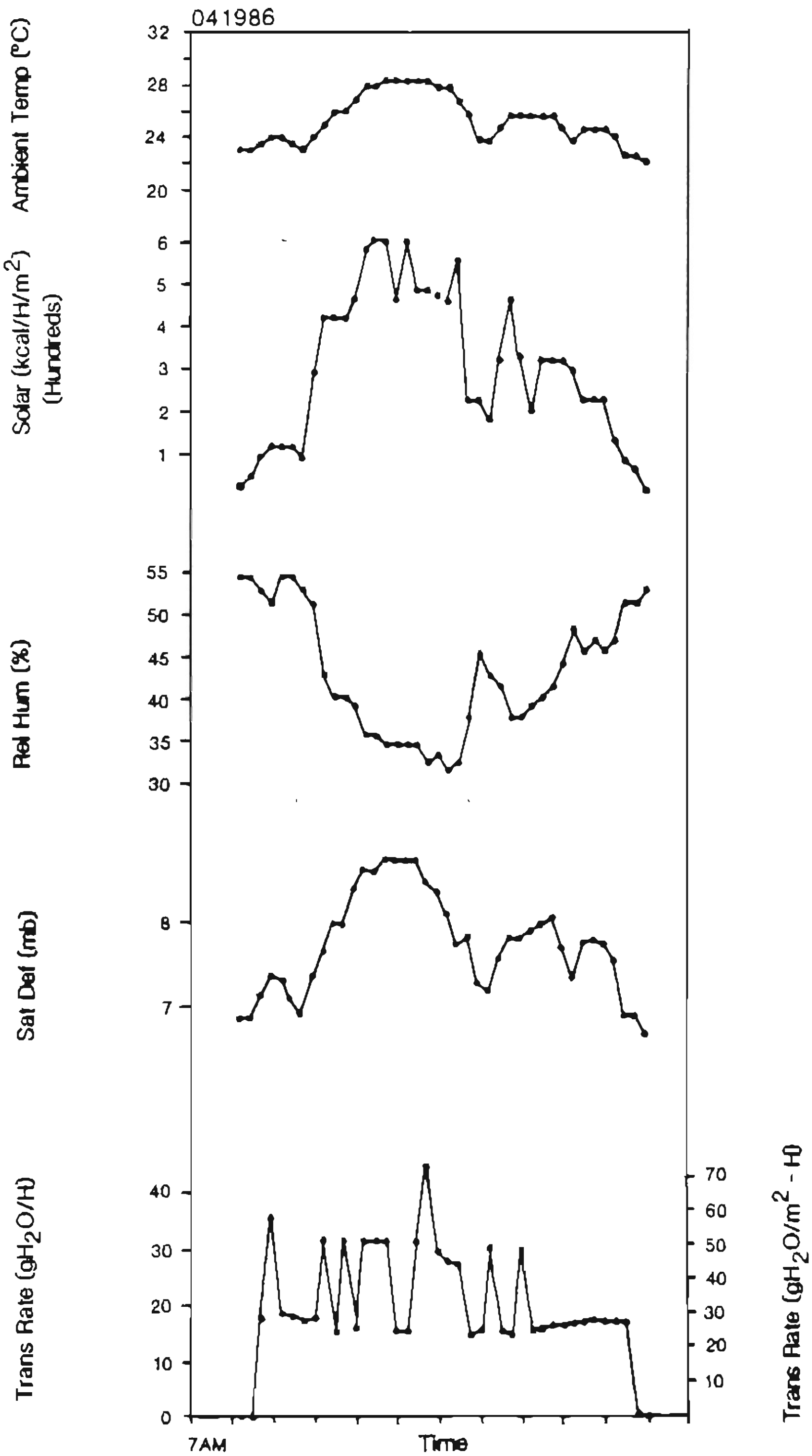


Figure 36. Transpiration run on sweetbay at Apalachicola wetland, 19 April 1986. DTRLA = 1553 g H_2O/m^2 leaf area-day.

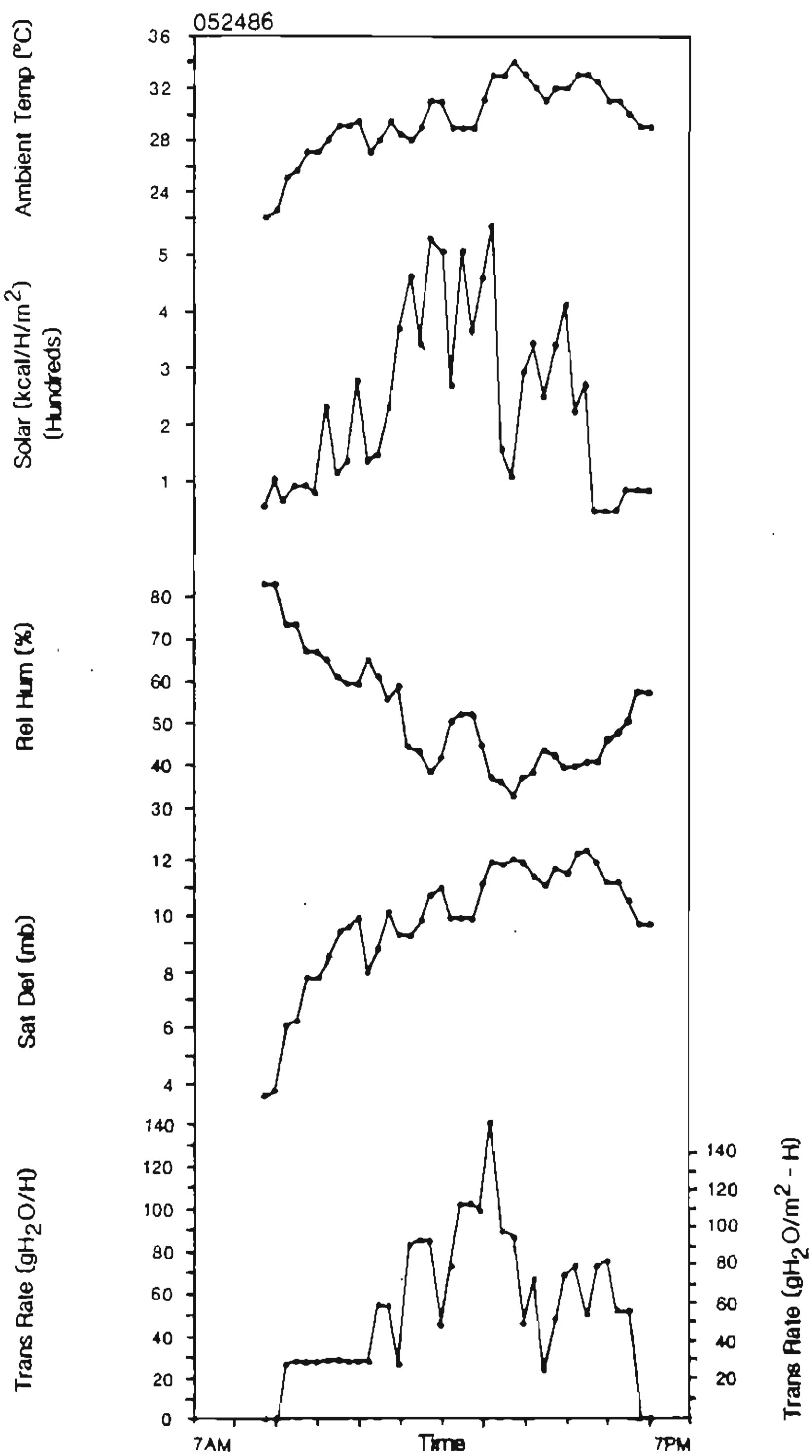


Figure 37. Transpiration run on sweetbay at Apalachicola wetland, 24 May 1986. DTRLA = 2318 g H_2O/m^2 leaf area-day.

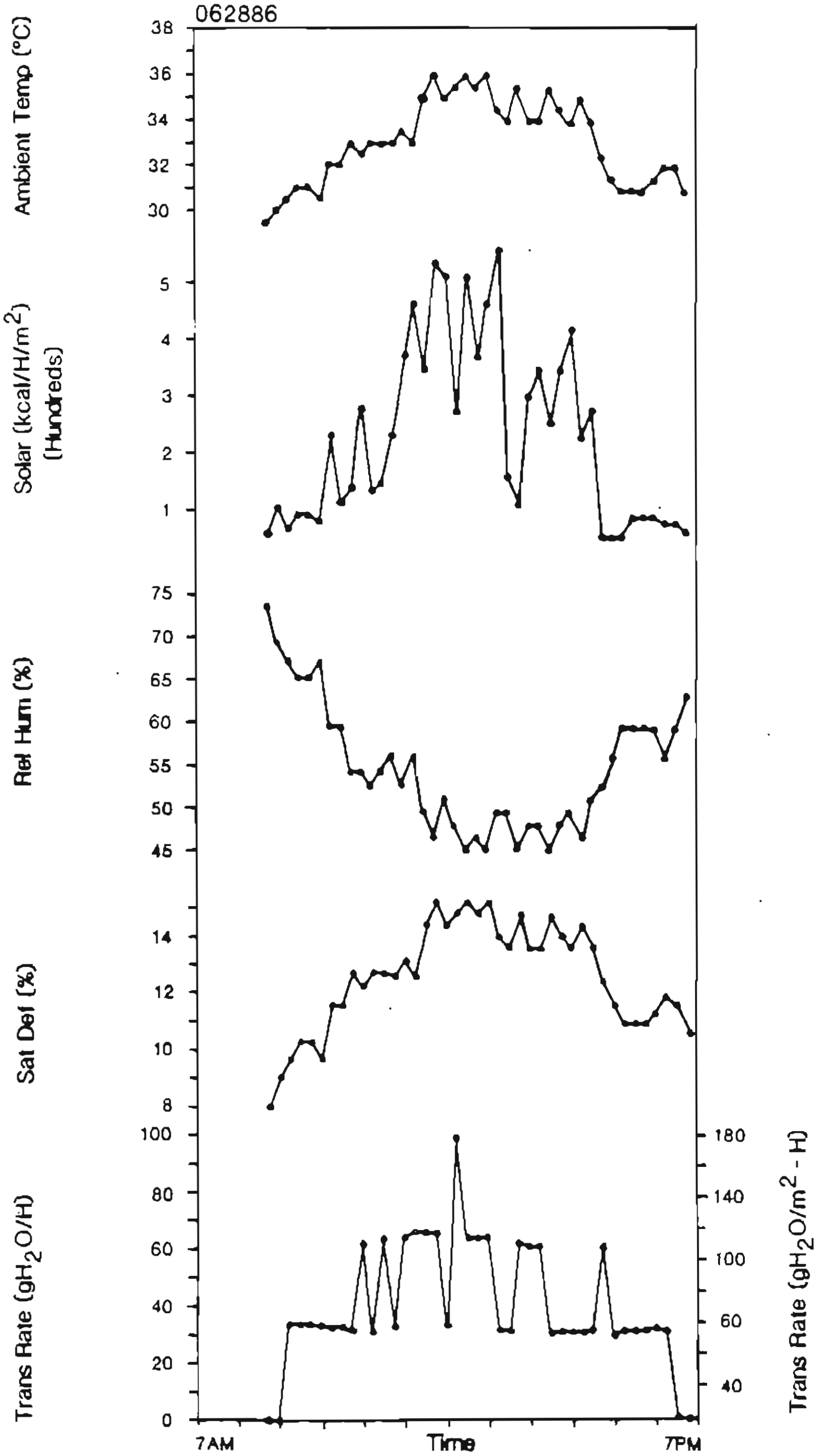


Figure 38. Transpiration run on sweetbay at Apalachicola wetland, 28 June 1986. DTRLA = 3124 g H_2O/m^2 leaf area-day.

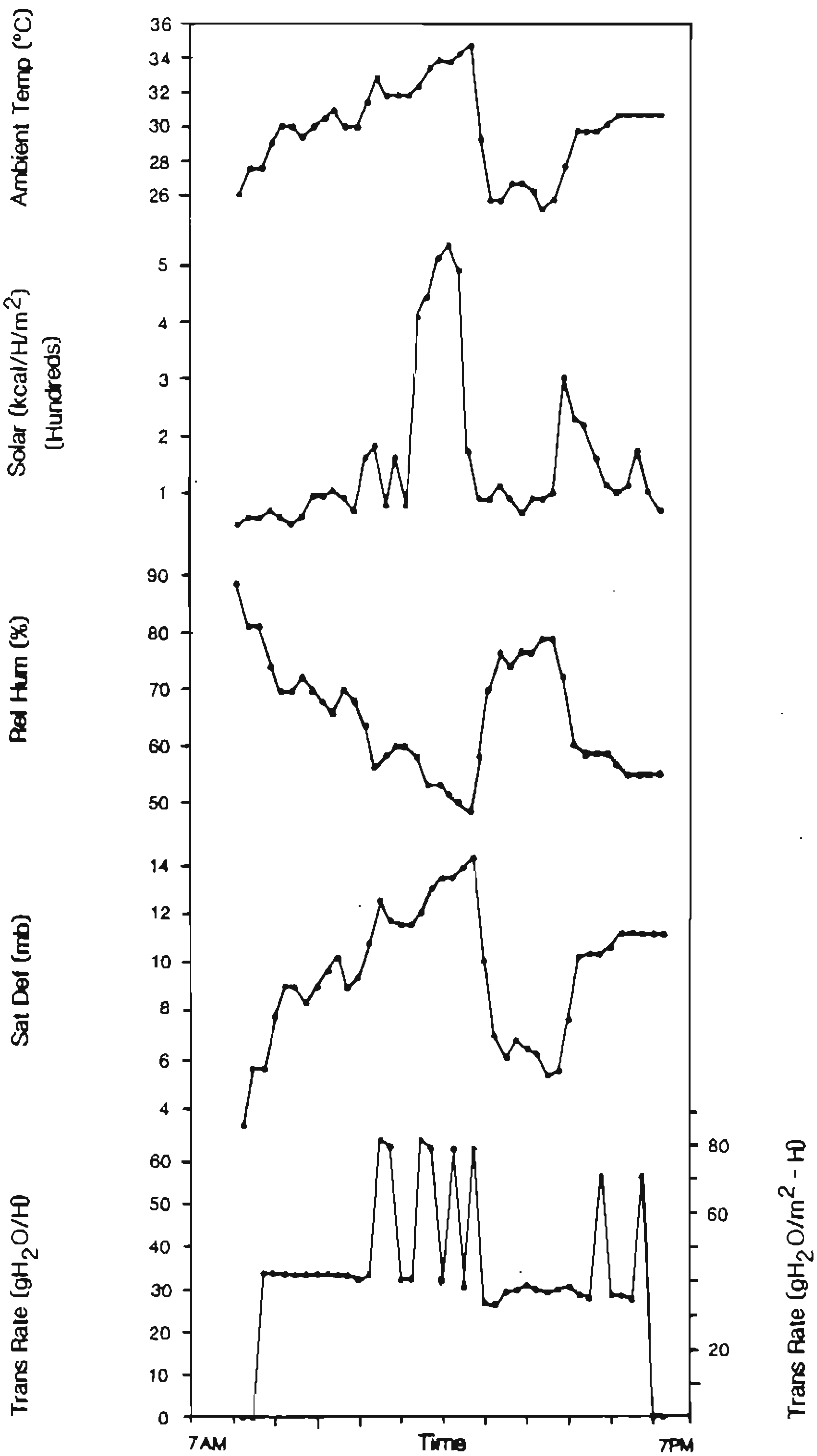


Figure 39. Transpiration run on sweetbay at Apalachicola wetland, 20 August 1986. DTRLA = 1459 g H_2O/m^2 leaf area-day.

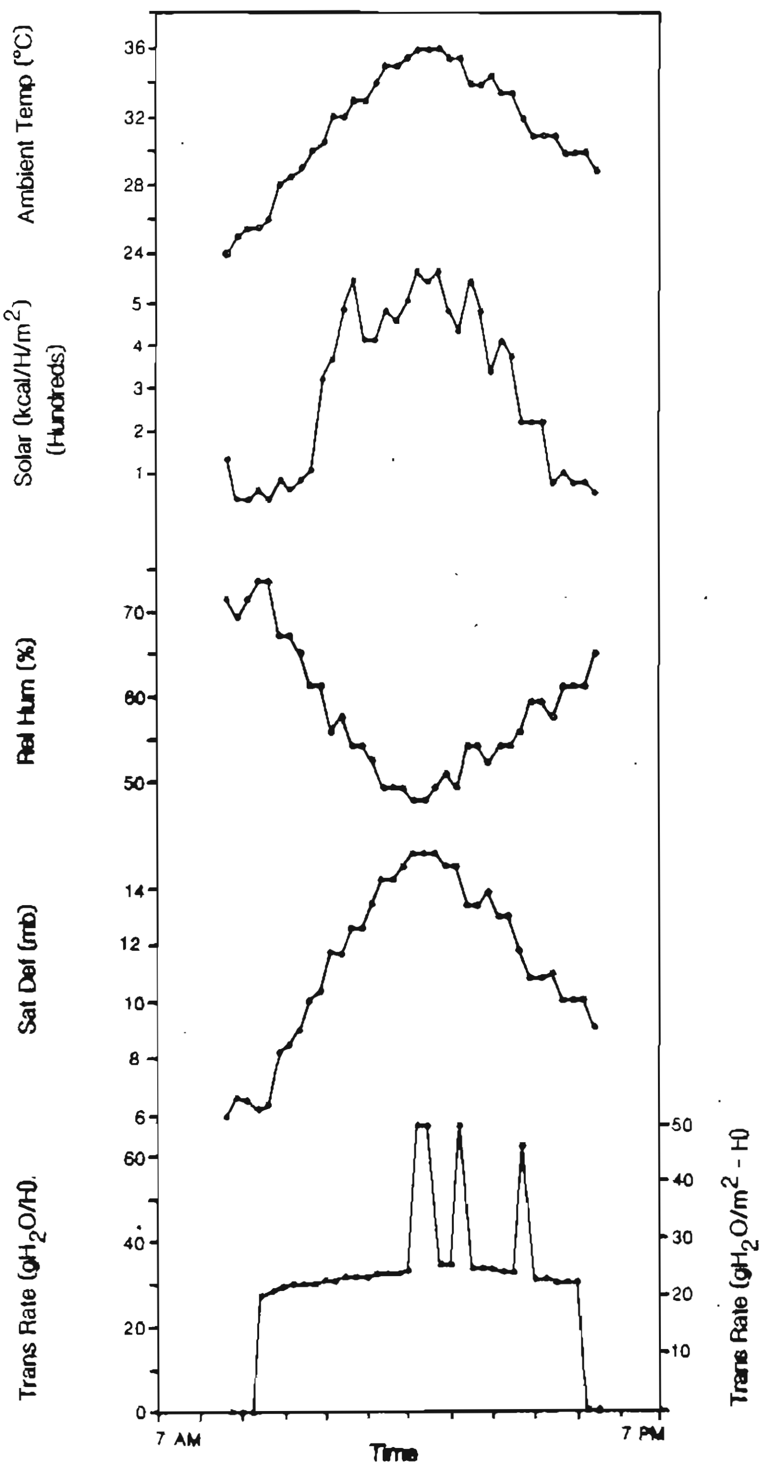


Figure 40. Transpiration run on sweetbay at Apalachicola wetland, 14 September 1986. DTRLA = 694 g H_2O/m^2 leaf area-day.

Table 47. The vertical distribution of leaf biomass (VDLB) and the leaf area index (LAI) for black titi and sweetbay in the bay swamp community.

Species	interval (m)	hits/ intervals	as % of 48	LBAR[1] g/m² leaf area	VDLB as % of total	LBGA[2] g/m² ground area	LAI[3] m² leaf area/ m² ground area
Sweetbay	9-12	23/48	0.48	109.4	74	141.90	0.96
	3-9	9/48	0.19	98.4	26	141.90	0.37
							1.33
Black titi	9-12	35/48	0.73	140.4	84	434.16	2.60
	3-9	7/48	0.15	126.8	16	434.16	0.55
							3.15
						TOTAL	4.48

[1] LBAR from Table 17
[2] LBGA from Table 18
[3] LAI = (LBGA/LBAR) x VDLB

Table 48. Daily transpiration rate (DTR), daily transpiration per leaf area (DTRLA) and daily transpiration rate per ground area (DTRGA) for eleven transpiration runs.

Date	Chamber	DTR gH_2O/day	LA m^2	DTRLA[1] gH_2O/m^2 leaf area -day	DTRGA[2] gH_2O/m^2 ground area -day	PE at Milton,FL (in)
Oct 21, 1984	I	1415	0.97	1459	1940	0.21
Nov 02, 1984	I*	1073	1.34	801	2523	0.14
Apr 21, 1985	II	730	0.73	1000	1330	0.22
Oct 05, 1985	III	1778	0.73	2436	3240	0.16
Dec 14, 1985	III	412	0.36	1144	1522	0.20
Mar 01, 1986	III	514	0.79	651	866	0.13
Apr 19, 1986	III	761	0.49	1553	2065	0.18
May 24, 1986	III	1924	0.83	2318	3083	0.25
Jun 28, 1986	III	1593	0.51	3124	4155	0.19
Aug 20, 1986	III	1348	0.87	1549	2060	0.22
Sep 14, 1986	III	1068	1.54	694	923	0.18

[1] DTRLA = DTR/LA
[2] DTRGA = DTRLA x LAI (Table 35)

DTR (mean of all sweetbay) = 1154 gH_2O/day
DTRLA (mean of all sweetbay) = 1593 gH_2O/m^2 leaf area-day
DTRGA (mean of all sweetbay) = 2118 gH_2O/m^2 ground area-day

*-Cliftonia monophylla-black titi
all other runs Magnolia virginiana-sweetbay

PE average of all runs = 0.19 in. = 4.8 mm appropriate because as Brown *et al.* (1984) indicated, transpiration rates decreased for leaves lower in the canopy for the wetlands investigated. When calculated in this manner, the average daily transpiration rate per ground area for the ten sweetbay runs was 1529 g H_2O/m^2 ground area · day, and the daily transpiration rate ground area for the black titi run was 2083 g H_2O/m^2 ground area · day. Therefore, if the measured rate of transpiration only occurs in the canopy, then the daily transpiration rate per ground area calculated in this manner for the bay swamp community was 3612 g H_2O/m^2 ground area · day.

Soil surface and water surface water losses in Austin Cary cypress dome were 460 and 973 g H_2O/m^2 ground area · day, respectively (Brown 1981). The average of these two values (717 g H_2O/m^2 ground area · day) for Austin Cary cypress dome was used in this study as an estimate of forest floor evaporation in the bay swamp community.

Methods to estimate actual evapotranspiration incorporate factors that reflect different levels of water availability. In the Thornthwaite method, if monthly precipitation is less than potential evapotranspiration then water was lost from the soil, and actual evapotranspiration is less than potential evapotranspiration. The Thornthwaite method does not correct for changes in relative humidity, cloud cover and other solar radiation effects, wind or cover type. Therefore, results from this method can only be interpreted as average values, particularly for estimates of actual evapotranspiration when water availability is high. Prediction formulae are broad generalized equations that do not compensate for variation in transpiration among species (Scheffe 1978).

Average potential evapotranspiration and actual evapotranspiration were calculated by Dohrenwend (1977) using the Holdridge method for 21 weather stations in Florida for a five year period. The precipitation, potential evapotranspiration, potential evapotranspiration to precipitation ratio, actual evapotranspiration and the actual evapotranspiration to potential evapotranspiration ratio reported by Dohrenwend (1977) for weather stations in Milton and Tallahassee, Florida, are presented in Table 49. These are the closest stations to the Apalachicola weather station for which data are reported. Corresponding values for these parameters and ratios calculated in this study are also presented. The average annual precipitation at the Apalachicola weather station, for the five year period analyzed, was greater than the average annual precipitation reported by Dohrenwend (1977) for the other two stations. Therefore, the potential evapotranspiration and the actual evapotranspiration values were also greater. Although these values were greater at the Apalachicola weather station, both the potential evapotranspiration to precipitation and actual evapotranspiration to potential evapotranspiration ratios for the Apalachicola weather station were similar to the ratios reported by Dohrenwend (1977) for the other two stations. Therefore, the estimate of these parameters in this study are consistent with what may be considered average values for this region of the state.

The leaf area index, daily transpiration rate per leaf area, daily transpiration rate per ground area, forest floor evaporation, total water loss and pan ratio for the dwarf cypress forest, Austin Cary cypress dome and floodplain forest reported by Brown (1981) and for the bay swamp community at the study site are presented in Table 50. The daily transpiration rate per leaf area for black titi in the bay swamp was similar to values for hardwood species in the floodplain forest. The average daily transpiration rate per leaf area for sweetbay in the bay swamp was lower than values for cypress in the dwarf cypress forest and in Austin Cary cypress dome. This indicates relatively low levels of transpiration per leaf area for these species.

Daily transpiration rates per ground area for the bay swamp were greater than values for the dwarf cypress forest and Austin Cary cypress dome. The increase from the species level to the community level is reflected in greater leaf area index values in the bay swamp as compared to the dwarf cypress forest and Austin Cary cypress dome. Total water loss for the bay swamp is greater than for the dwarf cypress forest and Austin Cary cypress dome but less than for the floodplain forest. The estimated leaf biomass of the woody vegetation in the bay swamp community where transpiration was measured was similar to the value reported by Brown (1978) for the floodplain forest. Therefore, this community may be structurally similar to a floodplain forest and may transpire at a similar rate.

When transpiration data are collected under different atmospheric conditions of humidity, wind and sunlight (as was the case in this study), a useful index for comparison purposes is the pan ratio (Brown 1981). The pan ratio is the ratio of total water loss to pan evaporation. The average pan evaporation for the eleven days when transpiration was measured was 0.19 in. (4.8 mm). Open water evaporation is typically 0.7 to 0.8 of pan evaporation (Veihmeyer 1973). The pan ratio of the dwarf cypress forest and Austin Cary cypress dome were lower than the pan ratio of open water suggesting that these swamps may conserve water. The pan ratio of the bay swamp and the floodplain forest were higher than the pan ratio of open water.

Potential evapotranspiration was 0.71 of pan evaporation suggesting that on average, evapotranspiration was similar to open water evaporation. But in communities where the pan ratio exceeds the pan ratio of open water it appears as though total water loss may be greater than the average calculated for the entire system. Considering that estimates of potential evapotranspiration may underestimate evapotranspiration for certain systems when water availability is high, and that transpiration may exceed open water evaporation, as are indicated above, it is not surprising that the total water loss from certain wetland communities is greater than the average calculated for the entire system. The relative aboveground biomass and estimated leaf biomass for the titi and black gum swamps at the study site were lower than for the bay swamp. Therefore less total water loss is expected from these communities but this can only be substantiated with field measurements.

Table 49. Precipitation (P), potential evapotransporation (PET), PET/P ratio, actual evapotranspiration (AET) and AET/PET ratio for Milton and Tallahasse, Florida reported by Dohrenwerd (1977) and for Apalachicola calculated in this study.

	P (in)	PET (in)	PET/P ratio	AET	AET/PET (%)
Milton	59.09	39.21	0.66	32.56	83
Tallahassee	56.85	39.68	0.68	32.95	83
Apalachicola (1982-1986)	65.64	41.08	0.63	36.48	89
Apalachicola (WBY)	64.71	43.20	0.67	37.14	86

Table 50. Leaf area index (LAI), daily transpiration rate per leaf area (DTRLA) daily transpiration rate per ground area (DTRGA), forest floor water loss (FFWL) and total water loss (TWL) for the dwarf cypress forest, Austin Cary cypress dome and floodplain forest reported by Brown (1981) and for the bay swamp community at the study site.

Community		Dwarf cypress forest	Austin Cary Cypress Dome	Bay swamp	Floodplain forest
LAI		0.5	3.4	4.6	8.5
<u>Transpiration</u>	<u>Vegetation</u>				
DTRLA	cypress	1840	2125	-	544
gH_2O/m^2	hardwood	-	527	-	868
leaf area/day	black titi	-	-	801	-
	sweetbay	-	-	1593	-
DTRGA	cypress	932	1679	-	2106
gH_2O/m^2	hardwood	-	1394	-	3099
ground area/day	black titi	-	-	2083	-
	sweetbay	-	-	1529	-
TOTAL		932	3073	3612	5205
<u>Evaporation</u>					
FFWL gH_2O/m^2 ground area/day		333	717	717	363
Total Water Loss (TWL)					
gH_2O/m^2 ground area/day		1265	3790	4329	5568
mm/d		1.27	3.79	4.33	5.57
Pan Ratio (PR) = TWL/pan evaporation		0.19	0.66	0.90	0.95

The range of daily evapotranspiration rates in three cypress swamps studied by Ewel (1985) was from 0.2 mm/day (in January) to 5.9 mm/day (in September). These daily maximum evapotranspiration rates were greater than the evapotranspiration rates for the floodplain forest measured by Brown (1978) and for the bay swamp community at the study site, expressed in Table 50 as total water loss. Evapotranspiration in cypress domes was measured by Heimburg (1976) by determining the change in water levels. Evapotranspiration varied seasonally in the cypress domes. Daily transpiration rates also varied seasonally in this study. It appears as though there is variability in evapotranspiration rates between forested wetland communities and between seasons. Forested wetlands may have low evapotranspiration rates when water is limiting and higher evapotranspiration rates when water is readily available.

Wetlands do go dry at one time or another and plants that have adaptations that limit water loss during dry periods may have an adaptive advantage in these systems. When water is limiting wetland vegetation may have the ability to ameliorate water loss and even survive drought periods through morphological and physiological adaptations. A species may require an increase in reflectance, as occurs in xeromorphic leaves, in order to maintain reasonable leaf temperatures (Odum 1984). A strategy of conserving water during dry seasons is achieved by increasing reflectance, reducing transpiration, maintaining peat to hold soil water and shielding water loss through high stem density (Odum 1984). It also appears that during wet seasons a high rate of evapotranspiration can occur and therefore these species have bimodal adaptation in that they are well adapted to both wet and dry conditions. Therefore, although certain forested wetland communities may exhibit high evapotranspiration rates the overall result may be still be conservation of water.

DEVELOPMENT OF ECOSYSTEM RESPONSE MODEL

Purpose

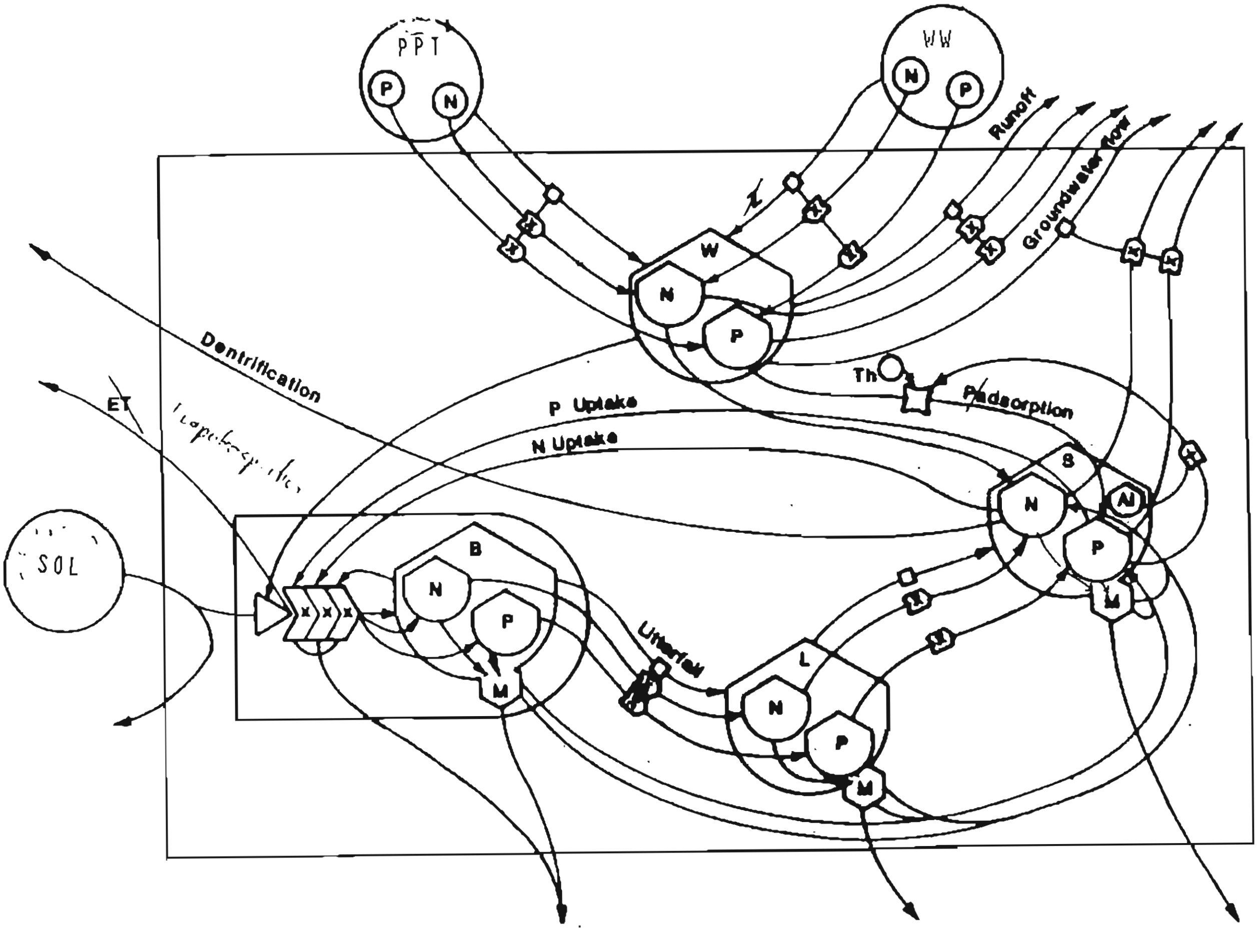
The objectives of this study were to characterize the main components and processes of a titi shrub swamp ecosystem in Apalachicola, Florida, and to predict their long-term response to wastewater addition. The main components were vegetation, water, and soil, and the processes were carbon, nitrogen and phosphorus cycling, and water flow. A model can be used to organize concepts, theories and data collection, describe patterns of energy and nutrient dynamics, and assess environmental impact or change (Mitsch *et al.* 1982). Therefore, a model (Figure 41) was developed in this study to characterize the main components and processes of a titi shrub swamp and to predict their long-term response to wastewater addition. The development of the model requires synthesis of the results from the other project tasks along with information from previous studies on other wetland ecosystems.

Literature Review of Freshwater Wetland Models and their Use in Simulating Wastewater Addition

The list for general ecosystem models is enormous and the number of models of freshwater wetlands in the literature is large (Costanza and Sklar 1985). These authors provided a systematic review of freshwater wetland models that use some kind of formal mathematical description, either explicit equations or system diagrams with implied equations. The representative but not exhaustive review turned up 87 models in 59 different studies. There were 18 forested swamp models, 9 bottomland hardwood models, 14 emergent marsh models, 5 floating marsh models, 30 shallow lake models, 2 bog or fen models, 4 tundra models, and 5 combination models. More than 60% of the models were non-linear.

There are two major types of ecological models, which can be classified for convenience as analytic models and simulation models (Hall and Day 1977). Analytic models use mathematical procedures to find exact solutions to differential and other equations. These models are not generally used to study whole ecosystems because they cannot be used to solve many non-linear systems of equations that may provide a better description of an ecosystem. Simulation models, on the other hand, do not give an

Figure 41. Conceptual systems diagram of the titi shrub swamp near Apalachicola, Florida. W = water, B = biomass, L = litter, S = soil, N = nitrogen, P = phosphorus, M = microbes.



exact solution to an equation over time, and, therefore, one type of error associated with these models is related to the inexact nature of the solution technique used. Simulation models can solve many equations nearly simultaneously and can incorporate non-linearity (Hall and Day 1977).

A review of models of North American freshwater wetlands, emphasizing models which were simulated on the computer, indicated there is a wide diversity of types of models used to describe and simulate wetland dynamics (Mitsch *et al.* 1982). The major types of wetland models were classified into seven categories: energy/nutrient ecosystem models, hydrology models, spatial ecosystem models, tree growth models, process models, causal models, and regional energy models.

In energy/nutrient ecosystem models, materials pass through or cycle among biotic and abiotic components and exchange with the surroundings. These models are generally non-spatial, aggregated models with feedbacks and interactions among components. Both energy flow and nutrient cycling can be combined into one model. In spatial ecosystem models the attributes of ecosystem models are combined with spatial transport models (hydrodynamic transport models) describing wetland hydrology and pollutant transport over short periods and large areas. Although the dynamics of wetlands have been represented by a variety of ecological models, often involving great detail and complexity, few spatially distributed models have emerged (Mitsch 1983). Hydrodynamic transport models describing stream flow and storm runoff have been developed for wetlands (Hopkinson and Day 1980), and a model has been developed for overland flow through vegetated areas (Hammer and Kadlec 1986). Some of the energy/nutrient and spatial ecosystem models described by Mitsch et al. (1982) were developed to simulate the effects of the addition of wastewater on wetland components. These are described in more detail below.

Simulation models were developed as part of a long range study in north central Michigan to investigate the feasibility of using peatlands for disposal of treated wastewater (Kadlec and Tilton 1979). More specifically, the model predicted long-term changes in biomass and nutrient concentrations in this marsh/bog peatland ecosystem.

Initially, Dixon (1974) developed a model emphasizing the biomass dynamics of the system. This was combined with models of water and nutrient components into a macromodel to predict the effects of the addition of wastewater on these wetland components. This large-scale simulation model was developed by Parker (1974). Spatially, the ecosystem was divided into blocks, which were further divided into units or compartments, each of which represents the behavior of a biotic or abiotic variable. Each unit or compartment was represented by a time varying differential equation. Therefore, a set of ordinary, first-order, non-linear differential, mass balance equations comprised the model (Dixon and Kadlec 1975). This was the first spatially distributed model of a wetland ecosystem used to predict the impacts of wastewater addition. A series of simulations was run varying nitrogen and water parameters to determine the effects on the biomass, water and nutrient components. The simulations were intended to give an indication of the relative effects of added water and nutrients on the wetland and not a prediction of actual result. Dixon and Kadlec (1975) pointed out that actual predictions should await complete updating and validation of the model.

Hammer and Kadlec (1983) then developed a simplified model of wastewater/wetland interactions to account for the movement of water, nutrients, and pollutants within the wetland. The model accounted for the movement of surface water in response to gradient and vegetation flow resistance, and allowed material balances to be determined in a wetland ecosystem receiving wastewater (Hammer 1984). This model also contained partial differential equations (Hammer 1984). The resultant analytical solution to the differential-integral equation described the solute balance in the surface water sheet for this idealized system (Hammer 1984).

The simulated removal of dissolved nutrients from surface waters is a two-step process, consisting of delivery and consumption. Delivery is accomplished by convective mass transfer within surface waters or by downward flow due to water infiltration. Consumption occurs principally at the surface of soil and plants. In addition, two treatment regimes exist in the wetland. In the vicinity of wastewater discharge a saturated region exists. Here component removal rates are quite slow, comprised of uptake due to adsorption in the deep soil, incorporation of material into new soil and woody plants, and microbial release of gases to the atmosphere. Outside this saturated region, surface water concentrations of wastewater components drop exponentially with distance. In this zone of rapid removal, the transport of dissolved components through the water sheet limits the overall rate (Hammer 1984). The combined total area required for assimilation of pollutants over time determines the treatment capacity of the system.

To facilitate the use of the model over long periods of time, all transfers between units or compartments were taken as the annual net accumulation in each compartment and, therefore, the cycling of nutrients and other materials on a seasonal basis was not explicitly addressed (Hammer and Kadlec 1983). This spatially distributed hydrological model provides a convenient means by which the response of natural or constructed wetlands components can be predicted using site specific information (Hammer 1984).

Simulation models were developed for a cypress dome in Florida to investigate management issues (Mitsch 1975a, 1975b; Odum *et al.* 1977; Deghi 1977; Ewel and Deghi 1978; Deghi and Ewel 1984). The models were developed in part to indicate long-term (100 years) dynamics of a cypress dome receiving wastewater.

The model described by Mitsch (1975a, 1975b) and Odum *et al.* (1977) was designed to deal with several management questions involving cypress domes, including the optimum rate of harvesting, possible effects of fire, and their wastewater treatment capability. The model included two autotrophic components, the cypress trees and the understory. The sediment component consisted of nitrogen, phosphorus, organic peat and water. The model was designed to run for 10 to 100 years; therefore, annual variations in solar radiation were ignored. Flows such as litterfall and gross primary production were determined from yearly averages. Primary productivity was modeled with a non-stratified approach (equal competition for sunlight between the two autotrophic compartments) and with a stratified approach (cypress canopy having a competitive advantage). Each plant compartment could utilize 5% of the flow that was available to it. Two pathways for decomposition were designed into the model, their operation dependent on water level. Several limiting nutrient schemes were utilized in the model.

The model described by Deghi (1977), Ewel and Deghi (1978), and Deghi and Ewel (1984) was designed to examine the long-term behavior of phosphorus in the cypress dome subsequent to wastewater addition. Four autotrophic components were distinguished in the model: cypress trees, hardwood trees, understory vegetation, and duckweed. The model was designed to run for 50 years; therefore, annual and seasonal variations in forcing functions were ignored. The amount of sunlight reaching any of the three strata within the cypress dome was related to the biomass of vegetation above it.

Methods

An ecosystem model was developed to characterize the titi shrub swamp wetland ecosystem in Apalachicola, Florida. This ecosystem model was developed using a diagrammatic language presented by Odum (1971, 1972). This energy-flow or material-flow symbolic language is based on a series of modules that represent both systems processes and mathematical functions connected by lines representing transfer pathways of energy, materials or information (Hall and Day 1977). The modular components can be used to construct compartmental models of ecosystems. The language is also a tool for developing computer programs to simulate a system of first order nonlinear differential equations (Costanza and Sklar 1985) and was used to develop a computer program to simulate the discharge of wastewater to the titi shrub swamp in order to predict the effects of the addition of wastewater on wetland components.

The model was developed with the state variable differential paradigm. The model is composed of modules including forcing functions and storage compartments or state variables. The forcing functions were solar radiation, precipitation and wastewater. The storage compartments or state variables were surface water, biomass, litter and soil, and the water, carbon, nitrogen and phosphorus in these compartments. The initial condition of each state variable was specified and transfer coefficients were determined from the values of the storages and flows.

The simulation model was developed to indicate long-term (100 yr) dynamics of the titi shrub swamp receiving wastewater. In order to facilitate the use of the model over long periods of time, all transfers between compartments or state variables were taken as the annual net accumulation in each compartment and therefore, the cycling of nutrients and other materials on a seasonal basis was not addressed. Annual and seasonal variations in the forcing functions were also ignored. There was only one autotrophic component (biomass) and production was modeled as the interaction of an external limiting factor (a flow limited source, solar radiation) and internal limiting factors (nutrients). Therefore, production was limited by the rate of supply of the external factor and by the recycle of internal factors.

The computer program was written in BASIC and the simulations were performed on a digital computer. Integration of the differential equations occurred with an integration interval of 0.1.

Results and Discussion

The simulation model of the titi shrub swamp receiving wastewater is presented in Figure 42. The differential equations for the simulation model are presented in Table 51. Initial conditions of the storages for the simulation model are presented in Table 52. Flow rates for the simulation model are presented in Table 53. The BASIC computer program to simulate the discharge of wastewater to the titi shrub swamp is presented in Schwartz (1989). This includes transfer coefficients which were determined from storages and flows. Results of model development and simulation include material (water, carbon, nitrogen, phosphorus) and energy budgets for the titi shrub swamp. The calculations of storages and flows for the simulation model and for the material and energy budgets are presented in Schwartz (1989).

Storages of carbon, nitrogen and phosphorus in biomass, leaf litterfall, litter and soil were determined for each community type on an areal basis and presented as a single value for the titi shrub swamp. The annual water budget (Table 43) was modified to include wastewater flow and an upper limit of deep seepage (groundwater flow). Runoff was then calculated as the balance of the budget. The amount of nitrogen in the surface water was held constant, and the amount of phosphorus in the surface water was held constant until maximum phosphorus adsorption in the soil was reached. The biomass component could use 5% of the solar radiation that was available to it. The average efficiency of gross primary productivity (GPP) was 0.3% and plant respiration was 40% of GPP. All of the nitrogen and phosphorus deposited in litter was assumed to remain in the soil, and the movement of nitrogen from surface water in to the soil was assumed to be at steady state. The maximum amount of phosphorus adsorbed in the soil was a function of the adsorption maxima of the two soil types (on an areal basis) and the percent of TAMM extractable aluminum in the soil. A switch was used to prevent phosphorus adsorption in the soil when maximum adsorption was reached.

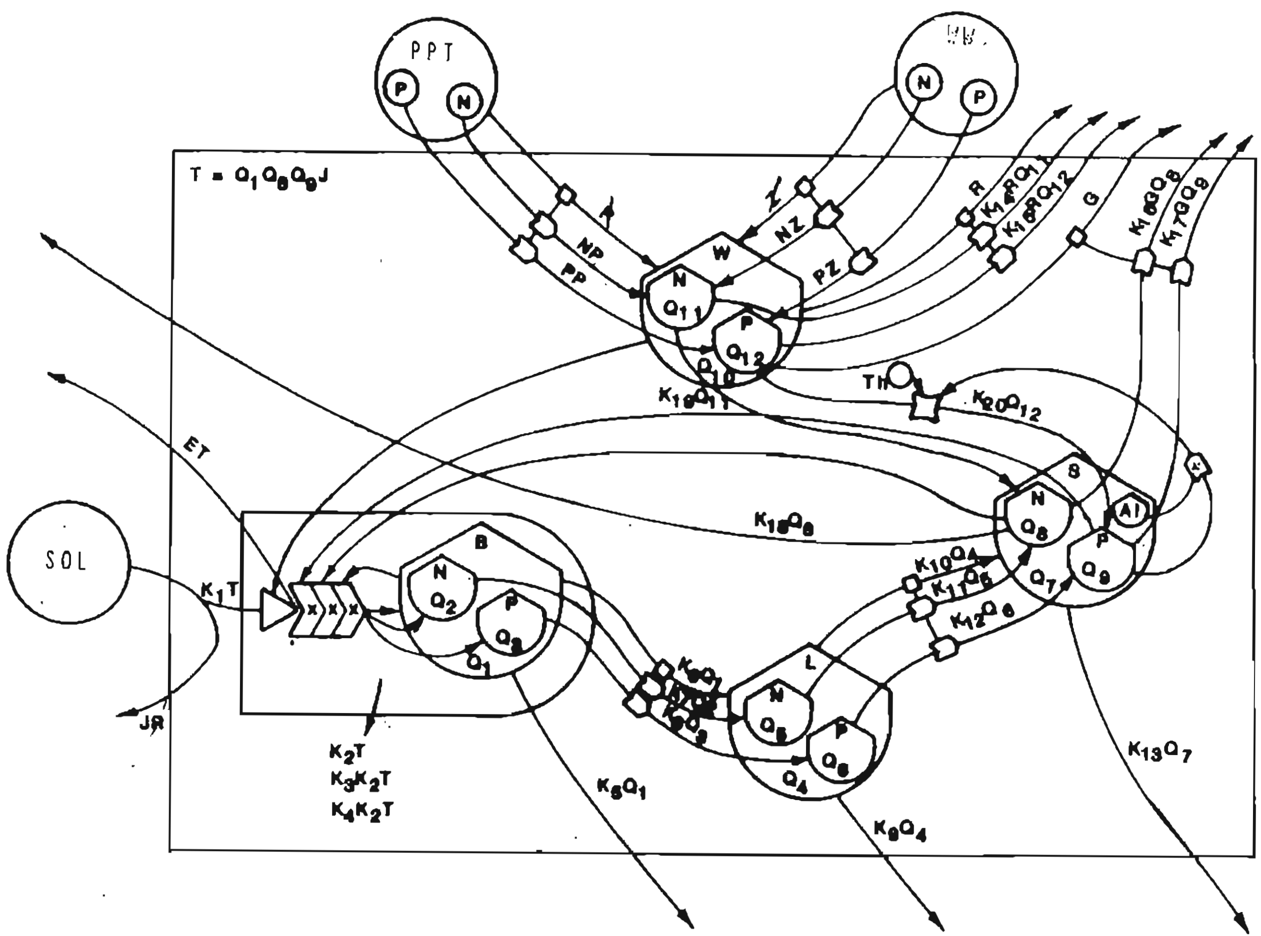


Figure 42. Systems diagram of the simulation model of the titi shrub swamp near Apalachicola, Florida. W = water, B = biomass, L = litter, S = soil, N = nitrogen, P = phosphorus.

Table 51. Differential equations for each state variable used in the simulation model of the titi shrub swamp in Apalachicola, Florida, presented in Figure 42.

$$dQ_1/dt = k_2Q_1Q_8Q_9J_r - k_5Q_1 - k_6Q_1$$

$$dQ_2/dt = k_3k_2Q_1Q_8Q_9J_r - k_7Q_2$$

$$dQ_3/dt = k_4k_2Q_1Q_8Q_9J_r - k_8Q_3$$

$$dQ_4/dt = k_6Q_1 - k_9Q_4 - k_{10}Q_4$$

$$dQ_5/dt = k_7Q_2 - k_{11}Q_5$$

$$dQ_6/dt = k_8Q_3 - k_{12}Q_6$$

$$dQ_7/dt = k_{10}Q_4 - k_{13}Q_7$$

$$dQ_8/dt = k_{11}Q_5 + k_{19}Q_{11} - k_3k_2Q_1Q_8Q_9J_r - k_{18}Q_8 - k_{16}GQ_8$$

$$dQ_9/dt = k_{12}Q_6 + k_{20}Q_{12} - k_4k_2Q_1Q_8Q_9J_r - k_{17}GQ_9$$

$$dQ_{10}/dt = P + Z - ET - G - R$$

$$dQ_{11}/dt = NP + NZ - k_{14}RQ_{11} - k_{19}Q_{11}$$

$$dQ_{12}/dt = PP + PZ - k_{15}RQ_{12} - k_{20}Q_{12}$$

Table 52. Initial conditions for the storages for the simulation model of the titi shrub swamp in Apalachicola, Florida. Sources for the values are presented in Schwartz 1989.

Storage	Description	Value	Note in Schwartz 1989
Q_1	Aboveground biomass	4.4 E4 kcal/m^2	1
Q_2	N in biomass	45.4 g N/m^2	1
Q_3	P in biomass	1.9 g P/m^2	1
Q_4	Dry weight of litter	3.1 E3 kcal/m^2	2
Q_5	N in litter	4.53 g N/m^2	2
Q_6	P in litter	0.13 g P/m^2	2
Q_7	Carbon in soil	1.99 E5 kcal/m^2	3
Q_8	N in soil	338.7 g N/m^2	3
Q_9	P in soil	15.8 g P/m^2	3
Q_{10}	Surface water	1.08 m^3/m^2	4
Q_{11}	N in surface water	1.07 g N/m^2	5
Q_{12}	P in surface water	0.01 g P/m^2	6

Table 53. Flow rates for the simulation model of the titi shrub swamp in Apalachicola, Florida. Sources of the values are given in Schwartz 1989.

Flow	Description	Value	Note in Schwartz 1989
P	Precipitation	1.67 m/yr	7
NP	N in precipitation	1.57 g N/m^2 · yr	8
PP	P in precipitation	0.08 g P/m^2 · yr	9
R	Runoff	3.98 m/yr	7
$k_{14}RQ_{11}$	N in runoff	3.30 g N/m^2 · yr	10
$k_{15}RQ_{12}$	P in runoff	0.4 g P/m^2 · yr	11
G	Groundwater flow	0.21 m/yr	7
$k_{16}GQ_8$	N in groundwater	0.28 g N/m^2 ·yr	12
$k_{17}GQ_9$	P in groundwater	0.01 g P/m^2 · yr	13
$k_1Q_1Q_8Q_9J_r$	J	7.3 E4 kcal/m^2 · yr	14
$k_2Q_1Q_8Q_9J_r$	GPP	2.78 E3 kcal/m2 · yr	14
$k_3k_2Q_1Q_8Q_9J_r$	N uptake by vegetation	2.86 g N/m^2 · yr	15
$k_4k_2Q_1Q_8Q_9J_r$	P uptake by vegetation	0.12 g P/m^2 · yr	16
k_5Q_1	Plant respiration	1.11 E3 kcal/m^2 · yr	17
k_6Q_1	Leaf litterfall	1.55 E3 kcal/m^2 · yr	18
k_7Q_2	N deposited by litterfall	2.11 g N/m^2 · yr	18
k_8Q_3	P deposited by litterfall	0.08 g P/m^2 · yr	18
k_9Q_4	Litter respiration	7.75 E2 kcal/m^2 · yr	19
$k_{10}Q_4$	Litter remaining in soil	7.75 E2 kcal/m^2 · yr	19
$k_{13}Q_7$	Soil respiration	3.88 E2 kcal/m^2 · yr	20

Table 53. continued.

Flow	Description	Value	Note in Schwartz 1989
$k_{11}Q_5$	Litter N remaining in soil	2.11 g N/m^2	21
$k_{12}Q_6$	Litter P remaining in soil	0.08 g P/m^2	21
Z	Wastewater flow	3.56 m/yr	7
NZ	N in wastewater flow	16.0 g $N/m^2 \cdot yr$	22
PZ	P in wastewater flow	7.0 g $P/m^2 \cdot yr$	23
$k_{18}Q_8$	Denitrification	2.01 g $N/m^2 \cdot yr$	24
$k_{19}Q_{11}$	Movement of N in surface water to soil	14.3 g $N/m^2 \cdot yr$	25
$k_{20}Q_{12}$	Phosphorus adsorption	6.68 g $P/m^2 \cdot yr$ up to a maximum of 96.6 g/m^2 in Q_9	26

Annual budgets were developed for water, carbon, nitrogen, and phosphorus in the titi shrub swamp prior to wastewater discharge and after 100 yr of wastewater discharge. The relative amounts of carbon, nitrogen, and phosphorus within the compartments prior to wastewater discharge were as follows: soil > biomass > litter (Figure 43). The total amounts of nitrogen and phosphorus entering the system were low. In addition, the total amounts of nitrogen and phosphorus leaving the system were also low.

The relative amounts of carbon and phosphorus within the compartments after 100 yr of wastewater discharge were as follows: soil > biomass > litter (Figure 44). The relative amounts of nitrogen within the compartments were as follows: biomass > soil > litter (Figure 44). A 19-fold increase occurred in the biomass and litter compartments and a fivefold increase occurred in the soil carbon compartment. A 26-fold increase occurred in the biomass and litter phosphorus compartments and a sixfold increase occurred in the soil phosphorus compartment. A 24-fold increase occurred in the biomass and litter nitrogen compartments and a twofold increase occurred in the soil nitrogen compartment. After 100 yr of wastewater discharge 58%, 36% and 6% of the stored nitrogen was in biomass, soil and litter, respectively. After 100 yr of wastewater discharge 33%, 65% and 2% of the stored phosphorus was in biomass, soil and litter, respectively. Although nitrogen storage was primarily in biomass, phosphorus storage was primarily in soil. At a phosphorus loading of 2 mg/l, the model predicts that phosphorus adsorption in the soil will end after 11.7 yr of wastewater discharge to the titi shrub swamp.

The plots of biomass, litter and soil carbon over time and the nitrogen and phosphorus in biomass and litter over time are limiting factor hyperbolas (Figure 45). The dynamics of nitrogen and phosphorus storage in the soil did not follow this pattern. Phosphorus was rapidly adsorbed in the soil until maximum adsorption was reached. Then the amount of phosphorus in the soil remained constant. Very little nitrogen was stored in the soil until a limit to production was reached. Then the amount of nitrogen entering the system was greater than required to maintain that level of production and storage of nitrogen in the soil increased.

Intrasystem cycling of nutrients (exchanges among the various pools or standing stocks of chemicals) in wetlands is dependent on the availability of the nutrients and the degree to which processes such as primary productivity and decomposition are controlled by the wetland environment (Mitsch and Gosselink 1986). Forested wetlands can be arranged according to the volume of water flowing to the wetland and the accompanying nutrients (Odum 1984; Brown 1981). Therefore, not all wetlands have high nutrient inputs and may not be highly productive ecosystems. Many isolated wetlands such as the wetlands at the study site have a low supply of nutrients. Increases in biomass are marginally supported by the nutrient pool. The scarcity of nutrients limits productivity in these systems as indicated in the annual budgets determined during model development (Figure 43). Wetlands in which precipitation is the primary nutrient input pathway depend on intrasystem cycling for nutrients (Mitsch and Gosselink 1986). Therefore, if primary productivity and decomposition in the wetland are limited

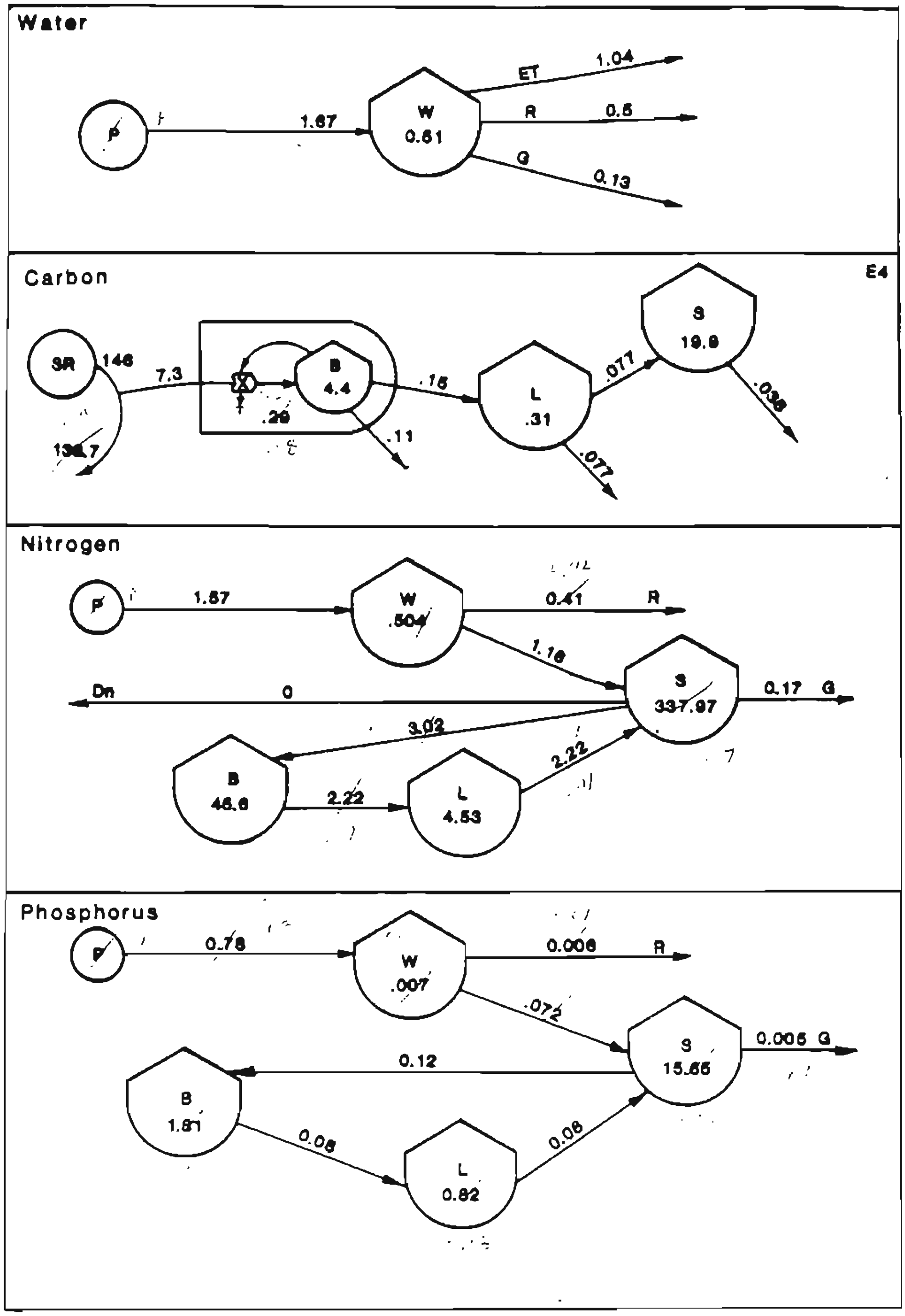


Figure 43. Material and energy budgets for the titi shrub swamp in Apalachicola, Florida. Calculation of the storages and flows presented in Schwartz (1989).

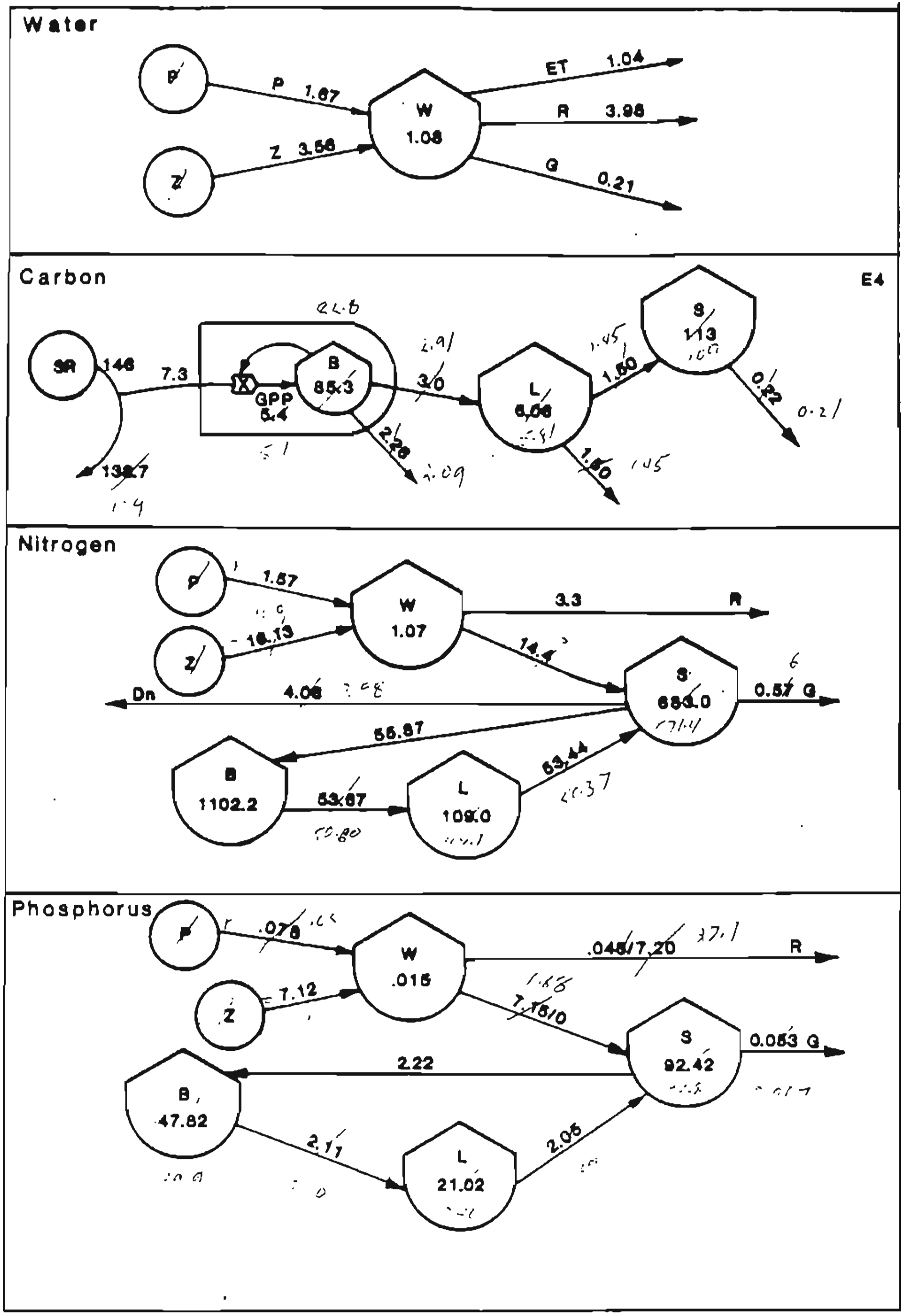


Figure 44. Material and energy budgets for the titi shrub swamp in Apalachicola, Florida, after 100 years of simulated wastewater discharge. Calculation of the storages and flows presented in Schwartz (1989).

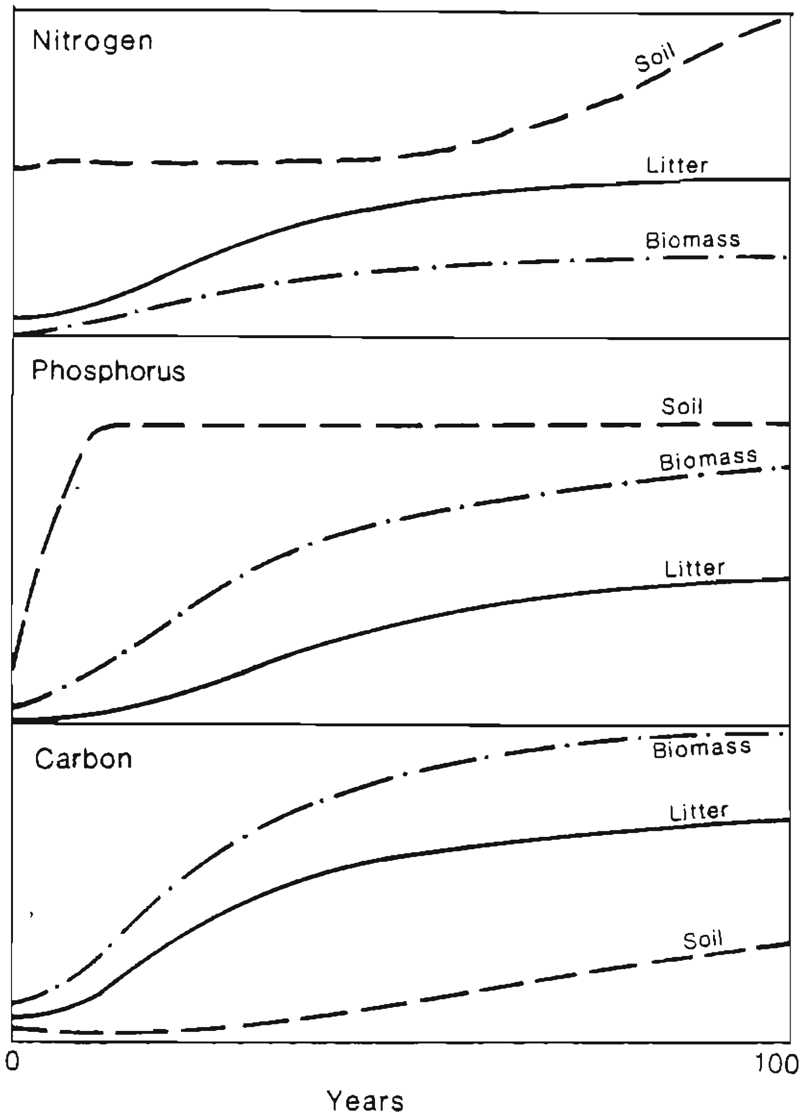


Figure 45. Results of the simulation model of the titi shrub swamp near Apalachicola, Florida, after 100 years of simulated wastewater discharge.

by hydrologic and nutrient conditions, then nutrients may be conserved. At the study site, the mean total nutrient concentrations in shallow groundwater are higher than in surface water. Therefore, the system may be acting as a sink for these nutrients. As titi shrub swamps develop they adapt to low water volume low nutrient conditions through conservation of those nutrients.

The response of the wetland to the simulated increase in nutrients was an increase in annual biomass and litter and increased storage of nutrients in biomass, litter and soil. This is consistent with the results from other models used to simulate the addition of wastewater to wetlands (Dixon and Kadlec 1975; Mitsch 1975b; Deghi and Ewel 1984; Hammer 1984) and with the results from research in which a quantitative determination was made of the storage in wetland compartments (Kadlec and Tilton 1979; Dierberg and Brezonik 1983b). Natural wetland retain and store much of their nutrient inputs even when loading increased 200-fold over natural inputs (Dierberg and Brezonik 1983b).

Without the addition of wastewater the simulated titi shrub swamp was a phosphorus limited system. The vegetation stored nitrogen and phosphorus at a N:P ratio of 25:1 (Figure 43). Wastewater was added to the system at a N:P ratio of 2.26:1 (Figure 44). Therefore, there was an overabundance of phosphorus added to the system relative to nitrogen. Production and storage in biomass increased as long as phosphorus was available to drive the process. Then, the amount of nitrogen entering the system was greater than that required to maintain the same level of production and storage of nitrogen in the soil increased. A greater rate of denitrification than that which was used in the model could account for an overall lower level of nitrogen storage in the soil. When phosphorus adsorption in the soil ended (11.7 yr) the model predicted that phosphorus discharge in runoff would greatly increase. The wetland system would no longer assimilate enough phosphorus to protect downstream receiving water quality. Wastewater with a higher N:P ratio (lower phosphorus concentration in wastewater) would increase the lifetime of the system for phosphorus assimilation and nitrogen assimilation would be accounted for through storage and denitrification. Therefore, nutrient loading criteria should be based on maximizing the longevity of the system for phosphorus assimilation.

A quantitative dynamic ecosystem response model was developed to characterize the titi shrub swamp and to predict the long-term response of the system to wastewater addition. Many of the model compartment values can now be more accurately estimated and with minimum site specific data, particularly soils analysis, the model could be used as a design and management tool for other freshwater forested wetland systems in Florida.

SUMMARY AND CONCLUSIONS

Changes in both water chemistry and macroinvertebrate composition near areas of wastewater addition were observed. These changes decreased with distance from the effluent outfall. Large standard errors were encountered with all macroinvertebrate observations, especially during post-effluent sampling.

Mean total macroinvertebrate abundance increased from less than $500/m^2$ to over $8000/m^2$ at sites near wastewater discharge. The mean at a site 900 m from the effluent outfall was just over $1000/m^2$. Decreased diversity as measured by the Shannon index was observed for sites near effluent discharge. A reduced number of samples taken may have accounted for decreased diversity at control sites during the post-effluent sampling period. Decreased evenness (Odum 1971), decreased species richness as calculated by the equation $D1 = S/\sqrt{T}$ (Odum 1971), where S is the number of species and T is the total number of individuals encountered, and increased dominance (Odum 1971) was observed at wastewater impacted sites.

Increases in abundance were predominant for certain groups of invertebrates, namely those classified in the collector feeding guild, whose primary function is to break down decomposing organic matter (Merritt and Cummins 1978). Chironomidae comprised the largest percentage of this feeding guild, especially the genera *Chironomus* and *Kiefferulus*. Coleoptera of the family Hydrophilidae (larvae) serve a similar function and also increased. The Crustaceans *Asellus*, *Crangonys*, and *Procambarus*, common residents of the wetland before the addition of wastewater, were eliminated at sites affected by wastewater. At downstream sites where only small changes were observed in water quality and vegetation, Crustaceans were eliminated, indicating that they are a group extremely sensitive to changes in water quality. Chironomidae became more common at these downstream sites.

Increased concentrations of most water chemistry parameters were observed as a result of wastewater addition. At the area of wastewater discharge, mean total phosphorus increased from less than 0.01 to 2.1 mg/l, mean total nitrogen increased from 1.12 to 3.44 mg/l, mean biological oxygen demand increased from 3.12 to 4.63 mg/l, mean pH increased from 3.7 to 6.6, mean dissolved oxygen increased from 3.96 to 6.77 mg/l, mean conductivity increased from 93 to 1075 µmho/cm, and mean chloride increased from 7.65 to 220.25 mg/l. Changes in water chemistry parameters were less significant at downstream sites.

Hurricanes could have played a role in forcing large quantities of wastewater to downstream sites, although the concentrations may not have been elevated for a long enough period of time to be detected by the water chemistry sampling program. A considerable amount of variation in monthly water chemistry values was observed, which could have been integrated by the macroinvertebrate community.

Nocturnal differences in dissolved oxygen are often the critical factor which determines the distribution of organisms. These differences in nocturnal dissolved oxygen are sometimes concomitant with supersaturation of dissolved oxygen in the daytime (Schwartz 1980). Anoxic conditions at sites D25, D200, and D400 were evident by the smell of hydrogen sulfide upon disturbance of the sediment. Chironomidae of the genera *Chironomus* and *Kiefferulus* are particularly adapted to life at low oxygen concentrations, which explains why their numbers increased greatly at these sites.

Changes in ground cover vegetation may have also played a role in invertebrate response. The gradual decline of *Sphagnum* created a large source of detritus for members of the collector feeding guild. The covering of *Lemna* at sites D25, D200, and D400 was so complete that all light was probably cut off from the water column, resulting in a decrease in pelagic algal photosynthesis and decreasing oxygen availability. Epiphytic algal production was enhanced at sites not covered by *Lemna* or emergent macrophytes. This would tend to increase the oxygen supply in the water column at these sites during daylight hours.

When calculating diversity indices, the ecological value of using varying levels of taxonomic precision must be examined. Identification of organisms to species may be unnecessary for detecting intersite differences in diversity value due to the existence of a similar graphical pattern between the order diversity and the species diversity (Hughes 1978; Osborne *et al.* 1980). Osborne (1977) reported a significant difference in diversity between sites and concluded that it would only have been necessary to identify organisms to the family level to detect intersite diversity differences. Seasonal variations between taxonomic levels may counteract the effect of one level on the total (Osborne *et al.* 1980). The higher the taxonomic level employed, the smaller the diversity values obtained (Osborne *et al.* 1980). It would thus be possible to get a significant difference in species diversity between sites, while with the same data using a higher taxonomic level, significant differences need not result. Osborne *et al.* (1980) advocate the use of a hierarchical diversity index which considers the effects of combining different taxonomic units in the same sample.

Although the mean water chemistry parameters monitored indicated a return to background levels at downstream sites, an invertebrate response was still evident. It is important to note that the samples were taken during the year immediately following input of wastewater, which is not necessarily indicative of the eventual system response in the long term (H. T. Odum, personal communication). A gradual leveling off in populations with time, following an initial increase in abundance, can be hypothesized for near-discharge sites. Ecological factors including predation by small fish and frogs, which were extremely abundant at sites D25, D200, and D400, may have cause this type of trend. Competition probably played a minor role since the food supply was so plentiful in this system. Alternatively, the patterns of invertebrate abundance observed may be due to seasonal responses of the organisms.

It is difficult to determine accurate background levels of benthic macroinvertebrates, since so much natural variation exists in the samples. The factors influencing this variation have been discussed. For this reason, background sampling of macroinvertebrates may not be as indicative of the undisturbed natural system as properly selected and sampled control sites. The control sites may be located in an upstream area with similar vegetation and water chemistry characteristics, as in this study, or in a neighboring system with the same qualifications. This type of sampling, during the same time period, may more accurately simulate the natural environmental conditions of the disturbed sites than will background sampling, at the actual sites which will be impacted, in a different time period.

The use of benthic macroinvertebrates has been shown in this study to be a qualitative index of water quality conditions. Many smaller communities have been examining the feasibility of using neighboring wetland systems as a low-cost wastewater treatment alternative. Monitoring costs for water chemistry parameters may often exceed the capability of these small wastewater treatment systems. Benthic macroinvertebrate analyses may be an alternative for small systems such as these.

The role of wetlands as a treatment system for municipal wastewater has been widely discussed (Dierberg and Brezonik 1983; Dolan *et al.* 1981; Hyde *et al.* 1984; Kadlec 1978; Kadlec and Tilton 1979; Ryden and Pratt 1980; Tilton and Kadlec 1979; Vega and Ewel 1981). Data on wetland efficiency at removal of nutrients are reported in a variety of forms (percent concentration reduction, output mass versus background output mass, output concentration versus background output concentration, percent retained, concentration reduction versus time, concentration reduction versus distance), so comparisons of different studies are difficult unless the data are converted to a standardized format. Performance data on wetland treatment systems (reported as percent concentration reduction) range from 6% to 98% for TP and from 30% to 96% for TN (Hyde *et al.* 1984). However, about 70% of these concentration reductions may be due to dilution alone (Kadlec 1983). Because of the site-specific nature of wetland wastewater treatment, even standardized comparisons are of little value unless mechanisms of nutrient removal can be linked to specific measureable factors, such as soil type (especially clay content), water depth, water velocity, and residence time.

In conclusion, evaluating how long the wetland treatment system at Apalachicola will continue to perform effectively as a tertiary wastewater treatment alternative will require more follow up monitoring over the next several years. Although wastewater disposal to wetlands has not gained public or political acceptance, the practice is much less environmentally damaging than many alternatives.

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