COMPARATIVE ECOSYSTEM ANALYSIS OF HYDROLOGIC RESTORATION OF TATES HELL SWAMP

Northwest Florida Water Management District

Florida Department of Environmental Protection Contract WM 691

COMPARATIVE ECOSYSTEM ANALYSIS OF HYDROLOGIC RESTORATION OF TATES HELL SWAMP

DEP Contract WM 691

June 2000

Northwest Florida Water Management District

in cooperation with

University of Florida Center for Wetlands (CFW)
Florida State University Department of Biology
Florida Department of Environmental Protection (DEP)
U.S. Environmental Protection Agency (EPA)
Florida Department of Agriculture and Consumer Services (DACS)
Florida Division of Forestry (DOF)

This project and the preparation of this report was funded in part by a Section 319 Nonpoint Source Management Program grant from the U.S. Environmental Protection Agency (USEPA) through a contract with the Stormwater/Nonpoint Source Management Section of the Florida Department of Environmental Protection. The total cost of the project was $470,000 of which $225,000 was provided by the USEPA.

NORTHWEST FLORIDA WATER MANAGEMENT DISTRICT

GOVERNING BOARD

Charles W. Roberts, Chairman
Tallahassee

Joyce Estes, Vice Chairman
Eastpoint

Judy Byrne Riley , Secretary/Treasurer
Fort Walton Beach

Nancyann M. Stuparich
Pensacola

L. E. McMullian, Jr.
Sneads

Wayne Bodie
DeFuniak Springs

John R. Middlemas, Jr.
Panama City

J. Russell Price
Tallahassee

Sharon T. Gaskin
Wewahitchka

Douglas E. Barr, Executive Director

For additional information, write or call:

Northwest Florida Water Management District
81 Water Management Drive
Havana, Florida 32333
(850) 539-5999; Suncom 771-2080
FAX #: (850) 539-4380 (Main Bldg.)
FAX #: (850) 539-4379 (SWIM Bldg.)

ACKNOWLEDGEMENTS

This project and the preparation of this report was funded in part by a Section 319 Nonpoint Source Management Program grant from the U. S. Environmental Protection Agency (USEPA), through a contract with the Stormwater/Nonpoint Source Management Section of the Florida Department of Environmental Protection. Howard Marshall of the USEPA was instrumental in directing the project funding. Eric Livingston of the Stormwater/Nonpoint Source Management Section of FDEP provided the vision at the state level to perceive the importance of Tates Hell Swamp to Apalachicola Bay and to appropriate the necessary funding. Michael Scheinkman (project manager for FDEP) offered valuable guidance throughout the project.

Biological work, and much of the writing of this report were done by Chris Roberts and Dr. Thomas Crisman of the University of Florida Center for Wetlands. Plant community work was conducted by Dr. Loran Anderson of the Florida State University Department of Biology.

Carolyn Kindell of the Florida Natural Areas Inventory (FNAI) provided many valuable insights into the historic characteristics of the study area and expected responses to restoration.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF FIGURES (Continued)

LIST OF FIGURES (Continued)

LIST OF FIGURES (Continued)

LIST OF FIGURES (Continued)

LIST OF TABLES

EXECUTIVE SUMMARY

In 1998 the Northwest Florida Water Management District (NWFWMD), in cooperation with the Florida Department of Environmental Protection (DEP), the Florida Division of Forestry (DOF), and the Florida Game and Fresh Water Fish Commission (FGFWFC) completed a hydrologic restoration demonstration project in Tates Hell Swamp in the Florida Panhandle. A major portion of this project consisted of restoring a 3,000-acre tract located in the Big Slough Branch sub-basin. NWFWMD was awarded $225,000 in EPA Section 319 funds to establish baseline ecosystem status and evaluate initial ecosystem response hydrologic restoration of this tract.

Monitoring was conducted beginning three months prior to restoration and continued for a total of eighteen months. Hydrologic and water quality monitoring was conducted by NWFWMD. Biological and additional water quality monitoring were conducted under subcontract by the University of Florida Center for Wetlands. Plant community analysis was conducted under separate subcontract by the Florida State University Department of Biology.

Hydrologic monitoring established that restoration efforts were successful in raising and stabilizing water levels in the demonstration site. Wetland hydroperiods in the demonstration site have been significantly increased over those observed at the topographically similar control site. No consistent water quality response to restoration was observed. Water quality was excellent at all three sites, both prior to and after restoration.

Over one hundred macroinvertebrate taxa, thirteen fish species, and more than three hundred plants were identified in this biologically rich and interesting area. Comprehensive baseline data was collected for benthic macroinvertebrate and zooplankton communities but no consistent response of these groups to hydrologic restoration was seen. A noteworthy finding was that corer sampling of macroinvertebrates yielded more consistent results than the more commonly employed sweep-net sampling method, suggesting that corer sampling may be appropriate for use in developing a wetland condition index.

Tentative evidence was observed for increased post-restoration use of wetlands by the fish species *Lepomis gulosus* (warmouth), as well as evidence of a species shift from pygmy killifish (*Leptolucania ommata*) to mosquitofish (*Gambusia holbrooki*). Drought conditions during pre-restoration monitoring and erratic rainfall throughout the study prevented definitive comparisons between the pre- and post-restoration conditions, and rendered comparisons among sites difficult. Longer-term studies capable of addressing inter-annual variability will be necessary to firmly establish the effects of restoration. Significant positive long-term biological responses to hydrologic restoration combined with a comprehensive management plan are expected.

INTRODUCTION

Over the past 200 years, more than 53% of the total area of wetlands in the continental United States have been lost, including over 46% of historical coverage in Florida (Mitsch and Gosselink 1993). Wetland loss in Florida represents over 10% of the U.S. total. Large expanses of wetlands still exist in the state, including the Everglades, Okefenokee Swamp, Tates Hell Swamp, Mallory Swamp, Gulf Hammock, Green Swamp, Big Cypress Swamp, and numerous riparian wetlands, salt marshes, and mangrove swamps. However, many of the remaining wetlands have been altered by humans and no longer resemble pre-Columbian conditions.

Tates Hell Swamp extends over approximately 200,000 acres of lowlands in Franklin and Liberty counties in the Florida Panhandle (Figure 1-1). The area was originally dominated by a diversity of wetland types, including wet savanna, cypress strand, and hardwood swamp. These wetlands have historically supported—and to a limited extent continue to support—a variety of rare plants, animals, and natural communities. The western portion of Tates Hell drains to East Bay, the primary nursery area for Apalachicola Bay.

Starting in the 1960s and 1970s, the hydrology of Tates Hell was altered by an extensive network of access roads and associated ditches constructed for the purpose of establishing pine plantations. Excavated fill from either side of the proposed roadway was used to establish routes across the low, poorly drained terrain. Excavations on either side of the roads aided in draining the land, thereby enhancing pine production potential. This ditching and subsequent draining has significantly lowered the water table, resulting in extensive loss of wetland habitat and alteration of wetland community structure. These alterations have adversely impacted water quality in East Bay by reducing storage and disturbing freshwater delivery patterns. The road-ditch system, in conjunction with silvicultural operations in the area, results in intense pulses of turbid, low pH runoff reaching the estuary following substantial rainfall events.

Efforts to stem the further alteration of Tates Hell Swamp were initiated in 1992. A 30,000-acre parcel was cooperatively acquired by the Northwest Florida Water Management District (NWFWMD), the Department of Environmental Protection (DEP), and the Florida Game and Fresh Water Fish Commission (FGFWFC) in 1994. Further acquisitions have brought over 150,000 acres of Tates Hell into public ownership.

In 1994, NWFWMD was awarded a grant from the EPA through the Florida DEP for a hydrologic restoration and BMP demonstration project in Tates Hell Swamp. Two areas, each approximately 3,000 acres, were selected for restoration—Big Slough Branch in the headwaters of Whiskey George Creek, a major tributary of East Bay; and Sand Beach/Blounts Bay, which contains approximately three miles of East Bay shoreline (Figure 1-1). Goals of the project were to initiate and implement nonpoint source pollution control strategies to protect and restore the natural watershed functions and the

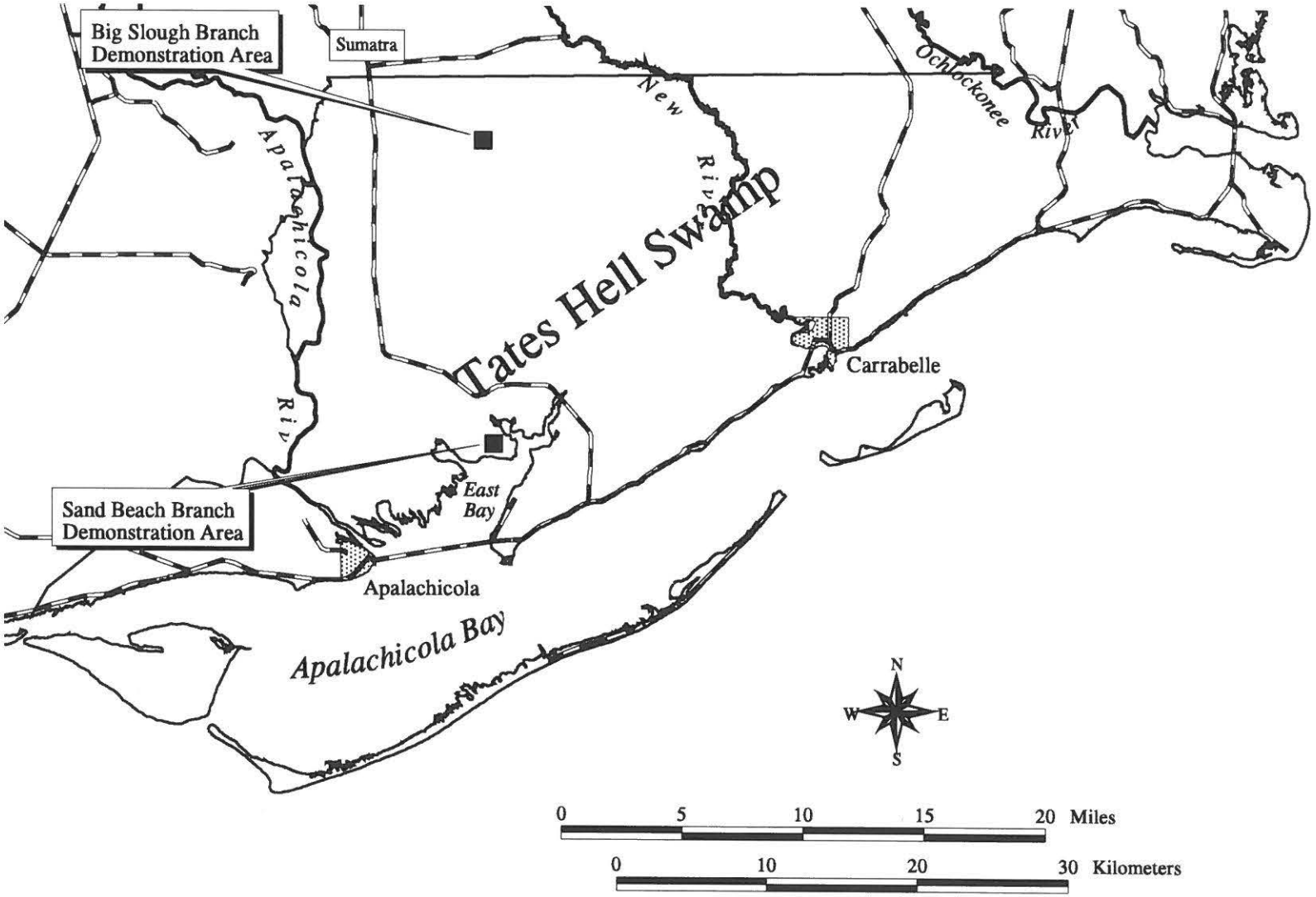
water quality of the East Bay drainage basin, and to restore the natural hydrology and wetland habitat of portions of Tates Hell Swamp. To accomplish these goals, 16 low water crossings (LWCs; segments of roads lowered to natural grade to re-establish natural flow patterns) and associated ditch plugs were installed at the two sites. In addition, four miles of roadside ditches were partially filled in order to restrict flow and redirect it to natural drainages. Numerous culverts were also plugged in order to facilitate the diversion of water into natural drainages.

Construction of the demonstration project was completed in the summer of 1998, and a final project report was submitted the following year (NWFWMD 1999). In addition to construction details, the report presented the results of the limited hydrologic and water quality monitoring component of the project. In 1997, NWFWMD was awarded $225,000 in additional EPA Section 319 funding (through DEP) for a separate, comprehensive ecological assessment of the Big Slough Branch demonstration site—the work presented in this report. This study was designed to evaluate the effects of restoration work by collecting pre- and post-restoration data at the demonstration site, as well as at nearby control (unrestored) and reference (relatively pristine) sites. Objectives were:

1) to monitor changes in hydrology and wetland community structure to determine effects of restoration on system ecology,
2) to measure changes in ecosystem function during hydrologic restoration using physical, chemical and biological water quality data collected from comparative restored, unrestored and reference sites, and
3) to establish standards of plant community response to hydrologic restoration.

An integrated hydrologic, water quality, and biological monitoring program was developed to accomplish these goals. Biological monitoring was performed under subcontracts with the University of Florida Center for Wetlands (CFW) and the Florida State University (FSU) Department of Biology. CFW examined fish, zooplankton, benthic macroinvertebrates, algae, and field-measurable water quality parameters. Vascular plant monitoring was done by FSU. NWFWMD performed hydrologic monitoring, additional water quality sampling, land surveying, and soil analysis. Data collection began in April 1998, two months prior to initiation of restoration work at the Big Slough Branch site, and continued through November 1999.

Figure 1-1. Location of Tates Hell Swamp.



METHODS

Study Organization

Site Selection

This study focused on the Big Slough Branch restoration demonstration site near the headwaters of Whiskey George Creek (Figure 2-1). This site was selected in preference to the Sand Beach Branch demonstration site because the level of disturbance at Big Slough Branch has been much greater than at Sand Beach Branch. Furthermore, the Big Slough Branch site contains significant areas of dwarf cypress swamp and remnant wet savanna habitats, both of which are of considerable ecological interest, and can be considered signature habitats of Tates Hell (Kindell 1997).

In order to help distinguish the effects of restoration from random variation, a relatively pristine (reference) site in a nearby area of the Apalachicola National Forest and a disturbed but non-restored (control) site in Tates Hell State Forest were also incorporated into the study (Figure 2-1). Every effort was made to select reference and control sites as similar as possible to the demonstration site in terms of original hydrology, historic plant community types, and soils.

The reference site is located in the headwaters of Fort Gadsden Creek, approximately six miles southwest of the demonstration site. The site is dominated by cypress swamp and wet savanna. Cypress at this site are somewhat stunted, but are larger than those in the demonstration site. While this site has been subjected to minor ditching, the level of disturbance is much less than at the demonstration site. The reference site has been managed by the United States Forest Service (USFS) to promote native plant communities.

The control site is located approximately two miles west of the demonstration site. The site is dominated by dwarf cypress-dwarf blackgum swamp with limited acreage of wet savanna. Both the control and pre-restoration demonstration sites were intersected by deep drainage ditches, and the two sites have undergone similar levels of hydrologic disturbance.

Biological, water quality, and hydrologic monitoring stations were established in all three of these sites. Exact distribution of stations varied for different types of sampling (Figure 2-1).

Biological and associated water quality sampling was conducted by CFW at three stations in the demonstration site (1, 2, and 3, Figure 2-1), and one station each in the control and reference sites (C and R, Figure 2-1). Sampling stations were selected to represent long-hydroperiod, relatively open, cypress-dominated wetlands. Stations in the demonstration and control sites were located where road-ditch systems cross broad natural cypress

swamp drainageways, while the reference station was placed in a similar unimpacted drainageway. Low water crossings were installed at the demonstration site stations during the course of the study.

Sampling of chemical water quality parameters was conducted by NWFWMD at four stations in the demonstration site (S533, S534, S537, S541), as well as upstream and downstream stations in the control (S538, S505) and reference (S539, S540) sites. Stations S537 and S541 in the northern (upstream) portions of the demonstration site correspond to biological sampling stations, while stations S533 and S534 at the lower end of the demonstration site do not. The latter stations were established to examine water quality leaving the site and to determine if the site functions as either a source or sink for nutrients.

Continuous stage monitoring equipment was installed at the lower end of the demonstration site (S533), an additional location in the interior of the demonstration site (S536), and at the downstream end of the control site (S505). Equipment was concealed in dense vegetation (primarily titi) in order to avoid vandalism. The NWFWMD had previously lost several thousand dollars worth of equipment to vandalism in this area. The biological and water quality sampling sites further north in the demonstration site, as well as the station in the reference site, lacked sufficiently dense vegetation in which to conceal stage measuring equipment, and equipment was therefore not installed at these locations.

Arrays of ground water monitoring wells were installed at locations G1 and G2 (Figure 2-1). The G1 array had been installed during an earlier study (NWFWMD 1999), and serves as a control ground water elevation site. The G2 array was installed during the present study, and reflects ground water conditions at a location in the demonstration site adjacent to partially filled drainage ditches.

A vascular plant survey was conducted throughout the demonstration and reference sites. Permanent intensive plant quadrats were established at locations near low water crossings at biological sampling stations 1 and 3. Specific quadrat locations were chosen to examine both dwarf cypress-dwarf blackgum swamp and wet savanna habitats.

Monitoring Schedule

Due to a variety of legal, administrative, and technical delays (NWFWMD 1999), monitoring did not begin until April 1998. Restoration construction activities on the demonstration site began in May of that year, and were essentially complete by late July. Thus, pre-restoration data was collected for only three months.

Biological sampling was conducted on approximately a 28-day cycle beginning April 1998 and continuing through October 1999. Dry conditions interfered with sampling on several scheduled collection dates, especially during the summer of 1998. However,

limited sampling (ditches, wetland benthic infauna) was conducted even during dry periods, and the resulting data set consists of 19 sampling events.

Water quality sampling was scheduled monthly from April 1998 through November 1999. Dry conditions prevented sampling on several occasions—samples were only collected during months when water was observed in natural drainageways. In order to assess storm runoff water quality, double samples were collected at short (one or two day) intervals following two major rainfall events. A total of 13 sampling events were conducted.

Continuous stage and rainfall data were collected from April 1998 through October 1999. Ground water elevations were measured on six occasions between March and November 1999. Soil samples were collected and intensive plant quadrats established and sampled in October 1999.

Hydrology

Continuous surface water stage data were collected with Handar 555 data loggers and Druck pressure transducers installed at the three locations indicated in Figure 2-1. Transducers were placed in ditches in order to monitor water levels lower than natural grade. Stations S533 and S505 are located at the outflows of the demonstration and control sites, respectively. Road/ditch systems intersect broad, relatively well defined natural flow-ways at both of these stations. Detailed cross-sectional elevation surveys of both drainageways were conducted for the purpose of examining the relationship between stage and wetland elevation. Station S533 is adjacent to LWC 7 on Gully Branch Road. Prior to restoration, a deep ditch on the north side of the road intercepted water flowing south in Big Slough Branch, diverting flow to the west. Subsequent to LWC construction, plugging of culverts, and partial backfilling of the ditch system, flow was restored to the natural flow path. At station S505, the ditch at West Boundary Road intercepts eastward flow from Hog Branch, diverting it southward. Conditions at the control site station are thus analogous to pre-restoration conditions at station S533. A third station, S536, represents the interior of the demonstration site. Both stage and rainfall (Handar tipping bucket gage) were monitored at this station.

An array of ground water piezometers had been established prior to this study at G1, a short distance west of the demonstration site on Tower Road (Figure 2-1). This array, which consists of one deep (12 ft) and seven shallow (5-6 ft) wells distributed from 0 to 170 ft from the roadside drainage ditch, served as a control in the present study. Another array, G2, was installed along North Boundary Road in the demonstration site in late 1998. This array consists of three deep (8 ft) and four shallow (4 ft) wells ranging from 15 to 300 ft from the roadside ditch. In both arrays, deep wells penetrate a low permeability clay loam stratum found through much of Tates Hell at depths varying from 3 to 7 feet. All piezometers were of two-inch PVC, with the lowest two feet consisting of slotted well screen, and all were grouted with bentonite (control site) or neat cement

(demonstration site). Piezometers were measured approximately bimonthly from March 1999 through October 1999.

Water Quality

Grab samples were collected at each of the water quality stations identified in Figure 2-1. Sampling was conducted monthly when there was water in natural wetlands, and on two occasions, in September 1998 and January 1999, paired sampling events were conducted following storm events in order to assess runoff quality. In locations with low water crossings, samples were collected at the upstream side of the LWC where it intersects the natural wetland channel. In locations without LWCs, samples were taken in the natural wetland channel. Samples were transported to the DEP Central Laboratory and analyzed for ammonium, nitrate-nitrite, total Kjeldahl nitrogen, total organic carbon, total phosphorus, ortho-phosphorus (field-filtered), and total suspended solids. Chlorophyll a analysis was performed for the first few sampling events, but was discontinued after it was found that all samples were at or below the DEP minimum detection limit of 1.0 µg/L.

Field water quality parameters were measured in conjunction with monthly biological sampling events. Measurements were taken adjacent to LWCs or, where LWCs were absent, adjacent to wetland channels. Temperature, dissolved oxygen, and specific conductance were measured with a YSI model 85 meter. One measurement of specific conductance was taken from the surface of each ditch and wetland, and in the absence of standing water, no measurement was taken. Oxygen and temperature were taken from the surface and bottom of each ditch and the surface of each wetland if standing water was present. A Fisher Scientific Accumet AP63 pH/mV/ion meter was used to measure pH. One measurement was taken from the surface of each ditch and wetland when standing water was present.

Benthic Macroinvertebrates

For core sampling, a stainless steel cylindrical corer 7.1 cm in diameter and 26.5 cm in length with a 64 cm attached handle was used. Sampling occurred every 28 days for the first three months (beginning in April 1998) prior to and three months after restoration and bimonthly after these initial six months through October 1999. Cores were collected regardless of whether there was standing water in the wetlands. Five cores were taken to a depth of approximately 15 cm in each wetland along a randomly chosen (0-180° from line perpendicular to road) 20 m transect. Water depths were taken at each coring location along the transect. Each core was deposited in a sieve bucket with 600 µm mesh (U.S. Standard no. 30) and rinsed in the field. The contents of the sieve bucket were then transferred to soil bags, the bags tied shut, and placed into a bucket containing 70% ethyl alcohol and the vital stain Rose Bengal.

Sweep nets with a mouth diameter of 20.2 cm and a net mesh of 800 x 900 μm were also used to sample benthos. Sampling occurred every 28 days for the first three months (beginning in April 1998) prior to and three months after restoration and bimonthly after these initial six months through October 1999. Sweep nets were utilized only when there was standing water in a wetland. Ditches were sampled bimonthly. Sampling consisted of five 0.5 meter sweeps through the bottom sediments in each ditch and wetland. These samples were then pooled and deposited into a sieve bucket and washed as done for cores. The contents of the sieve bucket were placed into individually labeled soil bags indicating the wetland or ditch from which they were sampled, the bags tied shut, and were placed into a bucket containing 70% ethyl alcohol and the vital stain Rose Bengal.

In the laboratory, the contents of each soil bag (cores or sweep) were washed with tap water and placed into white-bottomed trays for sorting. Macroinvertebrates were separated from the substrate, identified to order, and placed into separate, labeled one dram vials for further identification. For samples with large numbers of invertebrates, subsamples were taken. Soil bag contents were placed onto a gridded tray, and a square was randomly selected to be picked. Invertebrates were counted and placed into vials containing 70% ethyl alcohol. This process was repeated until at least 100 specimens were placed into a vial.

Each sample was sorted twice to ensure complete isolation of all macroinvertebrates from the substrate. As part of quality control, every tenth sample was resorted to check the efficiency of the sorters. Furthermore, new pickers had their samples routinely repicked until they were acceptably efficient.

Identification primarily took place under a 4.5x (with 10x oculars) Meiji stereoscope, though chironomid and ceratopogonid dipterans were mounted in CMC-10 mounting medium on a clean glass slide and viewed under a Fisher Scientific Micromaster CK compound microscope (4x to 100x with 10x oculars). Organisms were identified to the lowest practical taxonomic level using the most relevant taxonomic references, including: Pennak (1989), Daigle (1991), Thorp (1991), Daigle (1992), Epler (1995), Pescador (1995), Epler (1996), and Merritt (1996). Quality control consisted of a second qualified person re-identifying the contents of every fifth sample to check the accuracy of identifications. A reference collection was also maintained, and indentifications were verified by Dr. David Evans of Water and Air Research in Gainesville Florida.

Zooplankton

Zooplankton were collected using a U.S. Standard No. 20 (64 μm) Wisconsin plankton net with a mouth width of 10 cm towed over a horizontal distance of 2.5 m. Samples were collected monthly beginning in May 1998 through October 1999. Zooplankton were not collected in wetlands when the depth of the water was less than 10 cm, the diameter of the mouth of the plankton net. At the completion of each tow, the sides of the plankton net were washed with water into the receptacle bottle, and contents of the bottle were transferred to individually labeled Nalgene storage bottles, fixed with

Lugol's iodine solution, and placed on ice. Bottles were refrigerated in the laboratory prior to identification.

Zooplankton were counted and identified to the lowest practical taxonomic level using the most relevant identification manuals: Pennak (1989) and Thorp (1991). For identification, samples were filtered through a U.S. Standard No. 230 sieve (63 μm), washed with tap water, and then washed into a plankton wheel to aid in counting. Identification primarily took place under a Nikon SMZ-10 stereoscope (4.5x with 10x oculars), though finer morphological characters could only be viewed under a Fisher Scientific Micromaster Model CK compound microscope (4x to 100x with 10x oculars). In the case of samples with large numbers of organisms, a subsample of one quarter of the total was taken, and the contents counted and identified until 200 individuals were counted. If the quarter subsample did not contain at least 200 individuals, another quarter subsample was counted and identified. This procedure was continued until at least 200 total individuals were counted. The final number of each zooplankton taxon was divided by the fraction of the total subsampled to calculate a total for each taxon. Once counted and identified, samples were returned to the bottles and archived in a refrigerator.

Fish

Fish were collected monthly using standard galvanized steel minnow traps covered with window screening to reduce escape by smaller fishes through the mesh sides of the trap. The minnow traps measured 41.5 cm x 22.5 cm, had a 2.2 cm diameter funnel opening, and were constructed from 0.63 cm square mesh. Each month, five traps were baited with bread and randomly placed in each ditch, wetland, and low water crossing for a 24-hour period. Fish traps were placed a minimum of ten meters apart to decrease sampling bias. Traps were placed in the water so the funnel entrances were just below the water surface and about a third of the trap was above the water surface. Traps were tied with a rope to a tree branch to hold them in place. Tests prior to the project showed more fish were collected with the traps set in this manner as opposed to completely submerging them. Exposing the top third of the traps to the atmosphere also allowed fish to gain access to the surface so that they could engage in aquatic surface respiration (ASR) in the event of hypoxic conditions developing in the water. Those wetlands not having sufficient water to allow the entrances to the traps to be submerged completely were not sampled for fish that month. Each trap was removed from the water and the contents placed into a five-liter reclosable, clear plastic bag containing approximately 0.20 L of ditch water. Each fish was identified to species, measured total length to the nearest millimeter, and recorded. Once identified and measured, the fishes were returned unharmed to the water from which they were collected.

Community Metabolism

Diel dissolved oxygen curves were obtained using a YSI Model 600 probe and recorded on a Campbell Scientific CR500 Basic Data Logger. Temperature was recorded

simultaneously. At the end of the 24-hour data collection, all data were downloaded from the data logger to a notebook computer. Data were collected each month from the control site ditch and from the upstream ditch adjacent to the LWC at biological monitoring station 1 in the demonstration site. Ditches at these locations functioned as quasi-natural stream channels, having natural bottom substrates, and collecting flow from natural wetland drainageways.

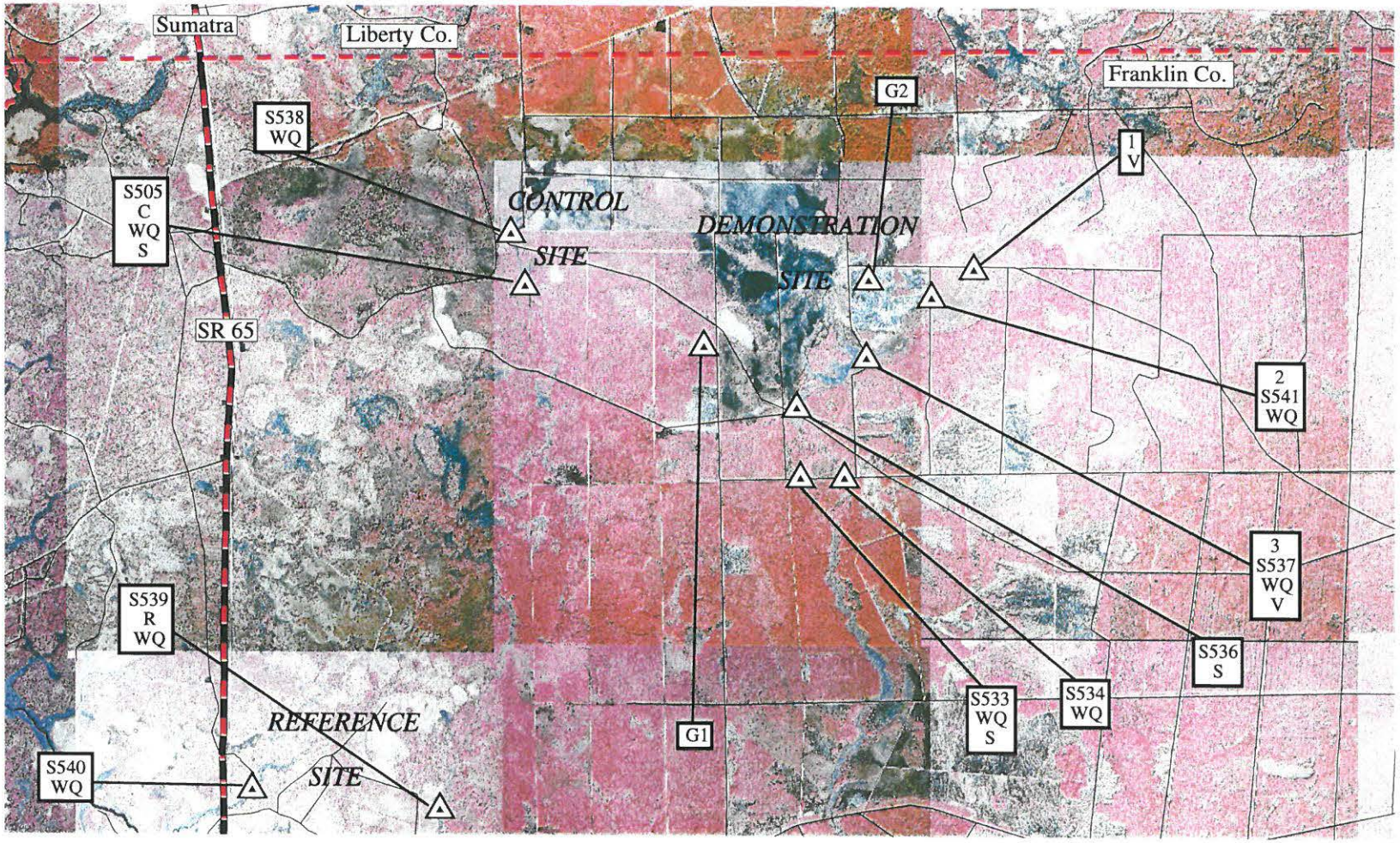
Metabolism was calculated by the single point diel oxygen curve method of Greeson (1985). Changes in dissolved oxygen were calculated over fixed 30-minute time increments for a 24-hour period. Respiration was calculated from the rate of DO decline during darkness, when productivity was zero. Gross primary productivity (GPP) was calculated from the rate of daytime DO increase, adjusted for daytime respiration, which was estimated by extrapolation from nighttime respiration values. Respiration and productivity values calculated in this way were then adjusted for oxygen exchange with the atmosphere. A literature-based oxygen exchange estimate of 0.05 g/m^2-hr at zero percent saturation, corrected for temperature, was used (Odum 1956). Net primary productivity (NPP) was calculated by subtracting adjusted respiration from adjusted GPP.

Periphyton samples were collected monthly at each biological sampling station (Figure 2-1) using an artificial substrate sampler modified from Patrick (1954). This sampler consisted of glass slides suspended vertically just under the surface of the water. The incubation period for samplers was one month. Upon retrieval, slides were placed into slide holders, covered with deionized water, and placed on ice for return to the laboratory. In the laboratory, standard method 10300 C (Greenberg et al. 1992) was used to extract chlorophyll-a.

Vascular Plants

Four intensive plant monitoring quadrats, each measuring 2 x 15 meters, were established adjacent to two LWCs in the demonstration site (Figure 2-1). Quadrat positions were established with GPS, and quadrat corners were marked with iron or PVC stakes. All plants within each quadrat were identified in October 1999, and percent cover was estimated for each species. General plant species inventories were conducted for the demonstration and reference sites through 1998 and 1999.

Figure 2-1. Location of sampling stations in the Big Slough Branch demonstration site, control site, and reference site.



HYDROLOGY

Any ecological benefits of the demonstration project are contingent on successful re-establishment of natural wetland hydrology. In its natural state, much of Tates Hell consisted of a network of broad, shallow, low-gradient drainageways with extensive fringing wetlands. Silvicultural ditching resulted in a general lowering of the water table and extensive loss of wetland habitat. A less obvious result of hydrologic alteration was an exaggeration of water level fluctuations within the wetlands that remained, caused by the damming effect of roadways during extremely wet periods and excessive drainage via the ditch-culvert system during dry periods. During extreme high water events the ditch-culvert system cannot convey water as rapidly as the broad natural drainageways had, but the ditches continue to convey water after levels have receded below the elevation of natural drainageways.

Hydrologic alteration of Tates Hell Swamp has also had an apparent impact on water quality in adjacent East Bay. The area had originally been characterized by continuous, gradually diminishing release of water for many weeks following rainy periods. This provided for relatively stable delivery of fresh water to the bay throughout both wet and dry seasons. Ditching caused rapid wet-season delivery and very limited dry-season delivery, resulting in large salinity fluctuations in the bay. Long-term studies have indicated that hydrologic alterations, together with extensive clear-cutting, cause periodic increases in nutrient levels and water color, and decreases in dissolved oxygen and pH in upper portions of the bay (Livingston and Duncan 1979).

In order to restore the natural wetland hydrology of the Big Slough Branch demonstration site, eight low water crossings (LWCs) were constructed at locations where roads obstructed natural drainage features (Figure 3-1). (Due to the large number of figures and tables in this and subsequent chapters, figures and tables are grouped at the end of each chapter.) Ditches were blocked in order to retain water on the site and direct flow toward the natural drainageways. To construct the LWCs, sections of road ranging from 100 to over 500 feet in length were reduced to natural grade in such a manner as to approximately duplicate cross sections of natural drainageways. The bottoms of some LWCs were lined with crushed limerock ("hardened") to allow vehicle passage, while others were left with natural bottoms. Restoration work is described in detail in the final construction report (NWFWMD 1999).

Surface Water Monitoring

Data collection began in April 1998, shortly before the onset of a severe summer drought that continued until after restoration activities were completed in July. Rainfall averaged less than two inches per month from April through June 1998, followed by nearly twelve inches in July, four inches in August, and over 20 inches in September (Table 3-1). Rainfall for the remainder of the study period was erratic, but generally high enough to

maintain hydrated conditions in the wetlands. Due to the drought, it is difficult to make meaningful pre- versus post-restoration comparisons. However, the data do allow comparison between the hydrology of the pre-restoration demonstration site and that of the control site.

The six weeks of stage data collected at the demonstration and control sites prior to the drought are presented in Figure 3-2. Conditions were already quite dry, and water levels at both sites were between 1.5 and 2.5 ft below median wetland elevation. The two stations are similar in terms of magnitude of response to rain events, slope and duration of recession curves, and water level relative to land elevation. Median stage was 1.8 ft below median land elevation at the demonstration site, versus 1.9 ft at the control site. The total stage range during this period was 0.78 ft at the demonstration site versus 0.81 ft at the control site. This strong pre-restoration resemblance suggests that post-restoration differences between the two sites can be at least partially attributed to the restoration.

Water levels fell below minimum recordable levels in middle May 1998, and did not rise to recordable levels until middle July, after restoration had been completed. (Due to lightning-induced equipment failure, recording of stage at station S505 did not resume until September.) Post-restoration stage data indicate that restoration was successful in raising and stabilizing water levels in the demonstration site. For the period October 1998 through October 1999 median stage at the demonstration site was approximately 0.1 ft above median land elevation, with a range of 1.58 ft (Figure 3-3). In contrast, median stage at the control site was approximately 0.6 ft. lower than median land elevation, with a range of 2.40 ft (Figure 3-4).

Examination of wetland hydroperiods provides additional perspective on differences between the demonstration and control sites. Detailed elevation cross-sections were developed for the two sites, both of which are broad sloughs distinctly defined by bordering uplands. Hydroperiods were determined by combining elevation data with stage data for water year 1998-1999, the only complete year for which data are available. (Rainfall in water year 1998-1999 totaled 62 inches at station S536, approximately an average year for this location.) Median hydroperiod at the demonstration site was 230 days, with 70% of the site exhibiting a hydroperiod of 120 days or longer (Figure 3-5). Median hydroperiod at the control site was only 34 days, and more than 60% of this site had a hydroperiod of less than 60 days. The long hydroperiods seen in the demonstration site are more consistent with the wooded swamp communities that exist at both the demonstration and control sites, and are likely to discourage the encroachment by pines and other inappropriate species observed at both sites.

Restoration did not raise water levels in the interior of the demonstration site (station S536) to the degree it had at the lower end of the demonstration site (Figure 3-6). However, the magnitude of post-restoration stage fluctuations were similar at the two stations. No detailed ground elevation survey was done at station S536 due to the poorly defined wetland cross section at this location.

Ground Water Monitoring

Piezometric ground water levels at the control well array ranged from 0.3 ft to six ft below land elevation (Figure 3-7). A great deal of variation was observed among individual wells, with levels generally rising with increasing distance up to 170 ft from the road-ditch system. This suggests that the approximately four foot deep ditches at this site influence the water table at considerable distances. Water levels at the demonstration site well array ranged from 0.2 ft above land surface to 3.5 ft below (Figure 3-8). Inter-well variation was less than that at the control site, with only the closest well, 15 ft from the ditch, showing significantly depressed water levels. Thus, the influence of the partially backfilled, approximately 2 ft deep ditches at the demonstration site appears small. While the differences between the two well arrays may be attributable to the restoration work, they may simply be due to the differing locations of the arrays, since no pre-restoration data are available to indicate otherwise.

Deep and shallow piezometers were installed in pairs at the demonstration site in order to examine the piezometric surface above and below the clay loam flow-restricting layer located at a depth of 4 to 6 ft at this site. These data can be used to detect any evidence of either downward infiltration or upward flow from the aquifer—the "diffuse upward leakage" hypothesized by Parker and Rasmussen (1998). Darcy's Law dictates that a lower piezometric surface at lower depths results in downward flow, while the opposite situation causes upward flow. Data collected to date (Table 3-2) tentatively indicate that flow direction varies according to conditions. The two measurements taken when the water table was low, on 3/30/99 and 9/18/99 both show an upward pressure gradient consistent with diffuse upward leakage. The four high water table measurements show either very little gradient, or a downward gradient, indicating infiltration. Depending on the hydraulic conductivity of the clay loam soil stratum, the magnitude of these pressure differentials may be sufficient to drive substantial upward or downward flow. Hydraulic conductivities for soils of this type can range from less than 0.01 to over 0.1 inches per hour (Chow 1964). Assuming a value of 0.1 inches per hour and a thickness of 1.5 ft for the low permeability stratum, a pressure differential of 0.2 ft will produce a flow of 0.32 inches per day. Actual measurement of hydraulic conductivity (a difficult procedure to perform correctly), as well as extensive additional water level measurements, will be necessary to clarify this complex issue.

Figure 3-1. Location of low-water crossings in the Big Slough Branch demonstration site.

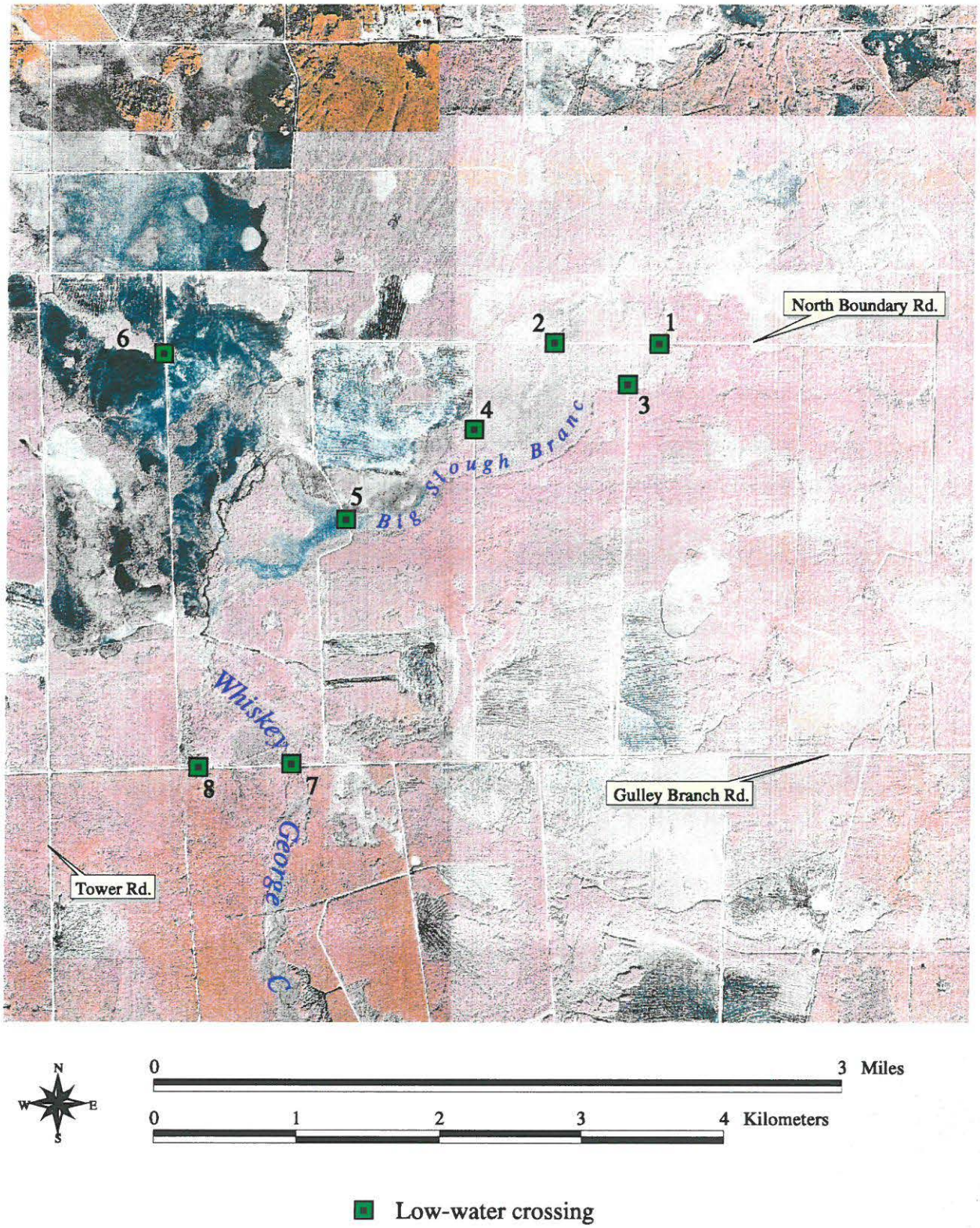


Table 3-1. Monthly rainfall at the demonstration site.

Month	Rainfall (in)	Month	Rainfall (in)
Apr-98	1.9	Jan-99	6.8
May-98	2.3	Feb-99	3.2
Jun-98	1.5	Mar-99	3.4
Jul-98	11.6	Apr-99	7.0
Aug-98	4.2	May-99	7.8
Sep-98	21.8	Jun-99	4.7
Oct-98	1.8	Jul-99	10.1
Nov-98	1.2	Aug-99	6.1
Dec-98	2.1	Sep-99	8.8
		Oct-99	4.11

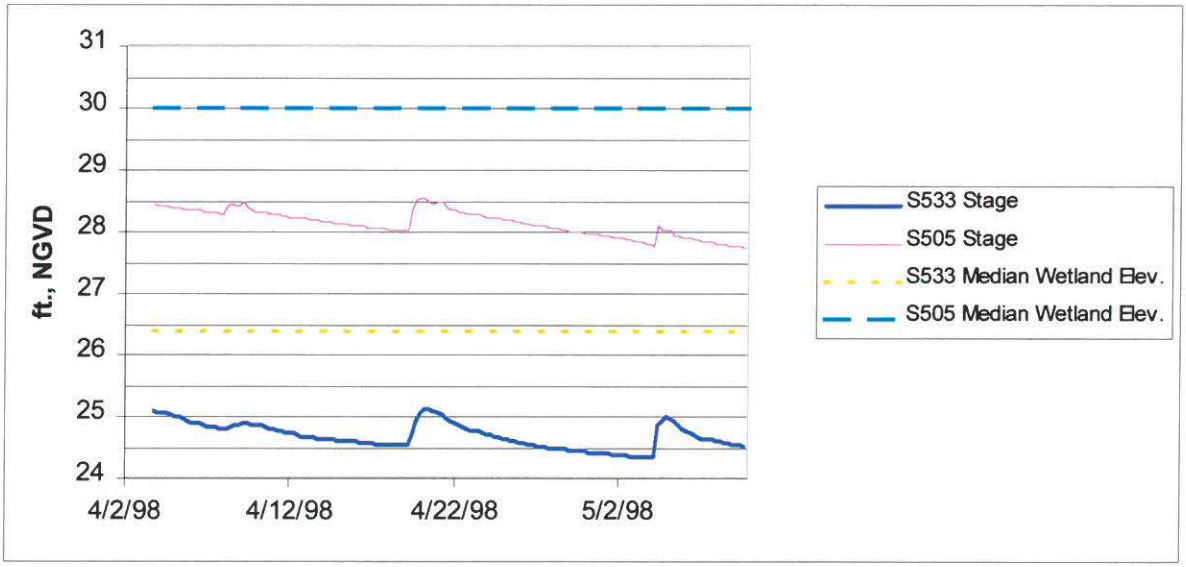


Figure 3-2. Pre-restoration stage at the demonstration (S533) and control (S505) sites

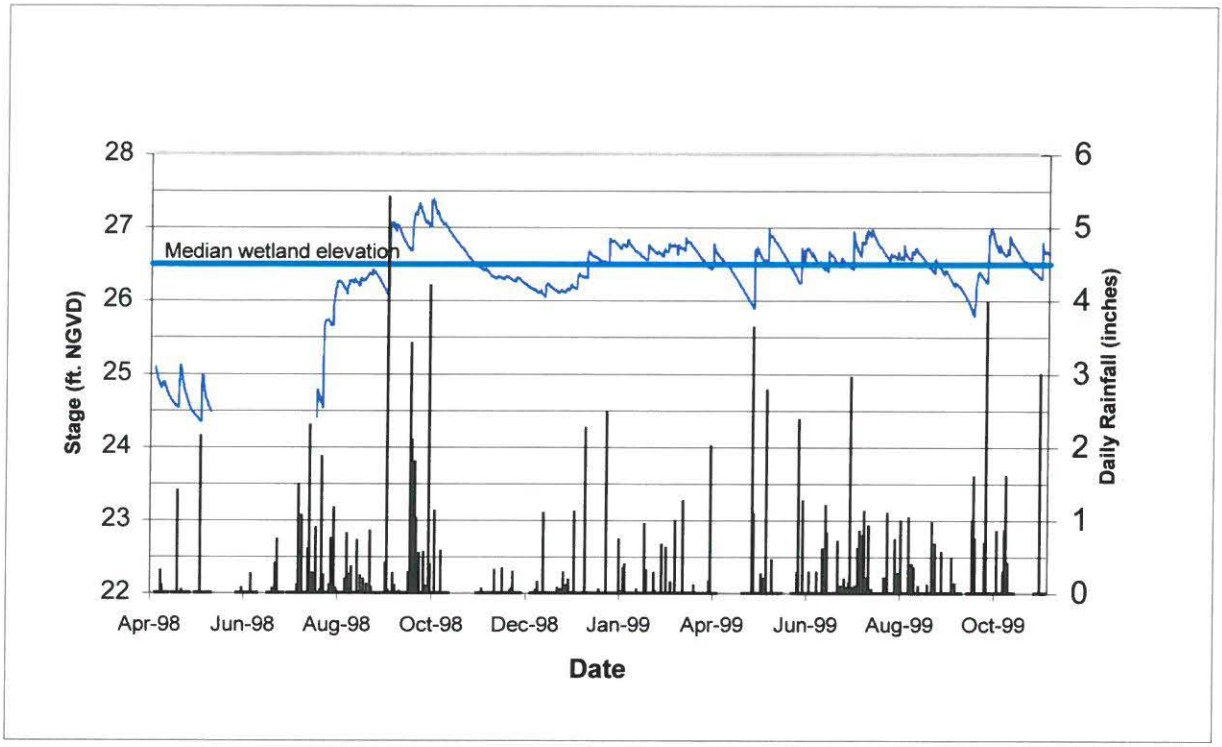


Figure 3-3. Stage and rainfall at the demonstration site (S533).

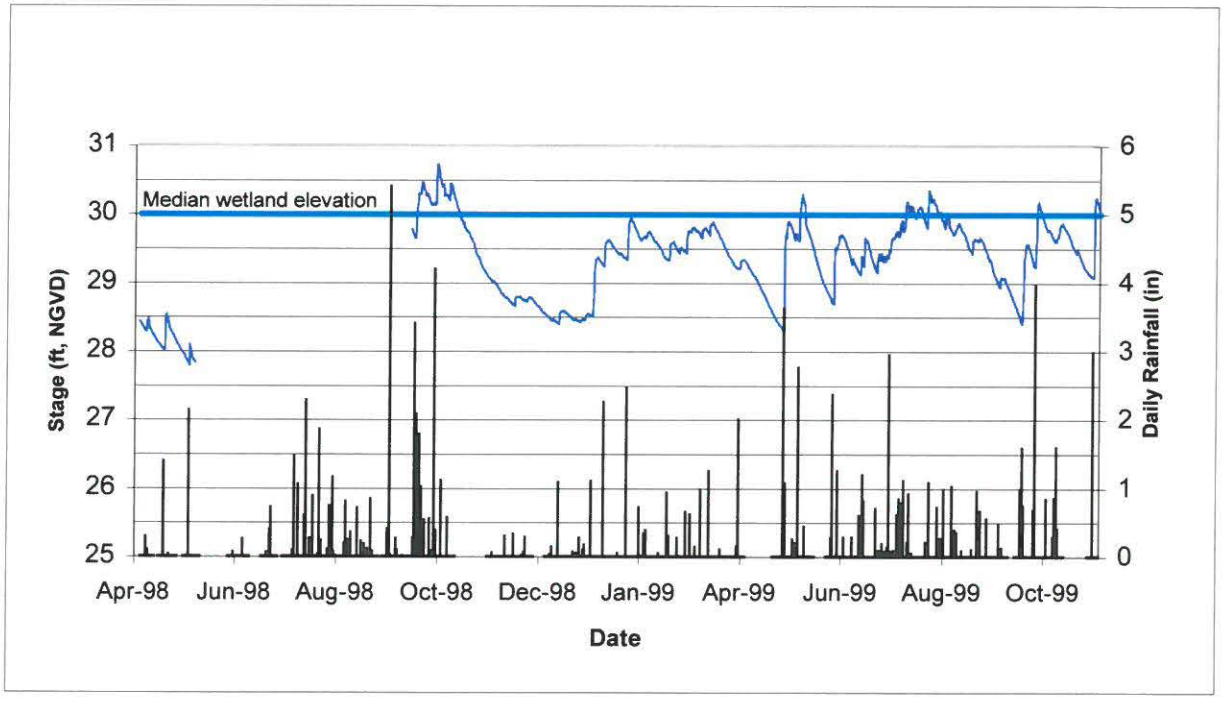


Figure 3-4. Stage and rainfall at the control site (S505).

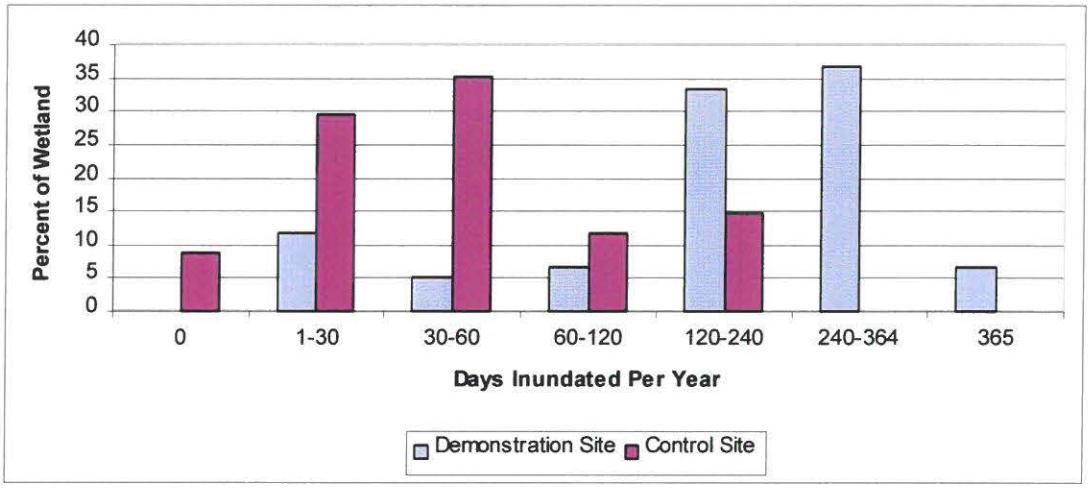
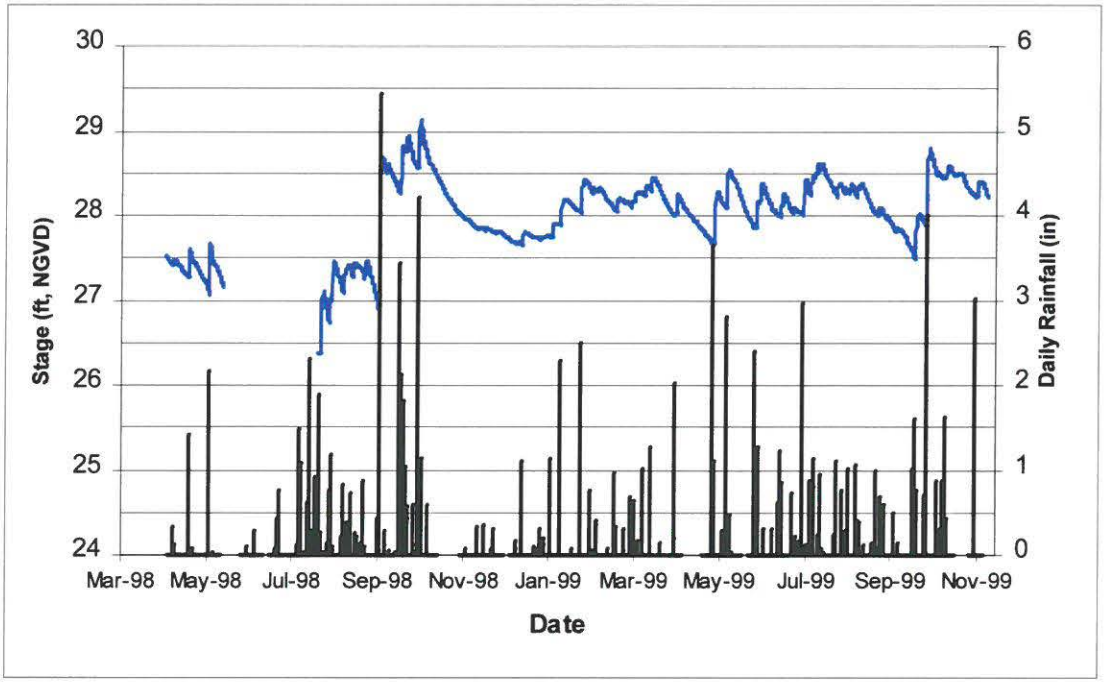


Figure 3-5. Wetland hydroperiods at the demonstration and control sites, water year 1998-1999.



Note: Median wetland elevation not determined at this site; see text.

Figure 3-6. Stage and rainfall at the interior of the demonstration site (S536).

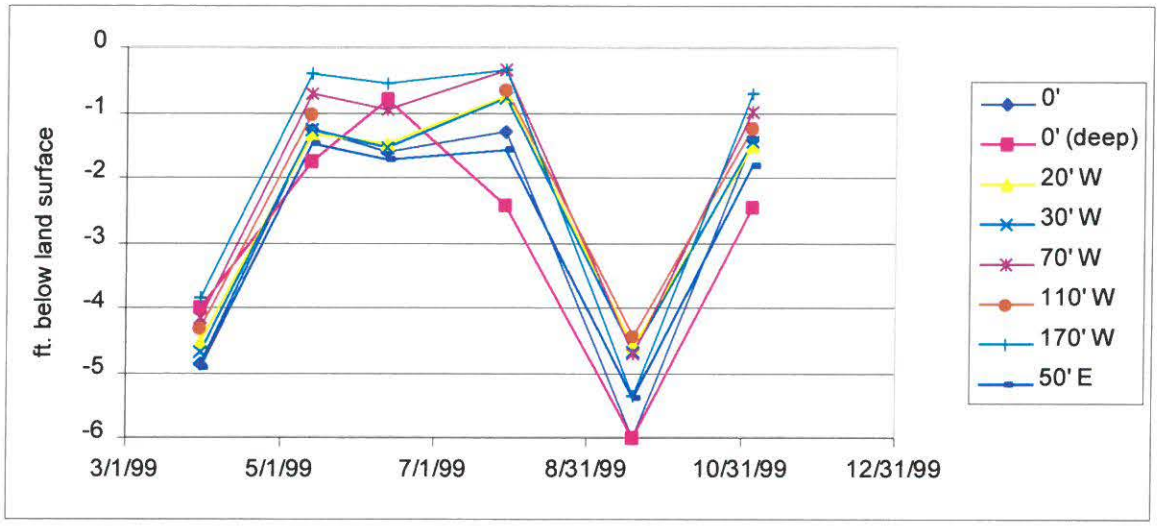


Figure 3-7. Piezometric ground water levels at the control site well array.

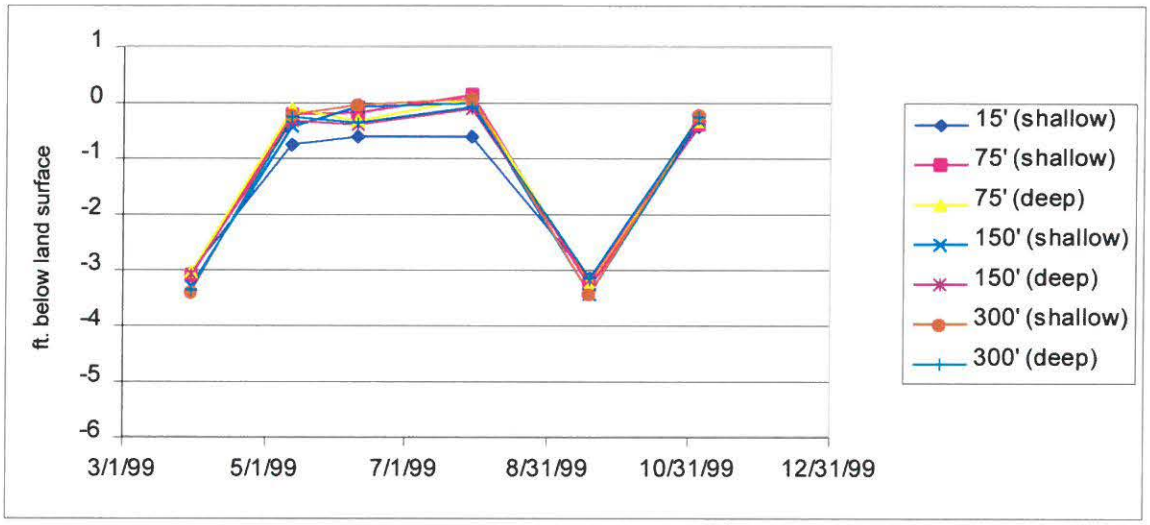


Figure 3-8. Piezometric ground water levels at the demonstration site well array.

Table 3-2. Mean piezometric surface at shallow and deep wells in the demonstration site.

Date	Shallow	Deep	Difference
	----------ft.----------		
3/30/99	3.253	3.147	0.107
5/13/99	0.290	0.223	0.067
6/10/99	0.092	0.359	-0.267
7/29/99	-0.080	0.039	-0.119
9/18/99	3.381	3.176	0.205
11/4/99	0.310	0.283	0.027
Average			0.003

WATER QUALITY

Water quality in Tates Hell is generally very good. The area is largely undeveloped, the only significant potential source of water quality degradation being silviculture. No silvicultural activities were conducted on or adjacent to any of the study sites during the monitoring period. Water quality monitoring was conducted for the purpose of establishing baseline conditions, assessing the impacts of restoration activities, and providing background data to support biological observations.

Nutrient levels were very low at all sites throughout the study. Of 107 observations, 52 ammonium values, 55 nitrate-nitrite values, 74 total phosphorus values, and 82 ortho-phosphorus values were at or below laboratory practical quantification limits (PQLs), with most of these also lying below the minimum detection limit (MDL). (PQL is the level at which the analyte can be detected, but not accurately quantified; MDL is the level below which the analyte cannot be detected. The DEP laboratory reports estimated values for samples that lie below the PQL but above the MDL. For values at or below the MDL, the MDL is reported. Data were tabulated for this report exactly as reported by the laboratory.) Heavily left-censored data of this type violate the assumptions of normality and mean-independent variance that are required for traditional parametric statistical analysis. For this reason , the nonparametric Wilkoxon rank sum test (for comparisons between two groups) or Kruskal-Wallis rank sum test (for more than two groups) were used to determine statistical differences in these data (SAS Institute 1995). Analysis is still problematic because none of the many data points that lie below the MDL can be distinguished from one another regardless of whether parametric or nonparametric methods are employed. (While means and standard errors are shown in the water quality figures, standard errors are presented for descriptive purposes only, and are not applicable for discriminatory purposes.)

Previous work conducted in Tates Hell (NWFWMD 1999) found little or no difference between storm event and base flow conditions with respect to nutrient and suspended solids concentrations. To examine this issue at the current study sites, five of the 13 sampling events were timed to occur during or within two days following rainfall events of two inches or greater. For most nutrients, no significant differences were seen between storm and base flow concentrations. However, both ammonium and total suspended solids concentrations were significantly higher during base flow than during storm runoff events (Table 4-1). These results were consistent among demonstration, reference, and control sites. Ammonium produced by ammonification of organic nitrogen in the sediments and suspended solids produced by a number of possible processes apparently accumulate in the water column during low flow conditions, and are flushed out during rain events. Failure to observe elevated nutrient and suspended solids concentrations during storm events—as is commonly observed in streams—was likely due to the low hydraulic gradients, low water velocities, and predominance of natural groundcover in the study area.

For the purpose of comparing the demonstration, control, and reference sites before and after restoration, sampling stations in the demonstration site were divided into upper (S537 and S541) and lower (S533 and S534) sectors. This was not done for the control and reference sites because preliminary analysis indicated minimal differences between upstream and downstream water quality for these sites. Comparisons are presented in Figures 4-1 through 4-7. Due to the limited number of pre-restoration sampling events (two events for most sites, one event for the lower demonstration site) few statistically significant differences between pre- and post-restoration conditions were found.

Overall ammonium N concentrations averaged less than 0.05 mg/L. No statistically significant differences were observed between pre- and post-restoration samples. The reference site was slightly, but significantly lower in ammonium than the other sites. Nitrate-nitrite N concentrations averaged less than 0.02 mg/L, with no significant differences either among sites or between pre- and post restoration. Total Kjeldahl N averaged 0.8 mg/L, with the reference site significantly lower than the other sites.

Total organic carbon concentrations averaged 21 mg/L overall. The lower demonstration site was significantly higher in TOC than the other sites, while the reference site was significantly lower. No significant pre- post-restoration differences were observed.

Interpretation of total P and ortho-P data is particularly problematic due to the overwhelming number of observations at or below the PQL. No significant differences in total P could be detected either among sites or between pre- and post-restoration. Oddly, post–restoration ortho P was significantly higher than pre-restoration for all sites except the lower demonstration site. This difference is difficult to explain, but given the very low concentrations involved, it is believed that the effect may be spurious, or perhaps an artifact.

Total suspended solid concentrations were significantly higher in the upper demonstration site than in other sites. Suspended solids in natural wetlands tend to be largely autocthonous (generated in situ), and the dynamics of solids generation and suspension in wetlands is very complex. None of the suspended solids concentrations observed in this study are unusual, and no explanation is offered for the observed site effect.

The observed higher TKN and TOC values in the lower demonstration site than in the upper demonstration site suggest that the demonstration site functions as a source for these two elements. This is not uncommon in predominantly ombrotrophic wetlands such as Tates Hell. Nitrogen and carbon fixed in the rainfall-fed headwaters of these systems are exported downstream. The control and reference sites did not exhibit this effect. In the case of the control site, this is due to the close proximity of the upstream and downstream stations. The situation for the reference site is less clear. The lack of difference between upstream and downstream stations may be related to the relatively undisturbed nature of the reference site, or possibly to the flow configuration. The downstream end of the demonstration site receives water from a relatively linear system of wetlands, with long, extensive areas of sheet flow that allow for accumulation of

organic carbon and nitrogen in the water. The watershed configuration of the reference watershed is more palmate than linear; that is, a number of small, independent sub-basins each discharges into a common collecting waterway. The longer travel/residence time in the linear system further promotes carbon and nitrogen accumulation.

Nitrogen, phosphorus, and suspended solids concentrations observed in this study are typical for oligotrophic wetlands, and are comparable to those found in natural areas of the Everglades (Kadlec and Knight 1995). As noted earlier, no silvicultural activities took place in or adjacent to any of the study sites during the study period. However, incidental sampling was conducted during the study period at a silviculturally-impacted site approximately 10 miles southeast of the demonstration site. This sampling site was downstream of privately-owned property on which extensive silvicultural land-preparation and "ditch maintenance" activities had been occurring. Samples ranged as high as 1.5 mg/L ammonium N, 3.8 mg/L TKN, and 0.26 mg/L total P. This is a stark contrast to concentrations found at the main study sites, and represents severe water quality degradation, which is undoubtedly of considerable ecological significance. Studies indicate that forestry in North Florida can have a minimal impact on water quality if state BMP guidelines are observed (Frydenborg 1997). It is clear both from casual observation and from water quality impacts that adequate BMPs were not being practiced at this location.

Dissolved oxygen and pH measurements were taken in conjunction with biological sampling at three stations in the upper demonstration site and one each in the control and reference sites. Measurements were taken in wetlands and at top and bottom depths of demonstration and control site ditches. Due to logistic problems and dry conditions, no pre-restoration oxygen or pH monitoring was conducted in the control or reference wetlands, and only two pre-restoration monitoring events were conducted in the ditch sites. Little systematic variation was observed for either parameter (Figures 4-8 and 4-9), except for the expectably lower DO readings at the ditch bottoms. Both DO and pH values were typical for wetlands of this type.

Table 4-1. Mean nutrient and suspended solids concentrations under base flow and storm runoff conditions.

Parameter	Base Flow	Storm Runoff
	----------mg/L----------	
Ammonium N	0.066*	0.017*
Nitrate-Nitrite N	0.017	0.019
Total Kjeldahl N	0.92	0.79
Total Organic Carbon	21.5	22.2
Total Phosphorus	0.023	0.017
Ortho-phosphorus	0.011	0.006
Total Suspended Solids	11.74*	4.73*

*Base flow and storm values significantly different (p=0.05).

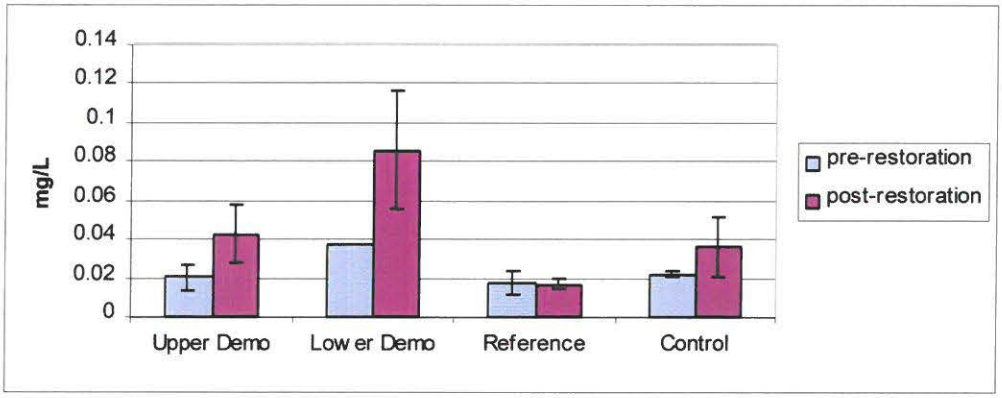


Figure 4-1. Mean ammonium nitrogen concentrations in demonstration, control, and reference sites before and after restoration.

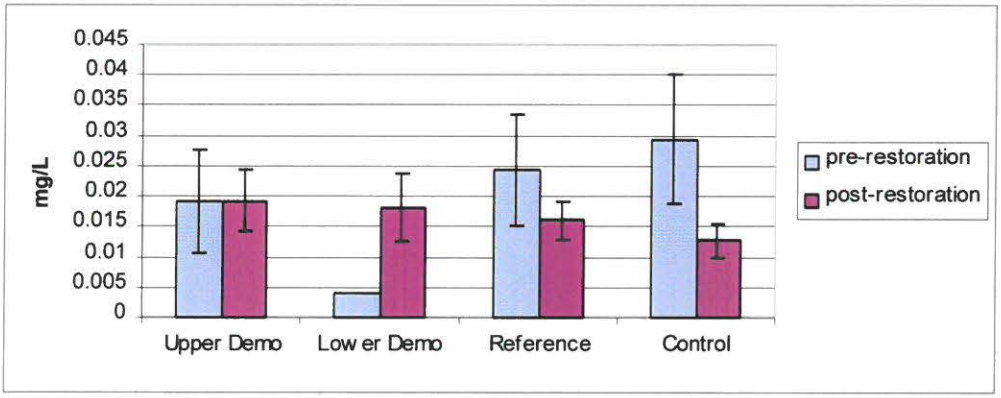


Figure 4-2. Mean nitrate-nitrite nitrogen concentrations in demonstration, control, and reference sites before and after restoration.

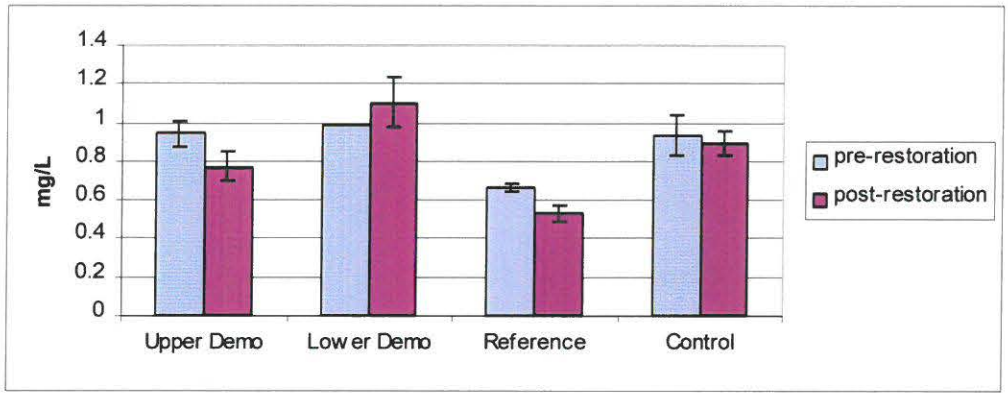


Figure 4-3. Mean total Kjeldahl nitrogen concentrations in demonstration, control, and reference sites before and after restoration.

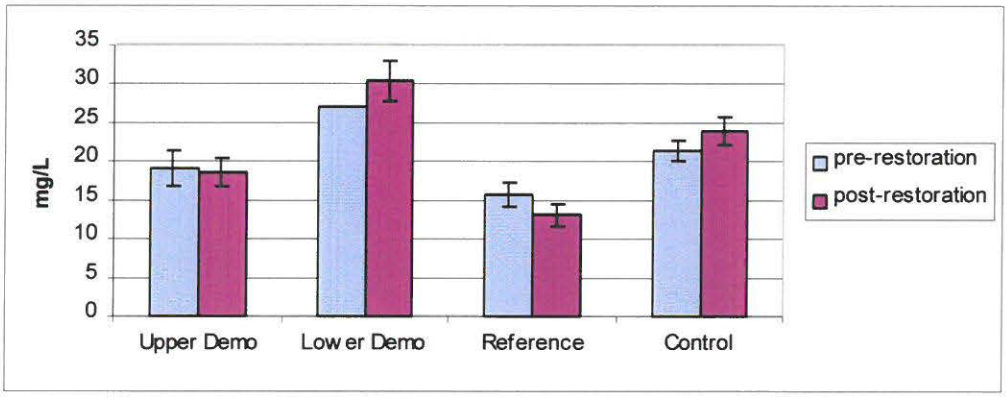


Figure 4-4. Mean total organic carbon concentrations in demonstration, control, and reference sites before and after restoration.

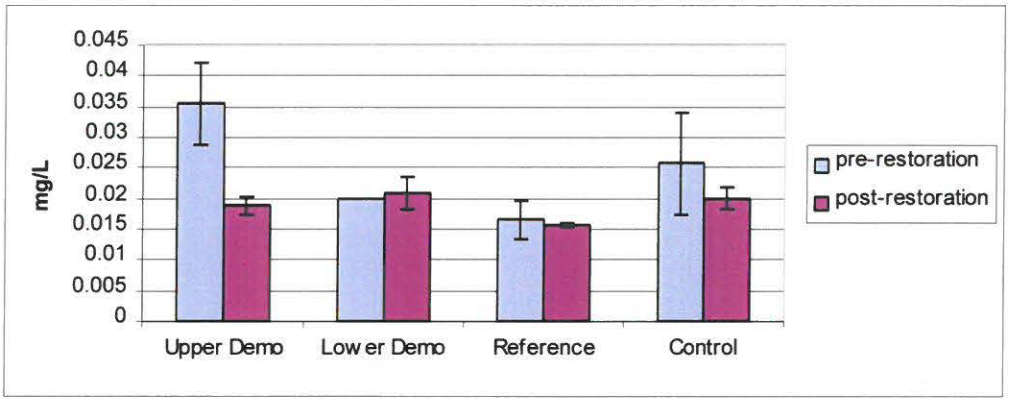


Figure 4-5. Mean total phosphorus concentrations in demonstration, control, and reference sites before and after restoration.

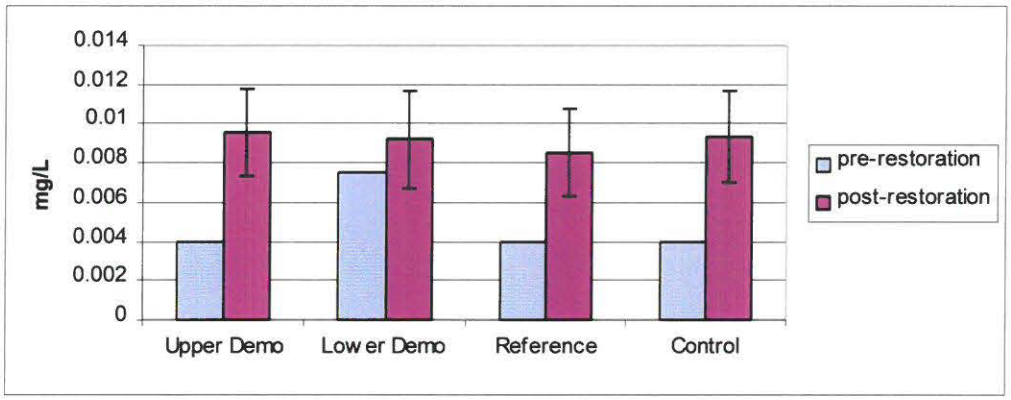


Figure 4-6. Mean ortho-phosphorus concentrations in demonstration, control, and reference sites before and after restoration.

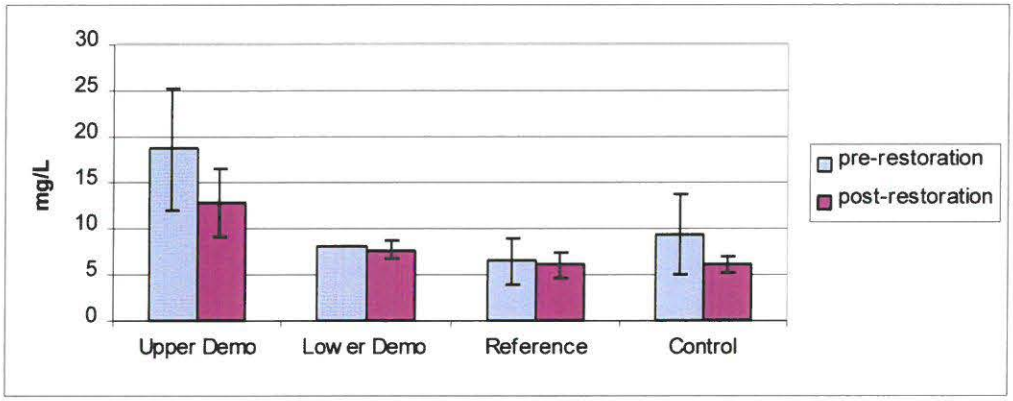


Figure 4-7. Mean total suspended solids concentrations in demonstration, control, and reference sites before and after restoration.

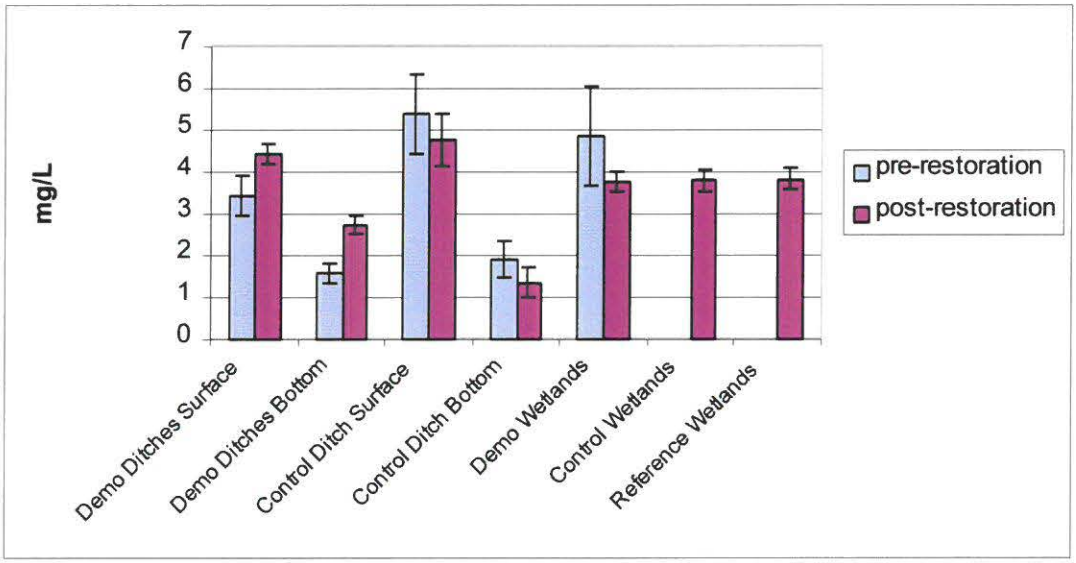


Figure 4-8. Mean dissolved oxygen concentrations in demonstration, control, and reference sites before and after restoration.

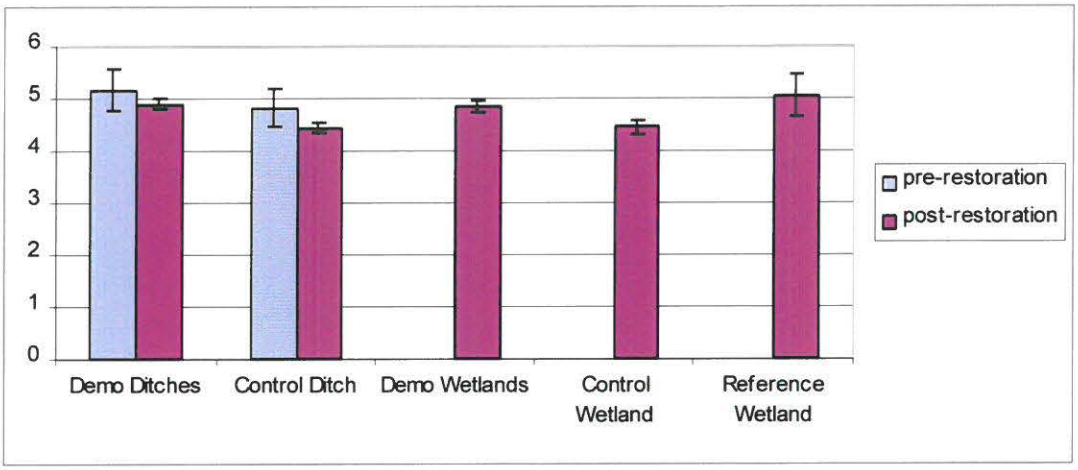


Figure 4-9. Mean pH in demonstration, control, and reference sites before and after restoration.

BENTHOS

Benthic macroinvertebrates, or benthos, are widely used in assessing the biological integrity of aquatic systems. The Florida Stream Condition Index (SCI) and Lake Condition Index (LCI) both rely heavily on benthic community structure. These indices employ a variety of benthos metrics based on community diversity, number of pollution-intolerant taxa, and related factors. Because conditions in wetlands are very different from those in streams and lakes, many of the methods and metrics used in the SCI and LCI may not be applicable to wetlands. In particular, many pollution-intolerant taxa found in streams and lakes do not occur in even the highest quality wetlands. Due to the difficulty of applying stream- and lake-based metrics to wetlands, the only metrics explicitly addressed in the following discussion are species richness and density. Specific taxa are, however, discussed in detail, and it is hoped that observations made in this study will be of use in the development of a wetland condition index.

The dip net collection methodology favored for stream and lake sampling is not always possible in wetlands with extremely low water levels and periodic lack of standing water. For this reason, both dip net and core sampling were employed for benthic macroinvertebrates in this study.

Benthos have been widely researched in wetlands (Bataille and Baldassarre 1993; Brown et al. 1997; Corti et al. 1997; Jeffries 1994; Murkin and Kadlec 1986; Rader 1994). Much of the research on cypress wetlands has focused on cypress-tupelo swamps (Beck 1977; Sklar 1985; Sklar and Day 1985; Thorp et al. 1985), although some analyses of cypress domes have been performed as well (Brightman 1984; Leslie 1996; Leslie et al. 1997).

Species Richness and Density

Over the course of the study, 111 benthic taxa were collected from Tates Hell Swamp. This richness is similar to the 104 taxa that were collected in a cypress dome in North Central Florida (Prenger *et al.* in prep.). The taxonomic richness in these two studies is within the range reported for other types of wetlands (Figure 5-1). In both these studies, however, taxa were identified to the lowest practical taxonomic level, which typically was to genus, where possible. The richness would have been higher in both studies had taxa been identified to the species level. In other studies where taxa were identified to the species level such as those of Moorhead (1998) and Leeper (1998b), the taxon richness was higher than other macroinvertebrate studies performed in wetlands. Leslie (1997) found 85 taxa of benthic macroinvertebrates in cores from a north central Florida cypress dome. Duffy (1994), working in forested wetlands, collected 48 taxa in cores in the winter and spring; and Golladay (1997), working in forested Georgia lime sink wetlands collected only 33 taxa from cores and woody debris. At the opposite end of the spectrum, Leeper (1998b) collected 115 taxa from a forested Carolina Bay. Riparian

swamps are variable, ranging from 72 taxa collected from substrate samplers (White 1985) to 98 taxa collected from cores (Corti *et al.* 1997). Marshes also fit this pattern, with taxa richness ranging from 40 taxa and 43 taxa in cores and sweep nets, respectively (Evans 1996) to 107 taxa from sweep nets and cores combined (Moorhead *et al.* 1998).

Due to the limited number of pre-restoration monitoring events and strong seasonality of the data, pre-/post-restoration comparisons will not be presented for biological parameters. Taxon richness and density data for the demonstration, control, and reference wetlands are presented on an individual sampling event basis in Figures 5-2 through 5-5.

Taxon richness for core samples averaged 8.9 taxa per sample (Figure 5-2). Little difference was observed between demonstration, control, and reference wetlands. Seasonal declines in richness during summer 1998 and winter/spring 1999 reflect dry conditions at those times.

Sweep net richness values (Figure 5-3) were slightly higher than those for cores, averaging 10.2 taxa per sample. Seasonal trends were similar to those observed in the core data, but somewhat more extreme. Lack of water in the control and reference wetlands precluded sweep net sampling on several sampling dates. For this reason, meaningful comparisons among demonstration, control and reference wetlands cannot be made.

The mean density of macroinvertebrates collected from cores in this study was 2,757 individuals/m^2, varying from a maximum of 8,081 individuals/m^2 in April 1998 to a minimum of 303 individuals/m^2 in July 1998 (Figure 5-4). These data are in the range of those reported from other cypress wetlands, but below that of marshes (Figure 5-6). Marshes typically possess high cover and diversity of submersed, floating, and emergent vegetation, which can provide structure and food for macroinvertebrate communities (Mitsch and Gosselink 1993). Many cypress wetlands and other swamps have limited understory vegetation, and this may limit the density of macroinvertebrates present.

Sweep nets yielded significantly lower invertebrate densities (Figure 5-5) than core sampling, mostly because sweep net sampling is not efficient at collecting sediment infauna, and the most common invertebrates (amphipods, isopods, chironomids, and ceratopogonids) are fauna associated with sediments. The mean density of macroinvertebrates collected from sweep nets in this study was 289 individuals/m^2, varying from a maximum of 1,632 individuals/m^2 in January 1999 to a minimum of 16 individuals/m^2 in November 1998.

Comparison of core and sweep net data illustrates the utility of corers for benthic work in intermittently flooded wetlands. Coring was possible on many occasions when insufficient water was present for use of sweep nets. Even when both methods could be employed, coring yielded much more consistent data than sweep nets. Taxa richness was only slightly lower for cores, while densities were much higher.

Coleopterans and dipterans (Table 5-1) dominate the benthos of Tates Hell Swamp. Within these two orders, the coleopteran family Dytiscidae accounted for 20% of the taxon richness, and the dipteran family chironomidae accounted for 37% of the taxon richness. These groups are often the dominant taxa in many drought-prone wetlands and other temporary waters (Leeper and Taylor 1998a; Leslie *et al.* 1999; Taylor *et al.* 1999; Williams 1987). The dipteran families Ceratopogonidae (10%) and Chironomidae (37%), the amphipod *Crangonyx* (20%), and the isopod *Caecidotea* (21%) were the dominant invertebrate groups in cores, contributing 88% of total density over the course of the study.

Coleopterans have limited ability to withstand drought conditions *in situ*, but they are very mobile and can simply fly or crawl to an area with standing water (Fernando and Calbraith 1972). This trait also allows them to reestablish populations quickly in cypress strands when water levels rise. No one coleopteran genus was dominant during the course of the study, with the total coleopteran density accounted for by many genera occurring at low densities.

Most dipterans can survive drought conditions in cypress wetlands (Leslie *et al.* 1997; Prenger *et al.* in prep.) and are presumed to go into diapause (Williams 1997). This is especially true of the ceratopogonids and the chironomids *Polypedilum* spp. and *Polypedilum tritum*, which contributed considerably to total dipteran densities in cores. Adults can also recolonize areas that were flooded and, after laying eggs, would be able to repopulate an area. *Polypedilum* spp., *Procladius*, and *Ablabesmyia* were the most commonly collected chironomids, though *Paratendipes* spp., *Parachironomus* spp., *Chironomus*, and *Georhthocladius* were also temporally important constituents of the dipteran fauna.

Densities of the insect orders Odonata, Ephemeroptera, Hemiptera, Lepidoptera, and Trichoptera were low in Tates Hell Swamp. Ephemeropterans, lepidopterans, and trichopterans have few taxa adapted to conditions in southeastern U.S. wetlands (see Merritt and Cummins 1996), and nowhere are they very rich. Ephemeroptera can be found in large numbers in other Florida wetlands but have not contributed much to the total invertebrate density in a series of north central Florida cypress domes (Leslie *et al.* 1999). Neither Lepidoptera nor Trichoptera have contributed significantly to the invertebrate densities in other southern wetlands (Leeper and Taylor 1998a; Leslie *et al.* 1999; Prenger *et al.* in prep.). The presence of ditches in Tates Hell Swamp does not appear to provide suitable habitat for these taxa either, as they do not occur in high densities.

Odonates are often ubiquitous members of the invertebrate fauna in wetlands. In Tates Hell Swamp, the periodic drying of wetlands may be a factor in the low odonate taxon richness and densities. Other wetlands prone to periodic drying also have low odonate taxon richness (Taylor *et al.* 1999).

Hemipterans also were neither a very rich nor abundant fauna in Tates Hell Swamp, despite the fact that many families are found in wetlands (see Merritt and Cummins

1996). Hemipterans are highly motile, even more so than odonates, and this could have presented a problem in adequately sampling them, even with dip nets. For example, the belostomatid *Lethocerus* was rarely collected in either cores or dip nets, yet a number of them were collected in fish traps.

Non-insect invertebrates were also collected in Tates Hell Swamp, but none showed high taxa richness, though amphipods and isopods were present in high densities. The orders Hydracarina, Amphipoda, Isopoda, and Decapoda were the most frequently encountered non-insect taxa. However, Hydracarina were enumerated only as Hydracarina, so this does not show up in the richness. Hydracarina are very difficult to identify, especially since they become fragile when preserved in ethanol (Smith and Cook 1991), so identification only to order was performed.

Amphipods, isopods, and decapods have poor richness at the genus level, though isopods do have high richness at the species level. Since isopods were identified only to the generic level, it is possible that they would have experienced greater species richness had they been identified to species. However, amphipods and isopods were, along with dipterans, the most commonly collected macroinvertebrates.

Amphipods and isopods are among the most numerous macroinvertebrates collected in several southern, forested wetlands (Duffy and LaBar 1994; Leslie *et al.* 1999; Porter *et al.* 1999; Prenger *et al.* in prep.; Sklar 1985). Both orders feed predominantly on periphyton and particulate organic matter, which are abundant in Tates Hell Swamp. One possible reason for their high densities is their ability to persist in moist soils, even if standing water is absent (Taylor *et al.* 1999). They are also known to migrate overland to standing water under drought conditions (Williams 1997).

Copepods were also collected in large numbers in the cores and often were the numerically dominant group. Since copepods were also collected in zooplankton tows (Chapter 5) and enumerated to lower taxonomic levels than simply copepods, copepod data collected in sweeps and cores were not included in the total invertebrate density calculations.

Benthic macroinvertebrates are important transformers of organic matter to usable energy for higher trophic levels. Amphipods, isopods and chironomids, the most abundant benthic invertebrates in Tates Hell Swamp, feed on the bacteria, fungi, and algae coating organic matter (Coffman and Ferrington 1996; Covich and Thorp 1991; Pennak 1989; Smock and Stoneburner 1980; Thorp and Covich 1991). Amphipods, isopods, and many chironomid dipterans serve as important food sources for other macroinvertebrates including odonates and dytiscid coleopterans as well as fish (Pennak 1989; Westfall 1996; White and Brigham 1996). A number of fish species found in Tates Hell Swamp, including pirate perch (*Aphredoderus sayanus*), swamp darter (*Etheostoma fusiforme*), topminnows (*Fundulus* spp.), bullheads (*Ameiurus* spp.), and warmouth (*Lepomis gulosus*) consume benthic macroinvertebrates as a major component of their diet (Hoyer and Canfield 1994; Lee *et al.* 1981).

Feeding Guilds

Tates Hell Swamp appears to be a periphyton-dominated system, although this is likely less the case in wetlands than in ditches and low water crossings. However, since nutrients are limited and other food sources such as microbe-rich particulate organic matter, dissolved organic matter, woody debris, and some phytoplankton are present, the majority of the benthic fauna should be generalists, as is the case in many cypress wetlands (Brightman 1984; Leslie *et al.* 1997; Porter *et al.* 1999; Prenger *et al.* in prep.). The amphipod *Crangonyx*, isopod *Caecidotea* and the chironomid *Polypedilum* are all generalists and together account for 89% of the total benthic macroinvertebrates collected in Tates Hell Swamp. These groups were also observed to be dominant taxa in a titi wetland in the Florida panhandle near Tates Hell Swamp (Haack 1984; Pezeshki 1987). Generalists have been found to predominate in highly colored, wooded sites (Haack 1984) as is the case in Tates Hell Swamp.

Cypress swamps often depend on allochthonous materials such as leaves, cypress needles, and macrophytes to provide coarse particulate organic matter (CPOM) as a food source for microbes, which are the base of the food web (Cummins and Merritt 1996). Generalists are often shredders (either detritivorous or herbivorous), though taxa feeding across several functional groups also are considered generalists. Since CPOM is always present in the cypress strands, regardless of water level and season, the dominance of generalists suggests that they may be better able to survive periods of drought or low primary production. Periodic drying and reflooding also can increase detrital protein levels as a result of microbial colonization (Barlocher *et al.* 1978).

Predators are also an important part of the Tates Hell Swamp benthic community (Table 5-1). As is the case with most aquatic systems, they do not form the dominant functional group (Merritt and Cummins 1996), although some chironomids can temporally contribute a considerable amount to total invertebrate density. Collector-gatherers are also a very rich fauna, but their contribution to the total benthic invertebrate density is minor.

Seasonality of Dominant Taxa

The following discussion deals primarily with corer data, since lack of standing water prohibited sweep net sampling during much of the study, especially in the control and reference wetlands. Total macroinvertebrate densities in Tates Hell Swamp followed the trend established by the four dominant taxa, *Crangonyx*, *Caecidotea*, ceratopogonids, and chironomids. Densities were low during the summer 1998 drought but rebounded relatively soon after the wetlands were filled with water. This was the case in both the demonstration wetland cores (Figure 5-4A) and reference cores (Figure 5-4C) and to a lesser extent in the demonstration wetland sweep nets (Figure 5-5A). Control wetland cores remained at near the same levels during the summer 1998 drought as afterwards (Figure 5-4B).

The amphipod *Crangonyx* occurred in fairly large numbers in the cores throughout the study (Figure 5-7). Abundances in the wetlands decreased during summer 1998 and shortly thereafter, probably due to the severe drought which dried the surface layers of the sediments. This summer 1998 decrease was not different from the summer 1999 decrease, suggesting the summer 1998 drought was not a limiting factor ($t = 0.416$; $P = 0.686$). Sweep nets showed a similar trend (Figure 5-8), though this is more likely due to the fact that sweep nets were not used in most sites during summer 1998 because of the lack of standing water. In cores, the highest densities of *Crangonyx* were collected from the control wetland prior to the drought. The reference wetland had variable densities and multiple peaks occurred throughout the year.

The isopod *Caecidotea* followed the same general trend as *Crangonyx*, although a more noticeable peak occurred prior to the drought/restoration with *Caecidotea*. As with *Crangonyx*, *Caecidotea* also experienced a summer 1998 decrease in density in both core (Figure 5-9) and sweep net samples (Figure 5-10), though this was only noticeable in demonstration wetlands. It is likely that the drought played a role in limiting *Caecidotea* during this time, although the difference in densities between summer 1998 and summer 1999 were not significant ($t = 1.70$; $P = 0.12$). Both the control and reference wetlands followed the same trend with larger densities prior to the drought restoration, though the highest density in the reference wetland occurred in March 1999.

Coleoptera were collected in moderate numbers in cores throughout the study (Figure 5-11). Coleopteran taxon richness was high in this study, but no one taxon dominated. Most coleoptera are very mobile, and cores are not the most efficient manner to collect them. However, the same trend also occurs for sweep nets, with low densities throughout the sampling period (Figure 5-12), although densities were higher in sweeps than cores. The same trend is seen in both the control and reference cores.

Diptera follow the same trend as *Crangonyx* and *Caecidotea*, with the densities in demonstration cores decreasing during the drought and rebounding thereafter (Figure 5-13). This is also evident with the control wetland cores. Reference wetland cores exhibited summer decreases, but without the rebound shown by demonstration and control wetlands. The reference wetland was dry for much longer than any of the other wetlands, and dipterans appear to have been unable to survive these conditions. Demonstration wetland sweep nets exhibited the same trend as did the demonstration and control cores (Figure 5-14).

Ceratopogonid densities were commonly collected throughout the study period. As with some other groups, ceratopogonid densities in demonstration wetland cores decreased during summer 1998 (Figure 5-15), although the summer 1998 densities were not lower than those of summer 1999 ($t = -1.05$; $P = 0.32$). An increase during the winter/early spring in the demonstration wetland cores could represent a repopulation of the wetlands after drought conditions subsided. Sweep net samples from demonstration wetlands showed a very similar trend, with the summer 1998 decrease, though densities remained low after the first three months of sampling (Figure 5-16). Cores from the control wetland resembled those from the demonstration wetlands, with more individuals

collected prior to the drought, and few during and afterwards. Reference wetlands did not, however, exhibit the same rebound as the other wetlands.

Chironomids followed the same general trends as other dipterans and cores (Figure 5-17) and sweep net (Figure 5-18), patterns were nearly identical. Chironomids make up the majority of the dipterans sampled in Tates Hell Swamp, and their densities influenced total dipteran densities considerably.

Three species of the genus *Polypedilum* were collected from Tates Hell Swamp. *P. fallax* was only collected in substantial numbers in demonstration wetland sweep nets during January 1999 (Figure 5-19). *P. trigonus* was collected in demonstration wetlands in sweep nets (Figure 5-20) and cores (Figure 5-21) only after the summer 1998 drought. *P. tritum* was the most common of the three species and, along with *Procladius*, the two most common chironomids in Tates Hell Swamp. As with most other taxa, the densities of *P. tritum* were lowest during the summer 1998 drought, though they were uncommon during the warmer months of 1999 as well. This was the case in both cores (Figure 5-22) and sweep nets (Figure 5-23). It is possible that *P. tritum* prefers cooler periods, although literature on environmental tolerances of most aquatic macroinvertebrates is sparse.

The chironomids *Chironomus* (Figure 5-24), and *Tanypus* (Figure 5-26) were collected in low densities throughout the study in demonstration wetland cores, but did show a trend towards decreased densities during summer and winter 1998. This likely represents the effect of the severe summer 1998 drought and its effects may finally have abated as late as early 1999. A similar trend was seen with these genera [*Chironomus* (Figure 5-25) and *Tanypus* (Figure 5-26) in control and reference cores. Demonstration wetland sweep net samples exhibited a similar trend of decreased densities during and immediately after the drought with the exception of *Tanypus*, which exhibited peak density in late fall-winter 1998.

Procladius was much like the preceding chironomids, but densities were generally much higher. Again, cores had low densities during summer 1999 (Figure 5-28). *Procladius* collected from sweep nets were more abundant during the first three months prior to drought/restoration (Figure 5-29).

Hemipterans (Figure 5-30) and odonates (Figure 5-32) were similar to coleopterans in that they were present in low densities in demonstration wetland sweep nets throughout the study. Hemiptera (Figure 5-31) and odonates (Figure 5-33) were also present throughout the study in demonstration wetland cores. The control wetland had very few hemipterans present, and only in the beginning of the study, while the reference site had low densities throughout the study. Odonates were not found in the control wetland cores and were rare in reference wetland cores.

Low odonate densities are not unexpected as, unlike forms which can survive drawdown *in situ* such as chironomids, *Crangonyx*, and *Caecidotea*, or those which can move to areas with standing water such as Coleoptera and Hemiptera, odonates in general do not possess any special adaptations to survive drawdown (Wiggins *et al.* 1980; Williams

1987). The fluctuating hydroperiod of Tates Hell Swamp would make the wetlands fairly inhospitable habitats for many odonates.

Other organisms were encountered infrequently in Tates Hell Swamp. The chironomid *Tanytarsus* was found in sweep nets at times in large numbers, but rarely occurred in cores. Dipterans of the family Tipulidae were collected at all sites, primarily in cores. They were collected throughout the study, but rarely were abundant.

Polypedilum halterale, another chironomid, was encountered in substantial numbers in the reference wetland and rarely elsewhere. The fact that this wetland has its natural hydroperiod and a general lack of large numbers of vertebrate predators could explain why this species was abundant.

Georthocladius, another chironomid, was rarely found at any site, with the exception of Site 2 downstream from November 1998 to March 1999. Site 2 downstream differed from all other stations in that it was heavily wooded, primarily with slash pine, *Pinus elliottii*, and the sediment was very moist, poorly decomposed organic matter. It is possible this could explain its presence at this station but not elsewhere.

Unlike zooplankton, benthic macroinvertebrate communities in southern U.S. wetlands do not appear to be influenced much by seasonality and this may be a reflection of mild winters (Leslie *et al.* 1999; Porter *et al.* 1999; Prenger *et al.* in prep.). Instead, any temporal population changes may be as a result of hydrological influences.

Water level did not appear to be correlated with invertebrate taxon richness or density. This is not surprising, as the dominant numerical taxa in Tates Hell Swamp (*Caecidotea*, *Crangonyx*, *Culicoides*, *Polypedilum* spp., *Procladius*, and *Ablabesmia*) are all capable of remaining *in situ* to survive drought periods (Wiggins *et al.* 1980; Williams 1997) (Taylor *et al.* 1999). A lack of correlation between density and water level was also noted by Leslie (1997), who found that the major taxa in her study of cypress domes in north central Florida were all capable of surviving *in situ* under drought conditions.

Despite the lack of correlation between water level and richness, there is a very obvious trend of decreased densities of invertebrates in cores during the severe drought of summer 1998. The environmental conditions present in the wetlands at this time appeared to have very significant effects on the densities of most taxa. This differs from the lack of water level vs. density correlation discussed above in that the previous measure addressed small changes in water level (0-20 cm) and may not have been sensitive enough to show changes on the order of wet vs. extreme drought.

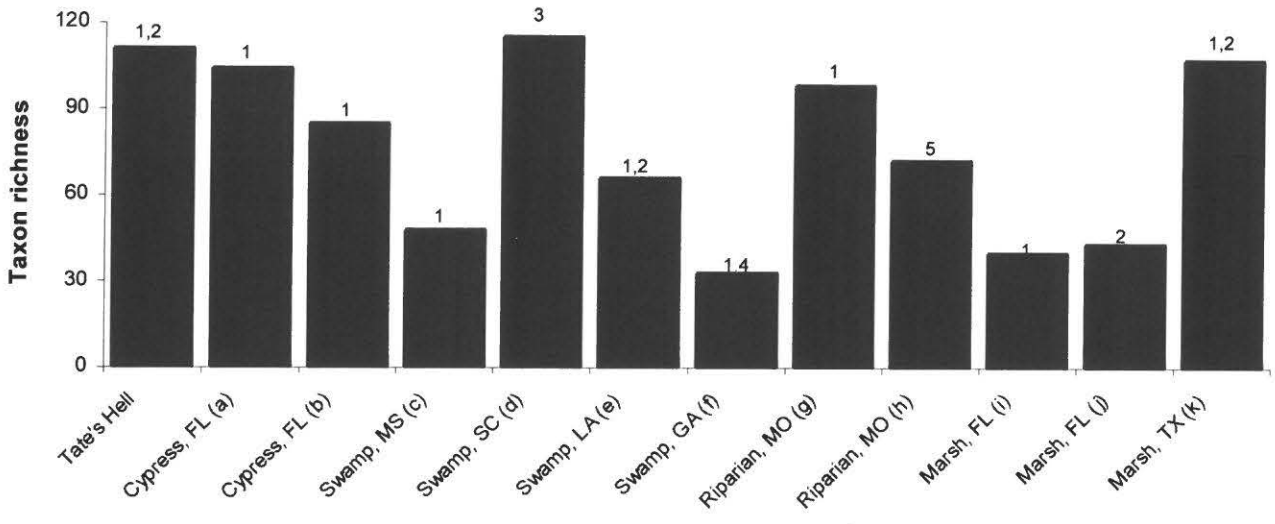


Figure 5-1. Benthic macroinvertebrate taxon richness from wetlands in the United States. Numbers indicate method of capture (1, core; 2, sweep net; 3, emergence trap; 4 wood; 5, substrate). a (Prenger *et al.*, in prep), b (Leslie *et al.* 1997), c (Duffy and LaBar 1994), d (Leeper and Taylor 1998), e (Sklar 1985), f (Golladay *al.* 1997), g (Corti *et al.* 1997), h (White 1985), i (Evans 1996), j (Evans 1996), k (Moorhead *et al.* 1998).

Table 5-1. Benthic macroinvertebrates collected from wetlands in Tates Hell Swamp using cores and sweep nets.

Taxonomic Group	Method of Collection		Trophic Guild
	Core	Sweep	
Class Oligochaeta	x		Collectors-gatherers
Class Arachnida			
Order Hydracarina	x	x	Predators
Class Crustacea			
Order Amphipoda			
Family Gammaridae			
Crangonyx	x	x	Shredders-herbivores
Order Isopoda			
Family Assellidae			
Caecidotea	x	x	Shredders-herbivores
Order Copepoda	x	x	Collectors-filterers
Order Decapoda			
Family Cambaridae	x	x	Collectors-predators
Class Insecta			
Order Collembola			
Family Entomobryidae	x	x	Collectors-gatherers
Family Isotomidae	x		Collectors-gatherers
Family Sminthuridae	x	x	Collectors-gatherers
Order Coleoptera			
Family Chrysomelidae	x	x	
Family Dytiscidae			
Agabetes (larvae)		x	Predators
Agabus (larvae)	x	x	Predators
Agaporomorphus (adult)	x		Predators
Bidessonotus (adult)		x	Predators
Copelatus (adult)	x	x	Predators
Coptotomus (larvae)	x	x	Predators
Coptotomus (adult)		x	Predators
Celina (larvae)	x	x	Predators
Celina (adult)	x	x	Predators
Cybister (larvae)	x	x	Predators
Desmopachria (adult)	x	x	Predators
Derovatellus (adult)	x	x	Predators
Eretes (adult)	x		Predators
Hydaticus (larvae)		x	Predators
Hydroporus(larvae)	x	x	Predators
Laccodytes (adult)	x		Predators
Laccornis (larvae)	x	x	Predators
Liodessus (larvae)	x	x	Predators
Liodessus (adult)		x	Predators
Matus (larvae)	x	x	Predators
Matus(adult)		x	
Neoporus (larvae)	x		Predators

Table 5-1 (continued)

Taxonomic Group	Method of Collection		Trophic Guild
	Core	Sweep	
Pachydrus (adult)		x	Predators
Pachydrus (larvae)	x	x	Predators
Thermonectus (adult)		x	Predators
Uvarus (larvae)	x		Predators
Family Elmidae			
Dubiraphia (adult)	x		Collectors-gatherers
Dubiraphia (larvae)	x		Collectors-gatherers
Family Hydraenidae			
Hydraena (larvae)	x	x	Predators
Hydraena (adult)	x	x	Collectors-gatherers
Family Hydrophilidae			
Anacaena (adult)		x	Collectors-gatherers
Berosus (larvae)	x	x	Collectors-gatherers
Berosus(adult)		x	Collectors-gatherers
Derallus (adult)		x	Collectors-gatherers
Enochrus (adult)	x	x	Herbivores
Helocombus (larvae)	x		Predators
Helocombus (adult)	x		Collectors-gatherers
Hydrobiomorpha (larvae)		x	Predators
Hydrobius (larvae)	x		Collectors-gatherers
Hydrobius (adult)	x		Collectors-gatherers
Hydrochus (larvae)	x	x	Shredders-herbivores
Hydrochus (adult)	x	x	Shredders-herbivores
Hydrophilus (larvae)	x		Predators
Family Noteridae			
Hydrocanthus (adult)		x	Predators
Notomicrus (adult)	x		Predators
Family Scirtidae		x	Collectors-gatherers
Order Diptera			
Family Ceratopogonidae	x	x	
Family Chaoboridae			
Chaoborus	x	x	Predators
Family Chironomidae			
Ablabesmyia	x	x	Predators, collectors-gatherers
Chironomini genus III	x	x	
Chironomus	x	x	Collectors-gatherers
Cladopelma	x	x	Collectors-gatherers
Cladotanytarsus		x	Collectors-gatherers
Clinotanypus	x	x	Predators
Corynoneura	x		Collectors-gatherers
Cryptochironomus	x	x	Predators
Cryptotendipes	x	x	Collectors-gatherers
Dicrotendipes cf. *modestus*	x	x	Collectors-gatherers
Endochironomus cf. *subtendens*		x	Collectors-gatherers
Fittkauimyia	x	x	
Goeldichironomus holoprasinus	x	x	Collectors-gatherers
Guttipelopia	x	x	Predators
Kiefferulus	x	x	Collectors-gatherers
Labrundinia virescens	x	x	Predators

Table 5-1 (continued)

Taxonomic Group	Method of Collection		Trophic Guild
	Core	Sweep	
Limnophyes	x		Collectors-gatherers
Monopelopia boliekae	x	x	Predators
Natarsia	x		Predators
Nilothalma	x		Collectors-gatherers
Orthocladius	x	x	Collectors-gatherers
Parachironomus carinatus		x	Predators, collectors-gatherers
Parachironomus alatus	x	x	Predators, collectors-gatherers
Parachironomus hirtalatus	x	x	Predators, collectors-gatherers
Paratendipes subaequalis	x	x	Collectors-gatherers
Polypedilum laetum	x	x	Collectors-gatherers
Polypedilum fallax	x	x	Collectors-gatherers
Polypedilum trigonus	x	x	Collectors-gatherers
Polypedilum tritum	x	x	Collectors-gatherers
Procladius	x	x	Predators
Psectrocladius	x	x	Collectors-gatherers
Pseudochironomus	x		Collectors-gatherers
Smittia	x		Collectors-gatherers
Tanypus cf. *carinatus*	x	x	Predators
Tanytarsus	x	x	Collectors-gatherers
Zavreliella marmorata		x	Collectors-gatherers
Family Tabanidae	x	x	
Family Tipulidae	x	x	
Order Ephemeroptera			
Family Baetidae			
Baetis		x	Collectors-gatherers
Family Caenidae			
Caenis	x	x	Collectors-gatherers
Order Hemiptera			
Family Belostomatidae			
Belastoma		x	Predators
Lethocerus		x	Predators
Family Corixidae			
Hesperocorixa		x	Piercers-herbivores
Trichcorixa		x	Predators
Family Mesoveliidae			
Mesovelia	x		Predators
Family Naucoridae			
Pelocoris		x	
Order Lepidoptera			
Family Noctuidae	x		Shredders-herbivores
Family Pyralidae	x	x	Shredders-herbivores
Order Megaloptera			
Family Sialidae			
Sialis	x		Predators
Order Odonata			
Family Coenagrionidae	x	x	Predators
Family Corduliidae			
Didymops		x	Predators
Epitheca		x	Predators

Table 5-1 (continued)

Taxonomic Group	Method of Collection		Trophic Guild
	Core	Sweep	
Family Gomphidae			
Aphylla		x	Predators
Arigomphus		x	Predators
Family Lestidae		x	Predators
Family Libellulidae			
Celithemis		x	Predators
Idiataphe	x	x	Predators
Libellula	x	x	Predators
Epitheca	x		Predators
Pachydiplax	x	x	Predators
Plathemis		x	Predators
Order Trichoptera	x	x	

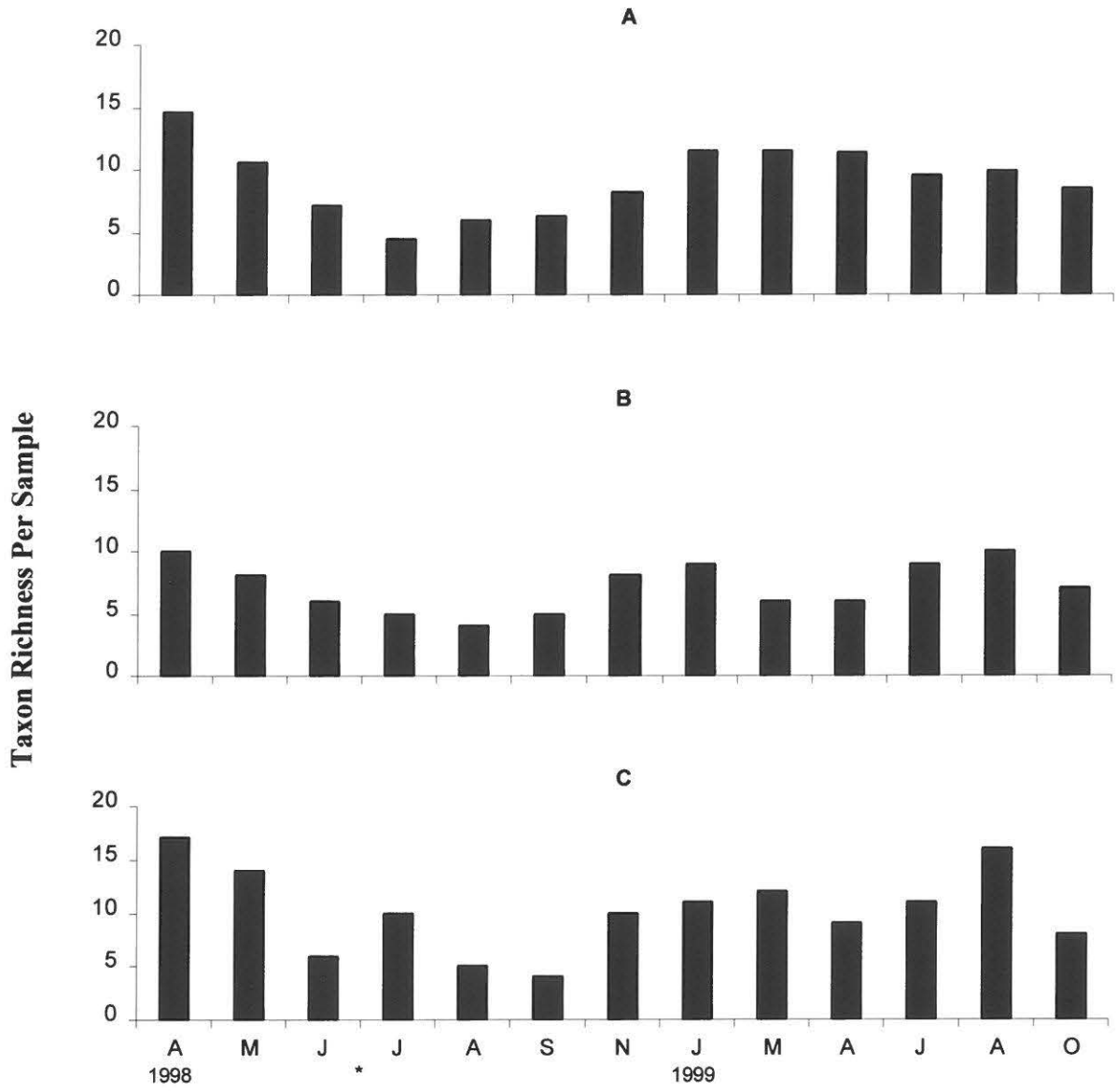


Figure 5-2. Mean taxon richness of total benthic macroinvertebrates collected from three habitats in Tates Hell Swamp using cores. A- demonstration wetlands, B- control wetland and C- reference wetland. *indicates initiation of restoration.

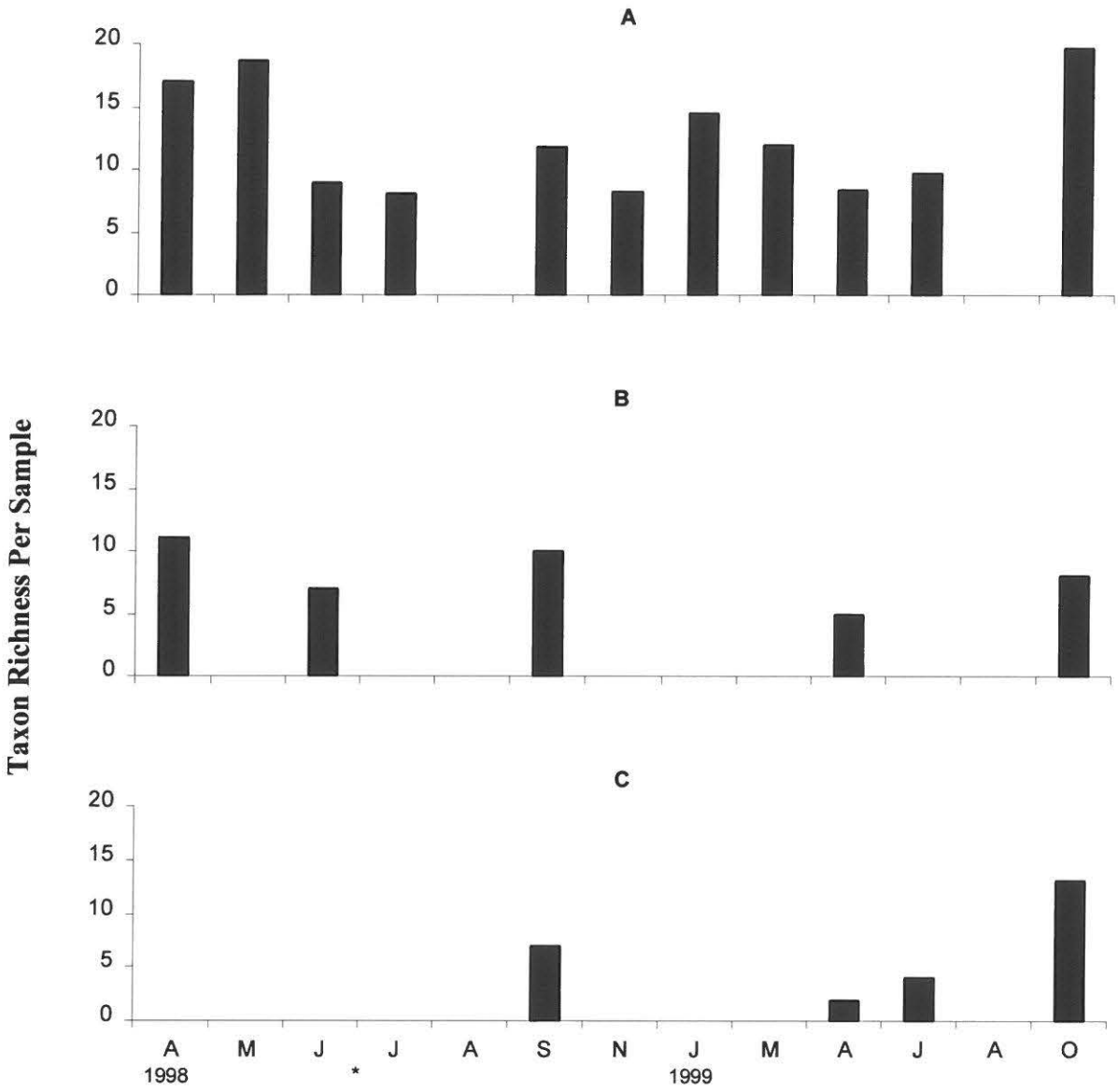


Figure 5-3. Mean taxon richness of total benthic macroinvertebrates collected from three habitats in Tates Hell Swamp using sweep nets. A- demonstration wetlands, B- control wetland and C- reference wetland.
*indicates initiation of restoration.

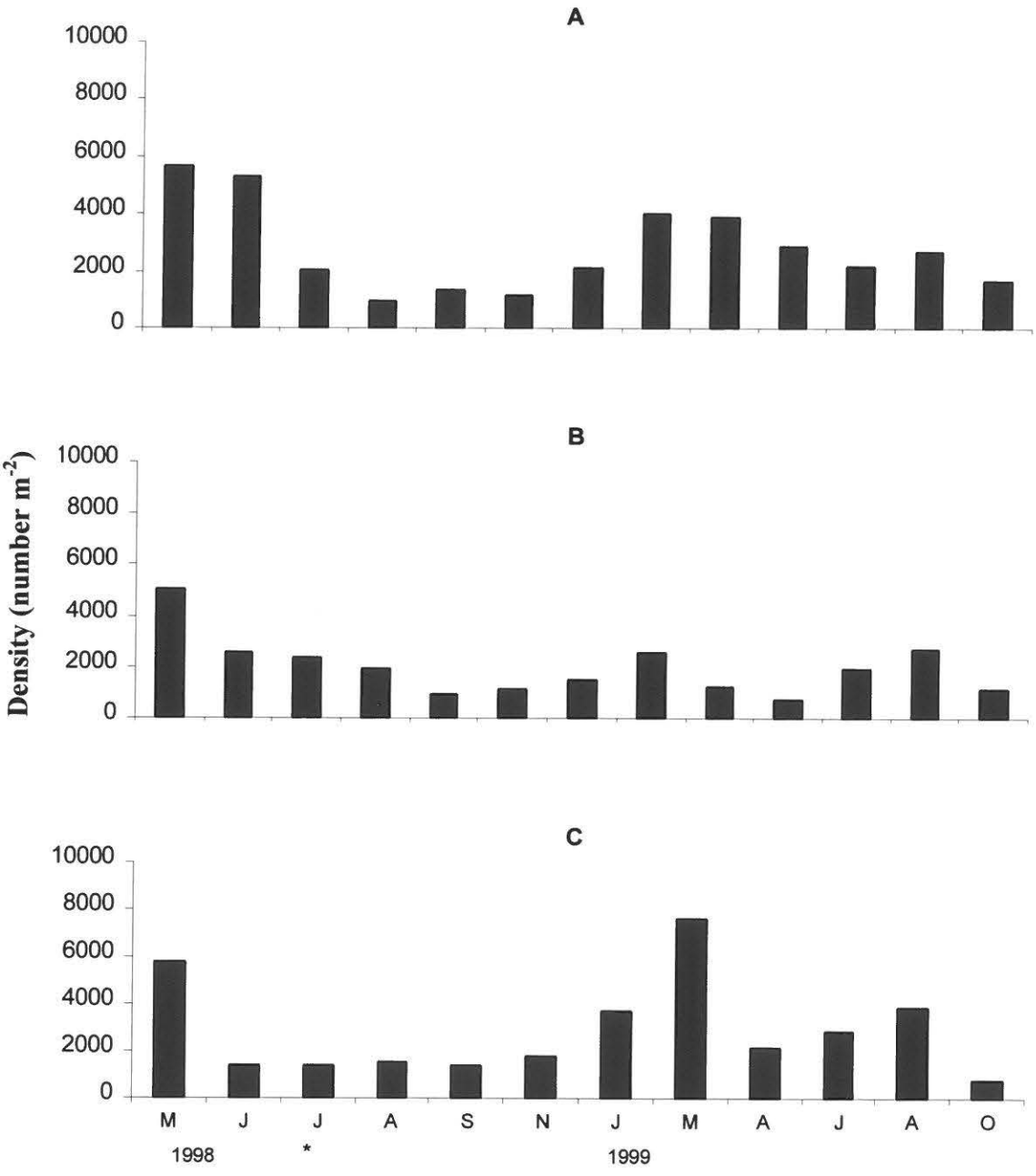


Figure 5-4. Mean density of total benthic macroinvertebrates collected from three habitats in Tates Hell Swamp using cores. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

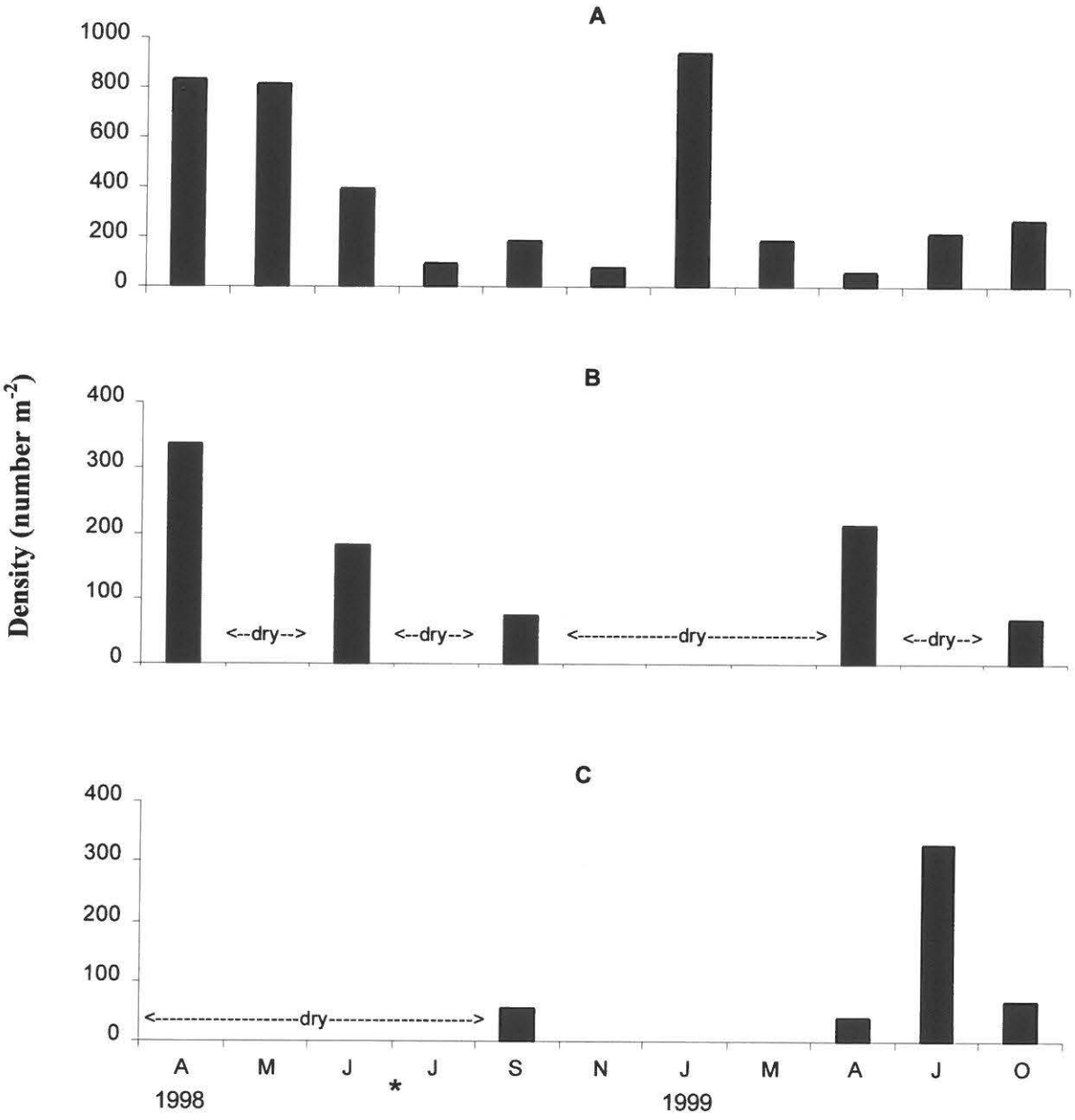


Figure 5-5. Mean density of total benthic macroinvertebrates collected from three habitats in Tates Hell Swamp using sweep nets. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

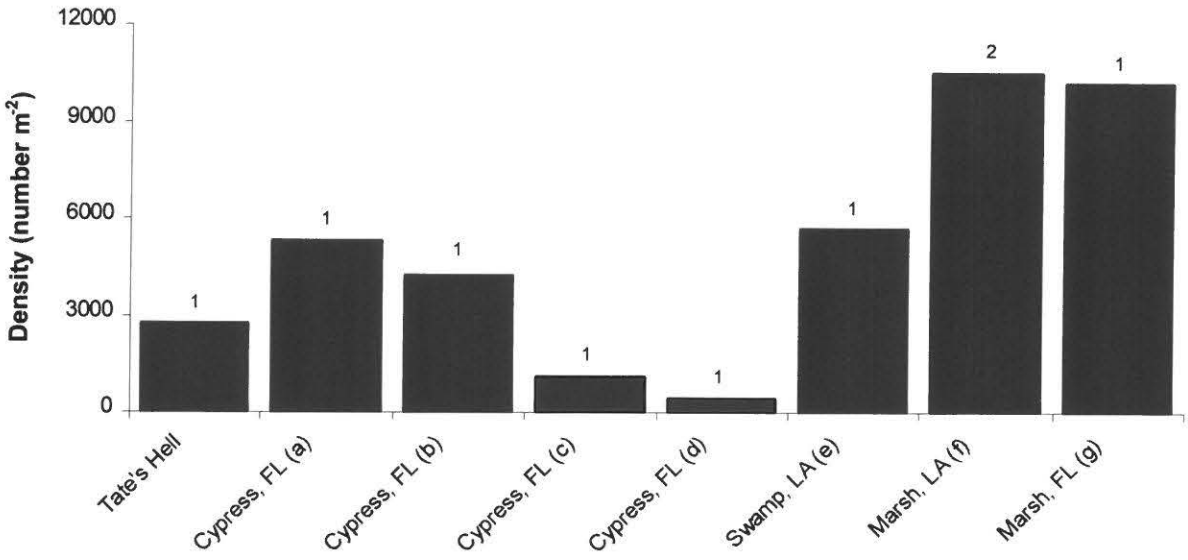


Figure 5-6. Benthic macroinvertebrate density collected from wetlands in the southeastern United States. Numbers indicate method of capture (1, core; 2, sweep net). a (Prenger *et al.* in prep), b (Leslie *et al.* 1997), c (Brightman 1984), d (Haack 1984), e (Sklar 1985), f (Sklar 1985), g (Evans 1996).

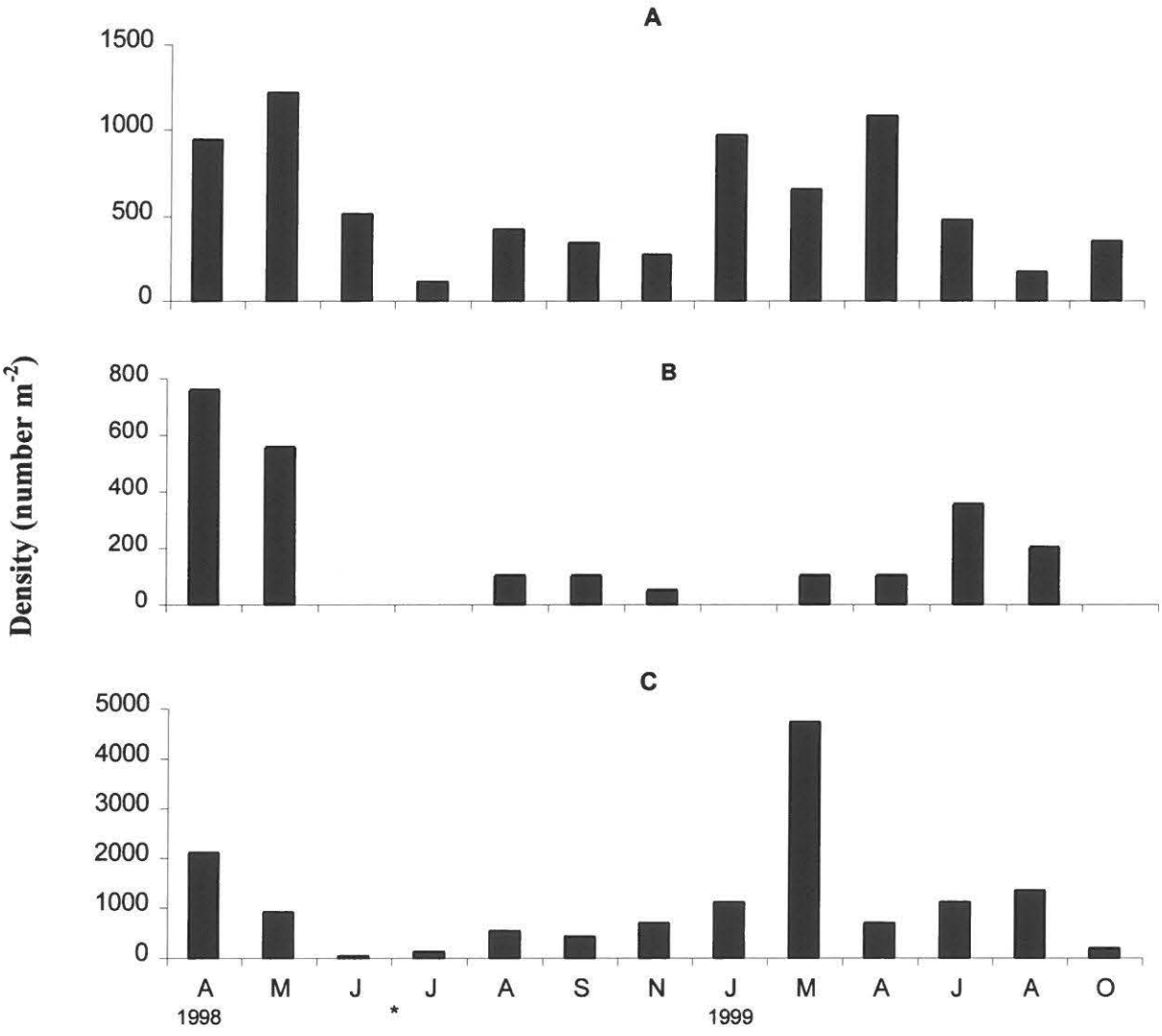


Figure 5-7. Mean density of *Crangonyx* collected from three habitats in Tates Hell Swamp using corers. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

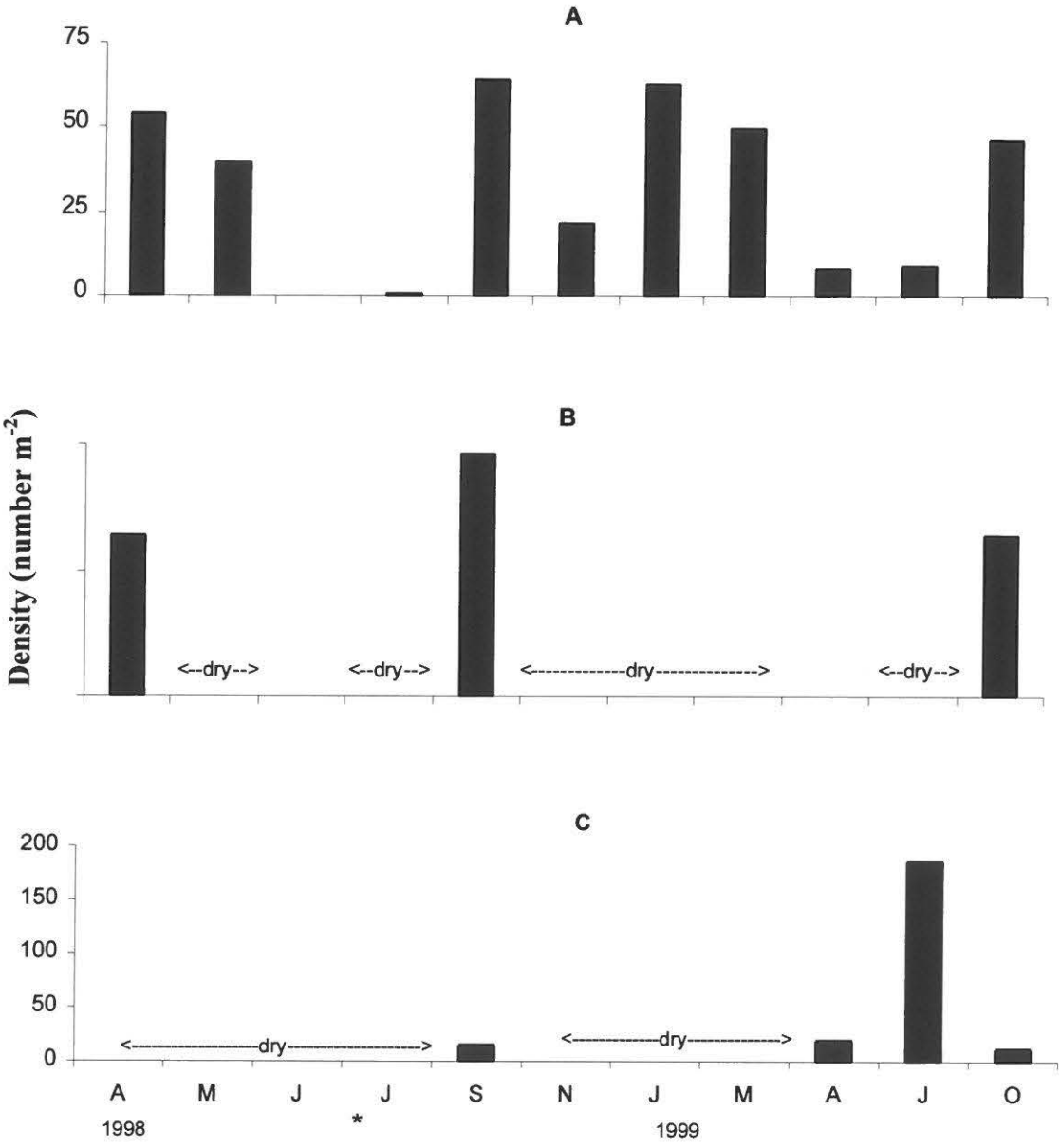


Figure 5-8. Mean density of *Crangonyx* collected from three habitats in Tates Hell Swamp using sweep nets. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

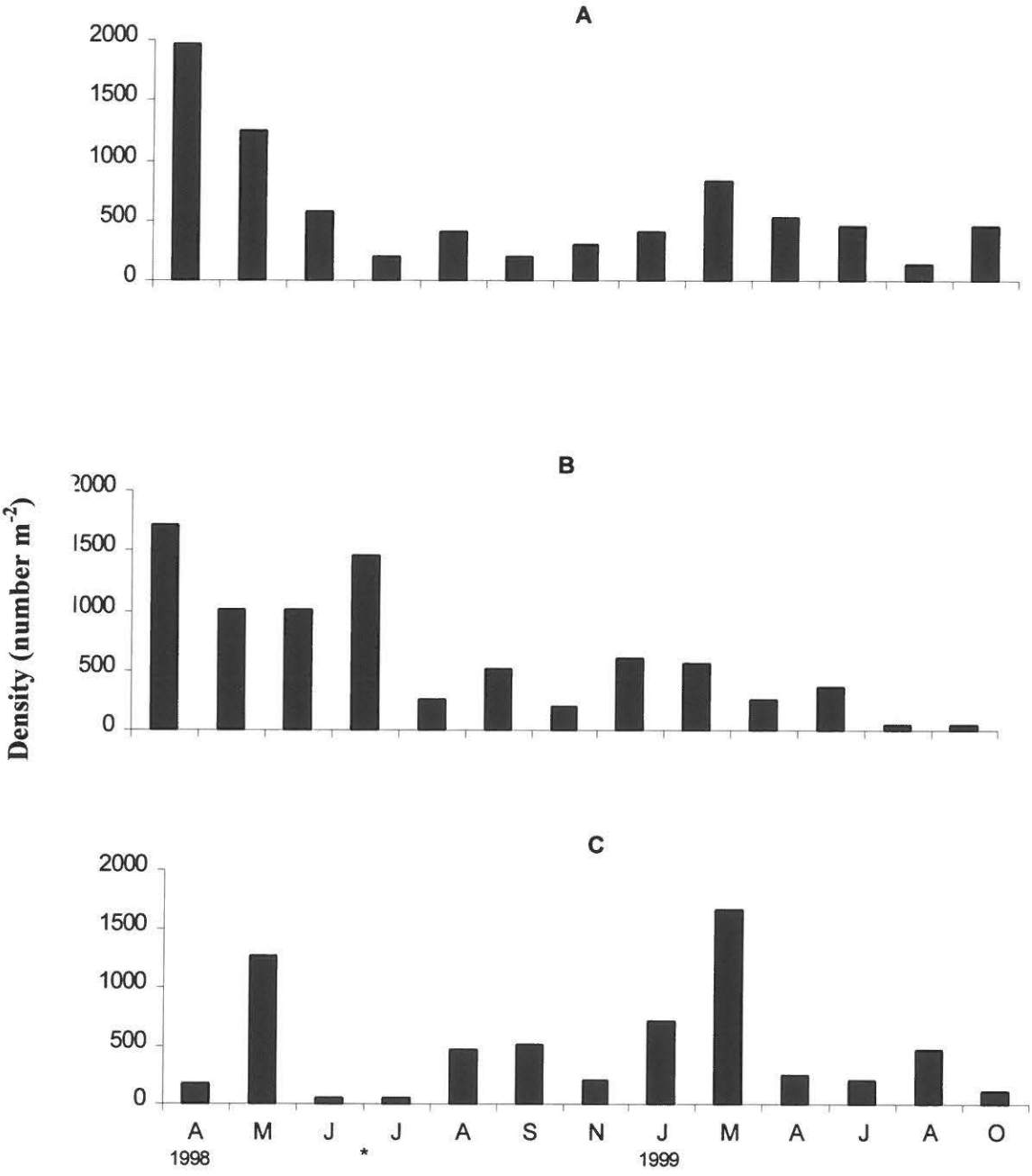


Figure 5-9. Mean density of *Caecidotea* collected from three habitats in Tates Hell Swamp using corers. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration

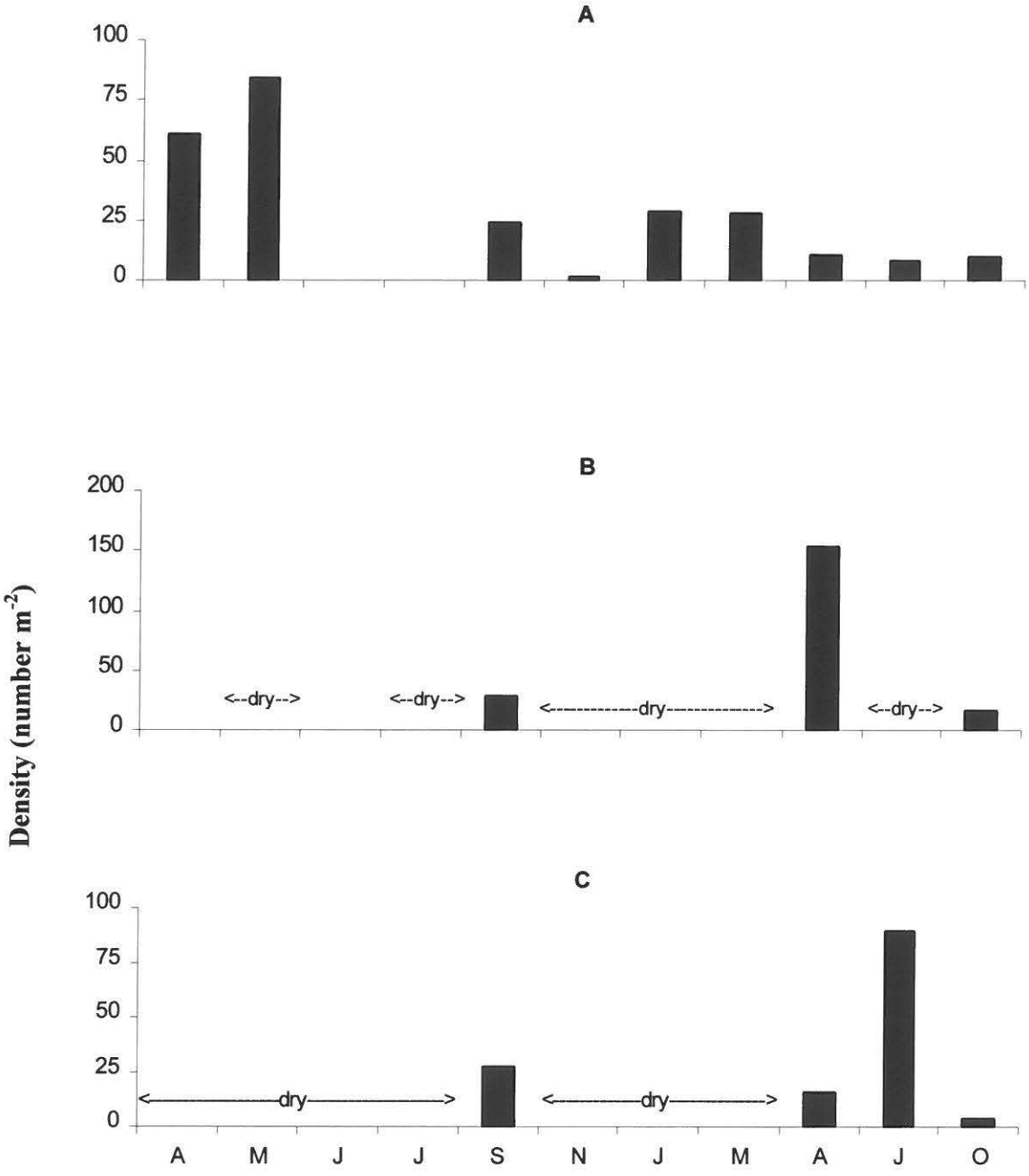


Figure 5-10. Mean density of *Caecidotea* collected from three habitats in Tates Hell Swamp using sweep nets. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

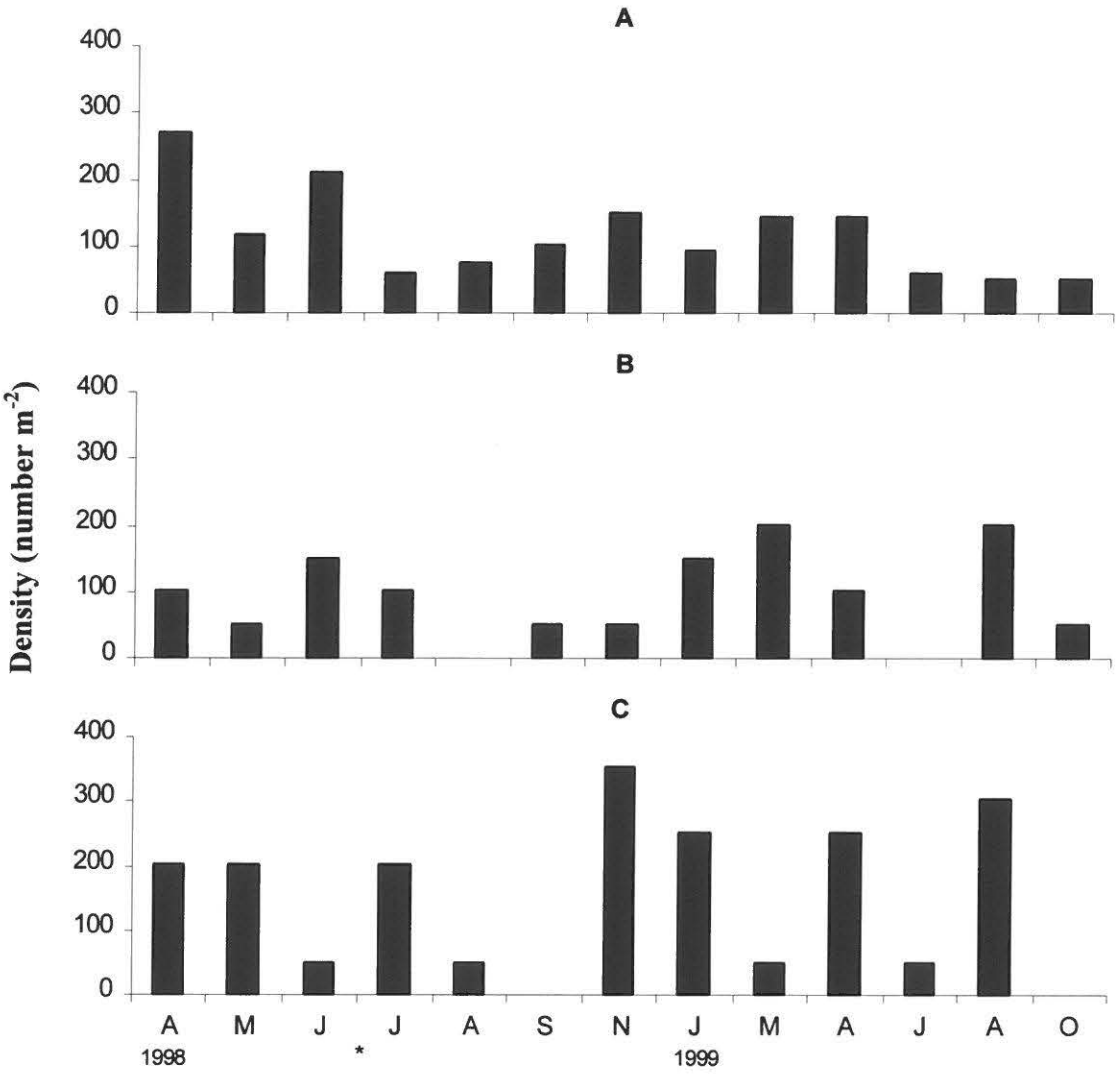


Figure 5-11. Mean density of Coleoptera collected from three habitats in Tates Hell Swamp using corers. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

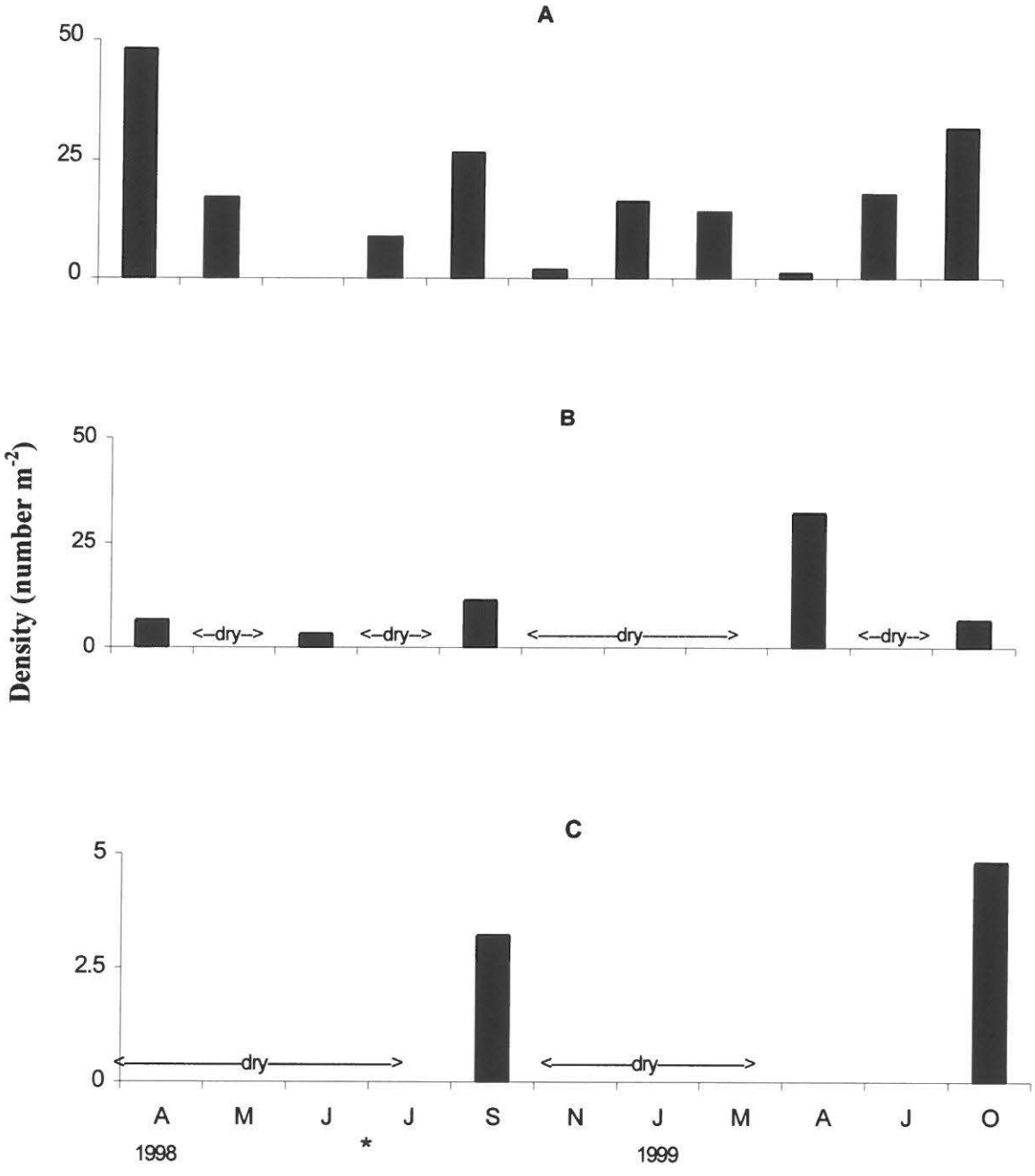


Figure 5-12. Mean density of Coleoptera collected from three habitats in Tates Hell Swamp using sweep nets. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

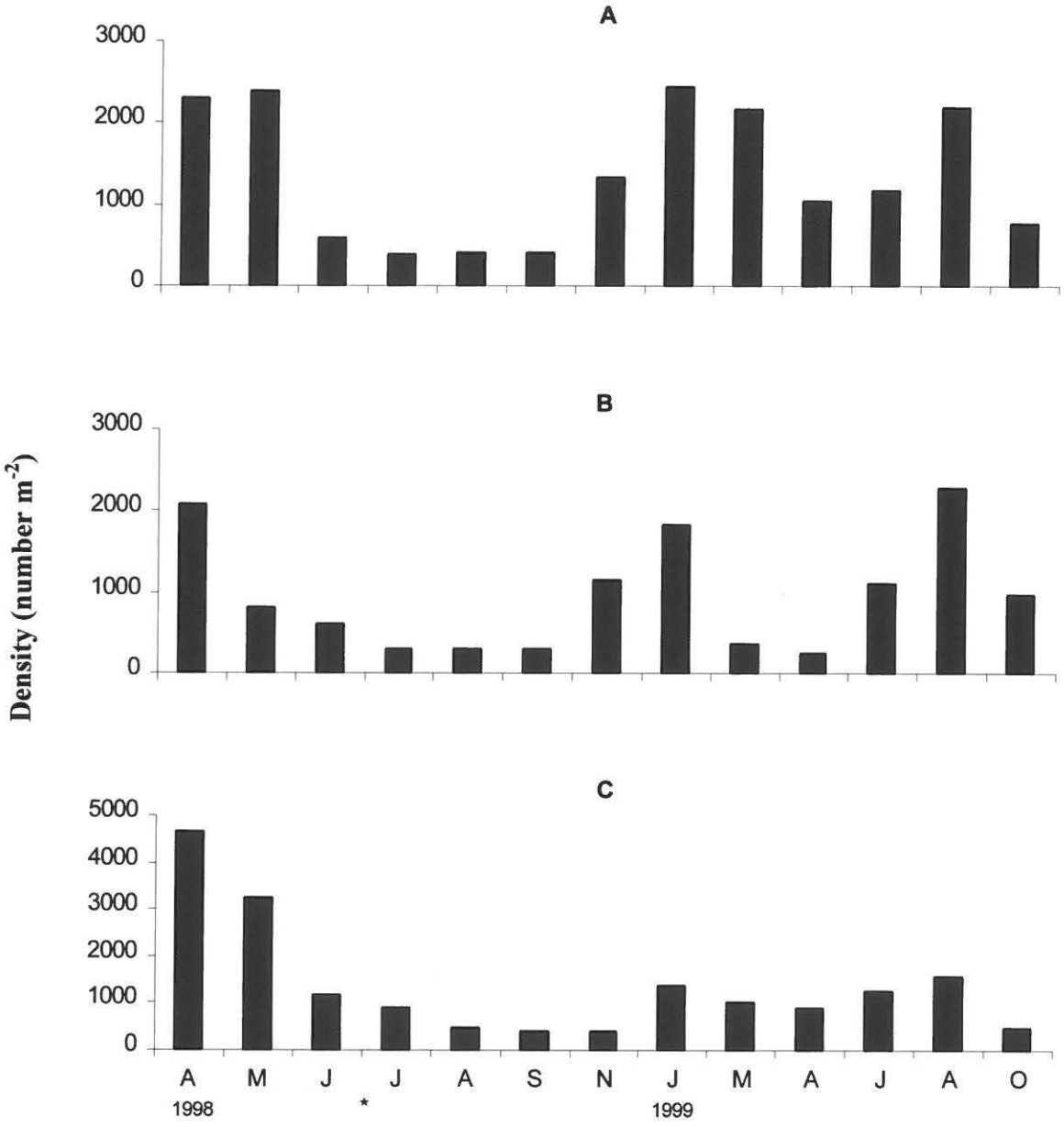


Figure 5-13. Mean density of Diptera collected from three habitats in Tates Hell Swamp using corers. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

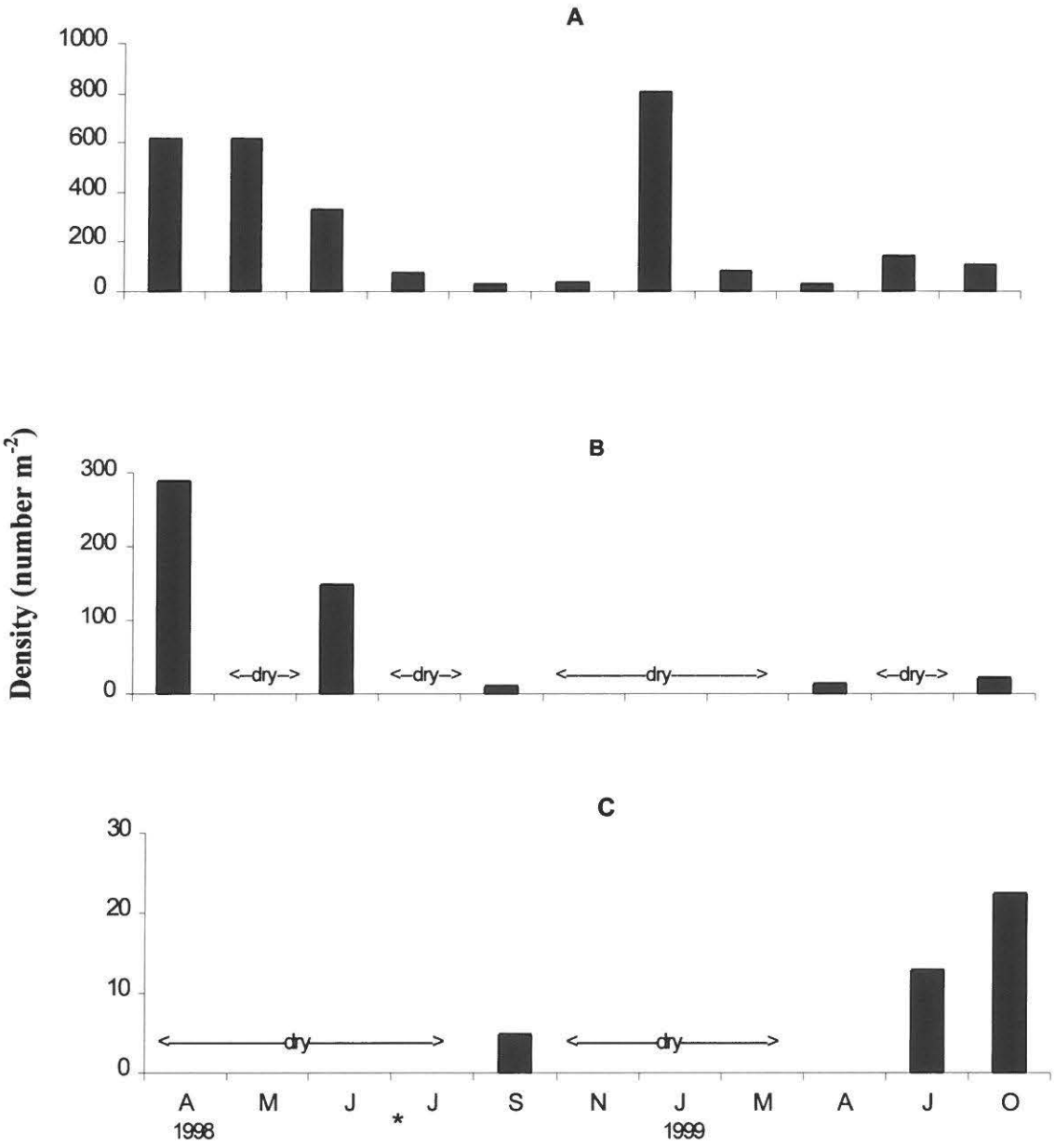


Figure 5-14. Mean density of Diptera collected from three habitats in Tates Hell Swamp using sweep nets. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

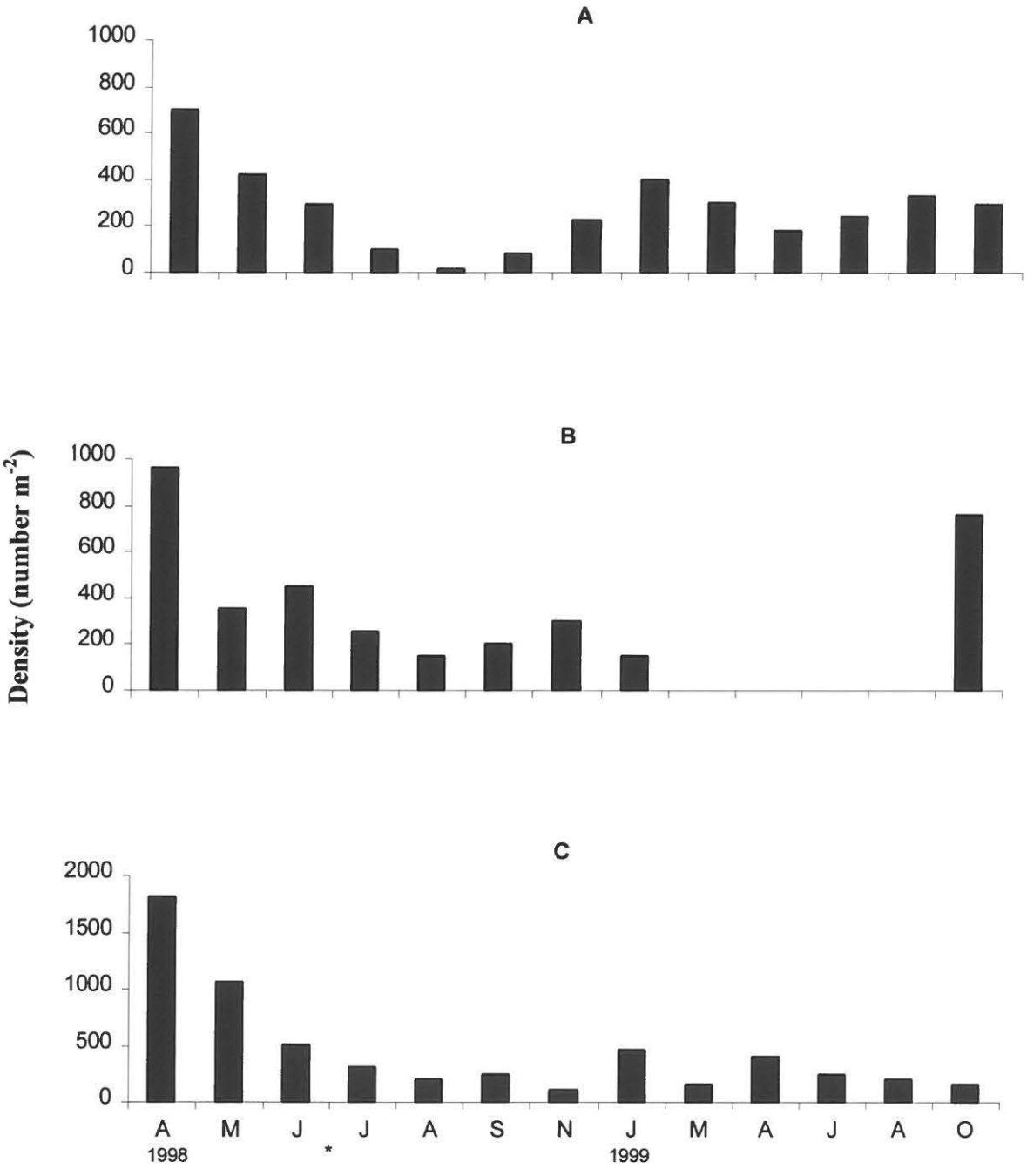


Figure 5-15. Mean density of Ceratopogonidae collected from three habitats in Tates Hell Swamp using corers. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

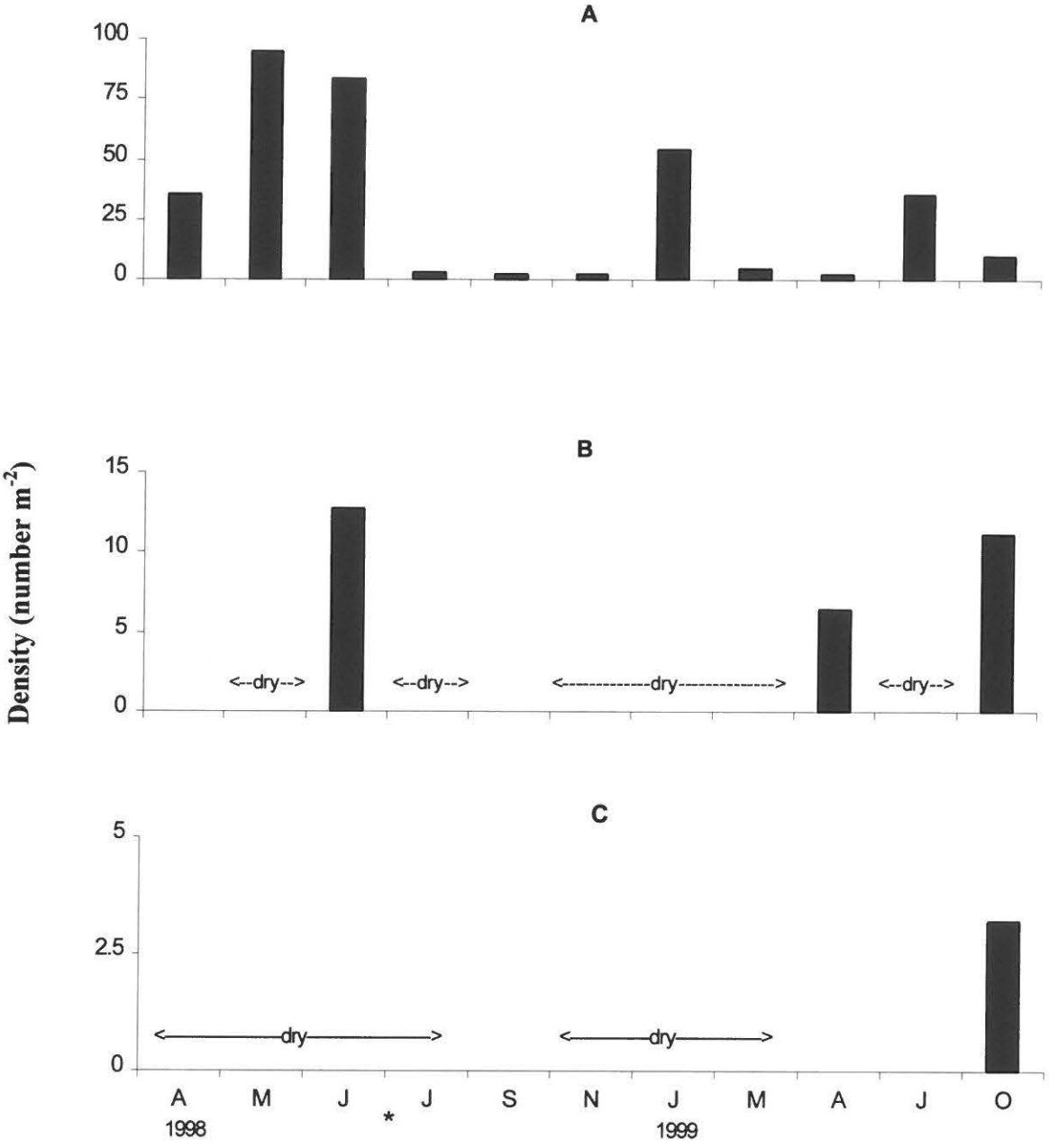


Figure 5-16. Mean density of Ceratopogonidae collected from three habitats in Tates Hell Swamp using sweep nets. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

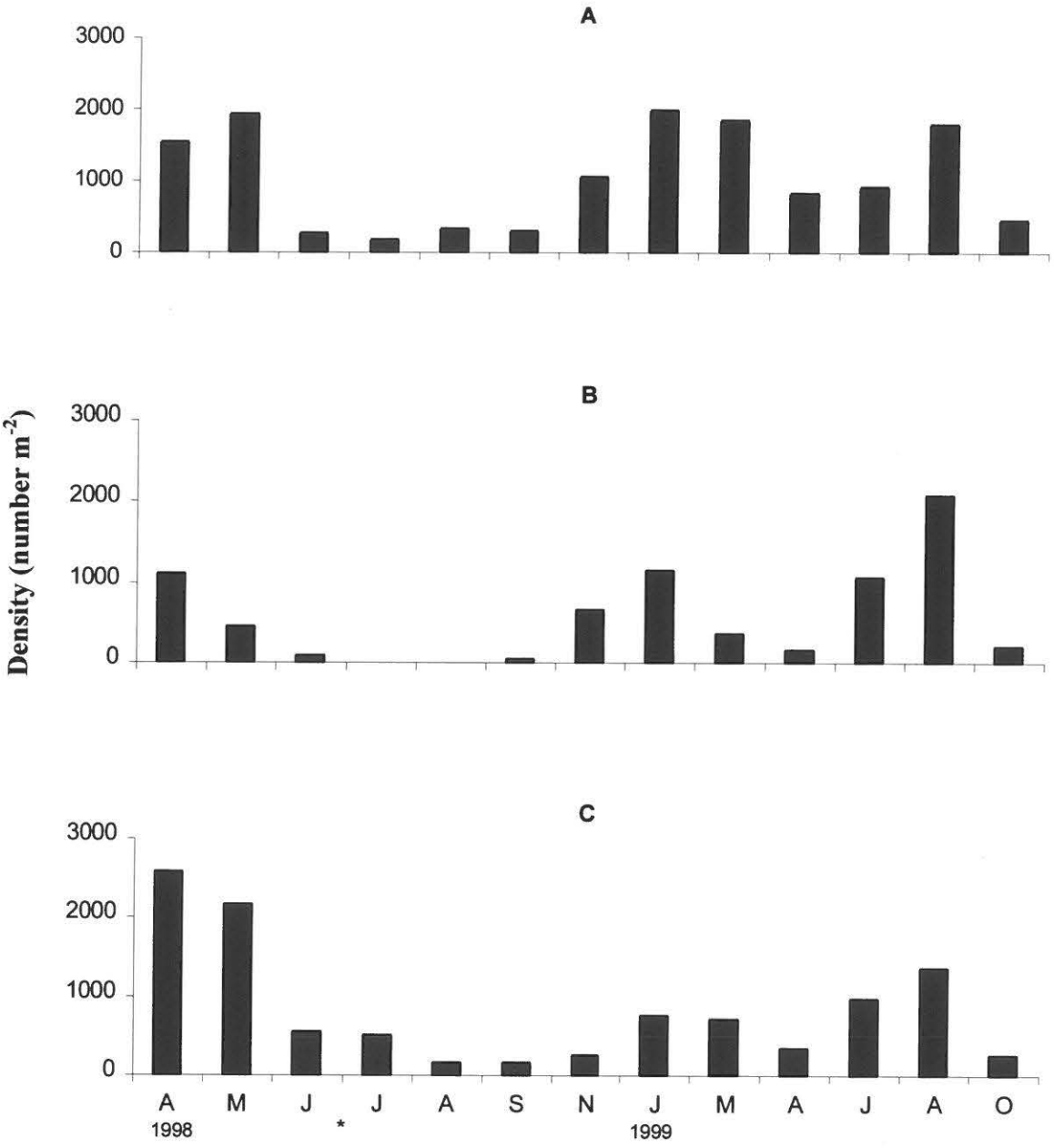


Figure 5-17. Mean density of Chironomidae collected from three habitats in Tates Hell Swamp using corers. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

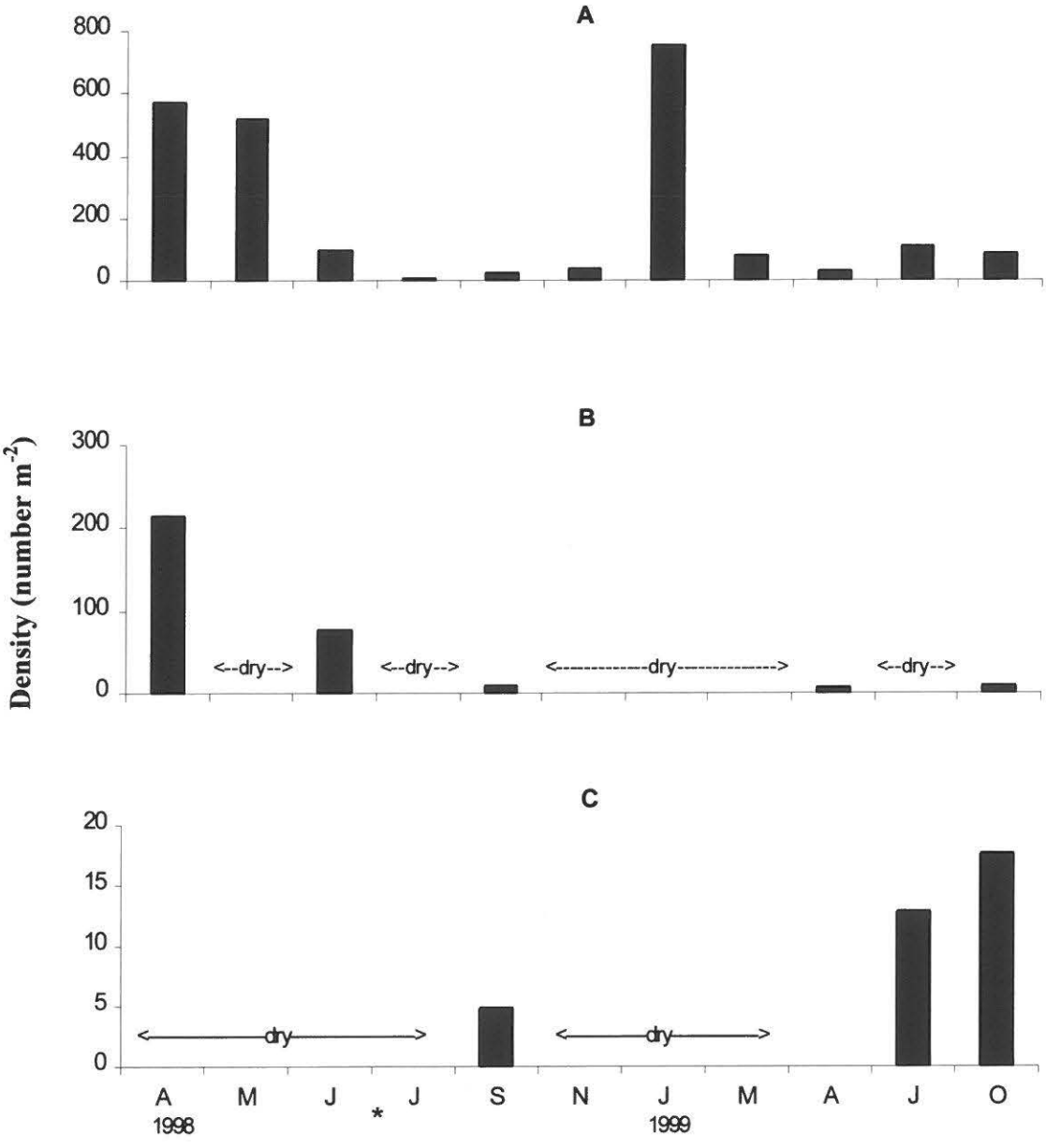


Figure 5-18. Mean density of Chironomidae collected from three habitats in Tates Hell Swamp using sweep nets. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

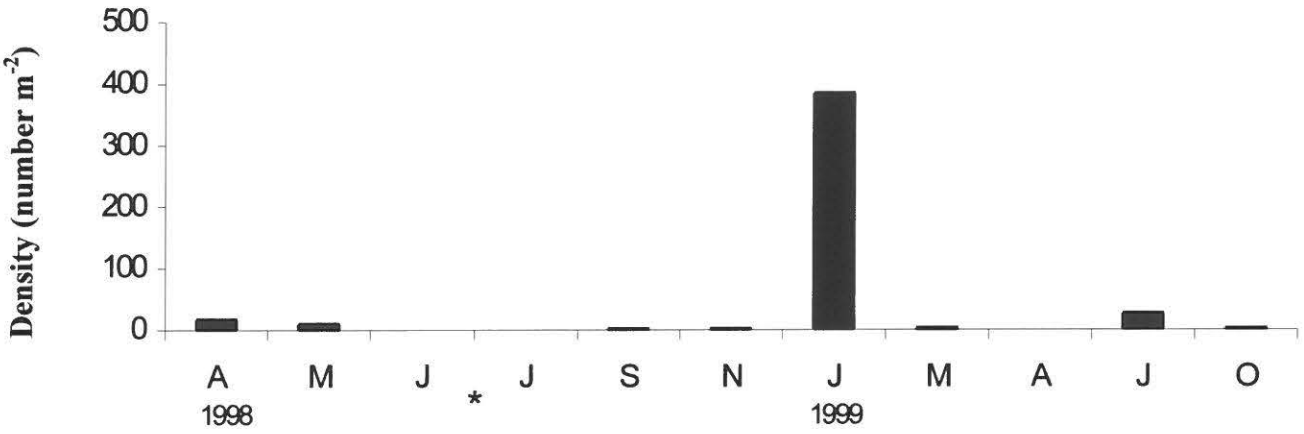


Figure 5-19. Mean density of *Polypedilum fallax* collected from demonstration wetlands in Tates Hell Swamp using sweep nets. No specimens were collected from control wetland or reference wetland. * indicates initiation of restoration.

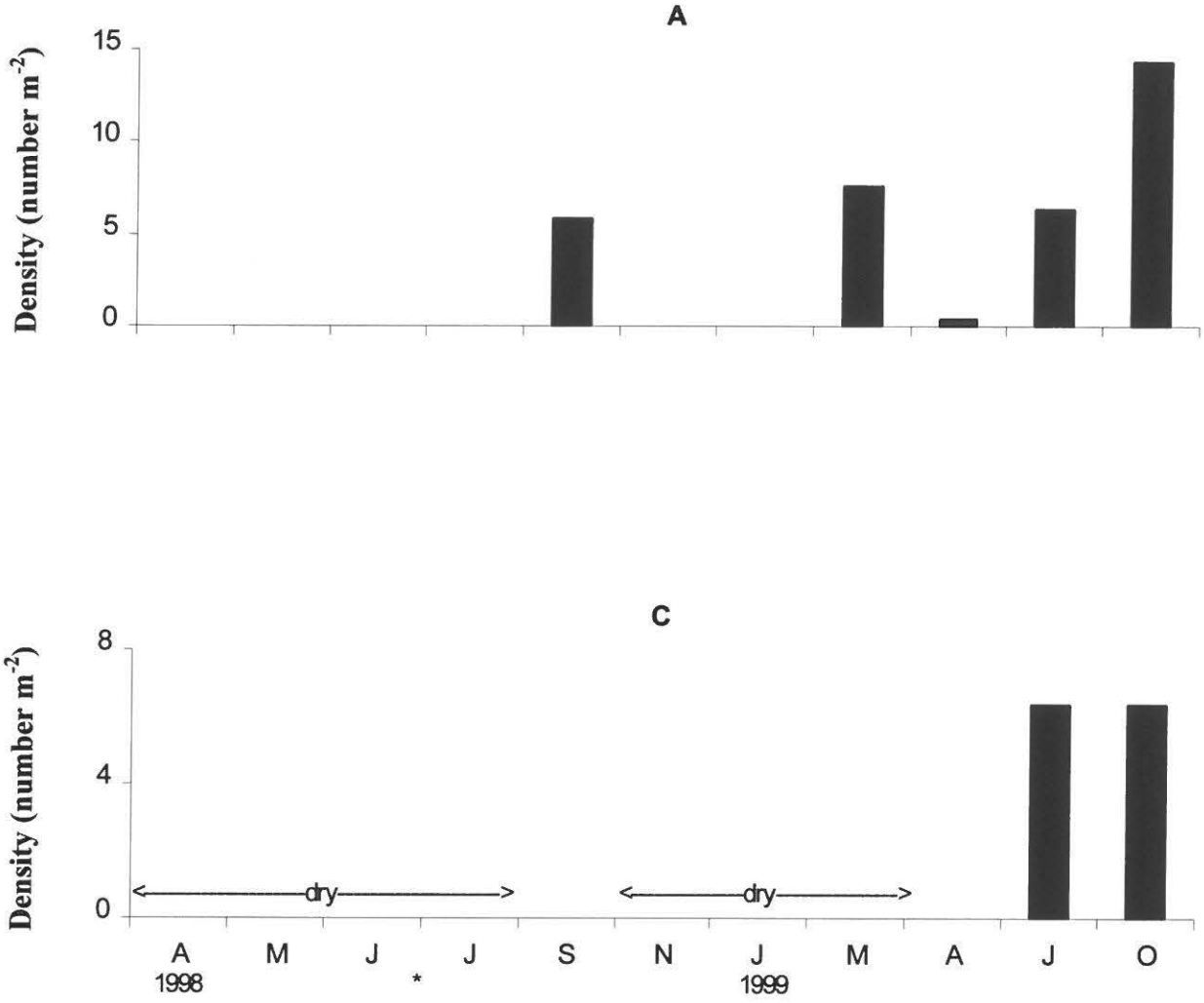


Figure 5-20. Mean density of *Polypedilum trigonus* collected from two habitats in Tates Hell Swamp using sweep nets. A- demonstration wetlands and C- reference wetland. No specimens were collected from control wetland. * indicates initiation of restoration.

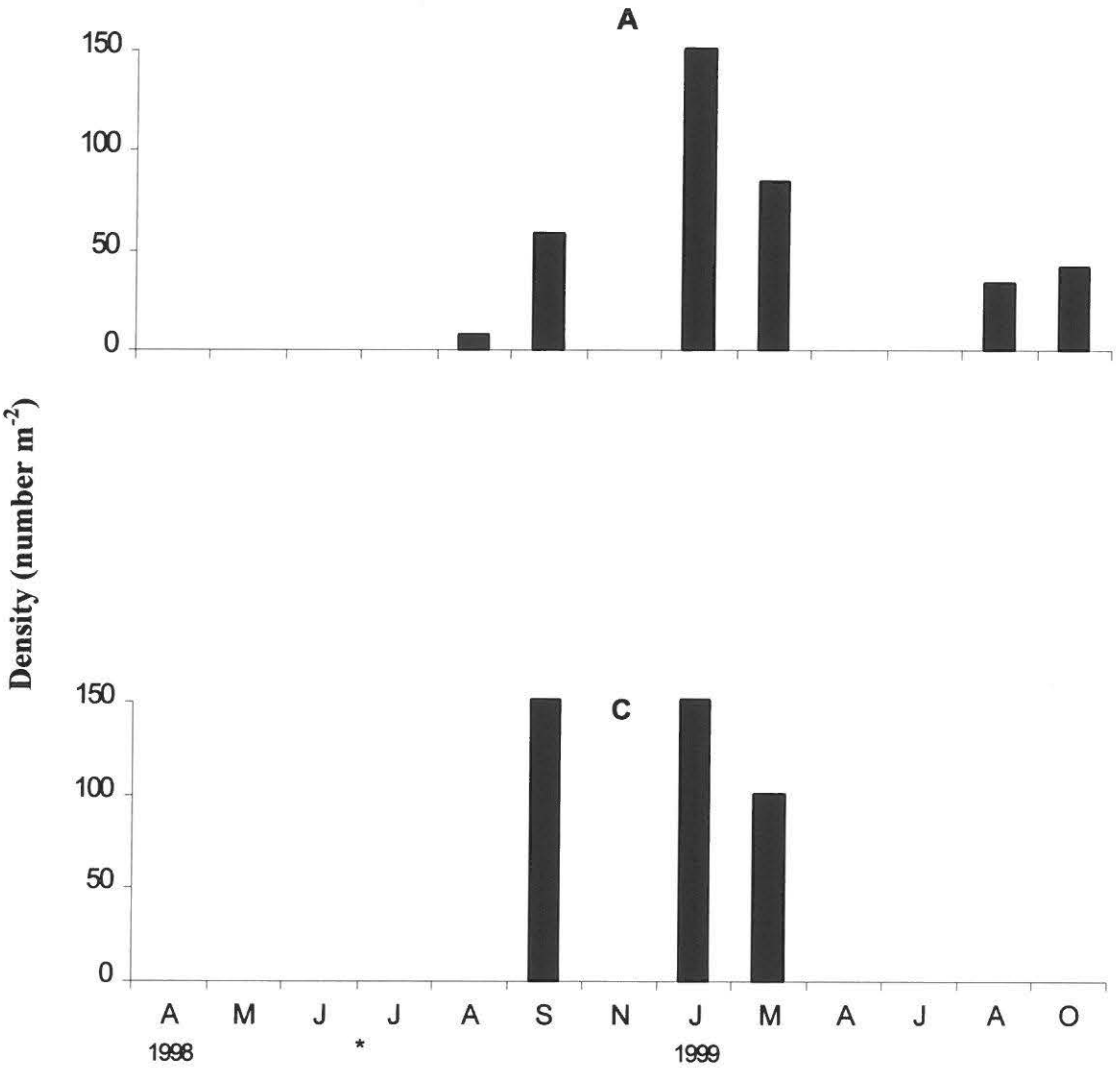


Figure 5-21. Mean density of *Polypedilum trigonus* collected from two habitats in Tates Hell Swamp using corers. A- demonstration wetlands and C- reference wetland. No specimens were collected from control wetland. * indicates initiation of restoration.

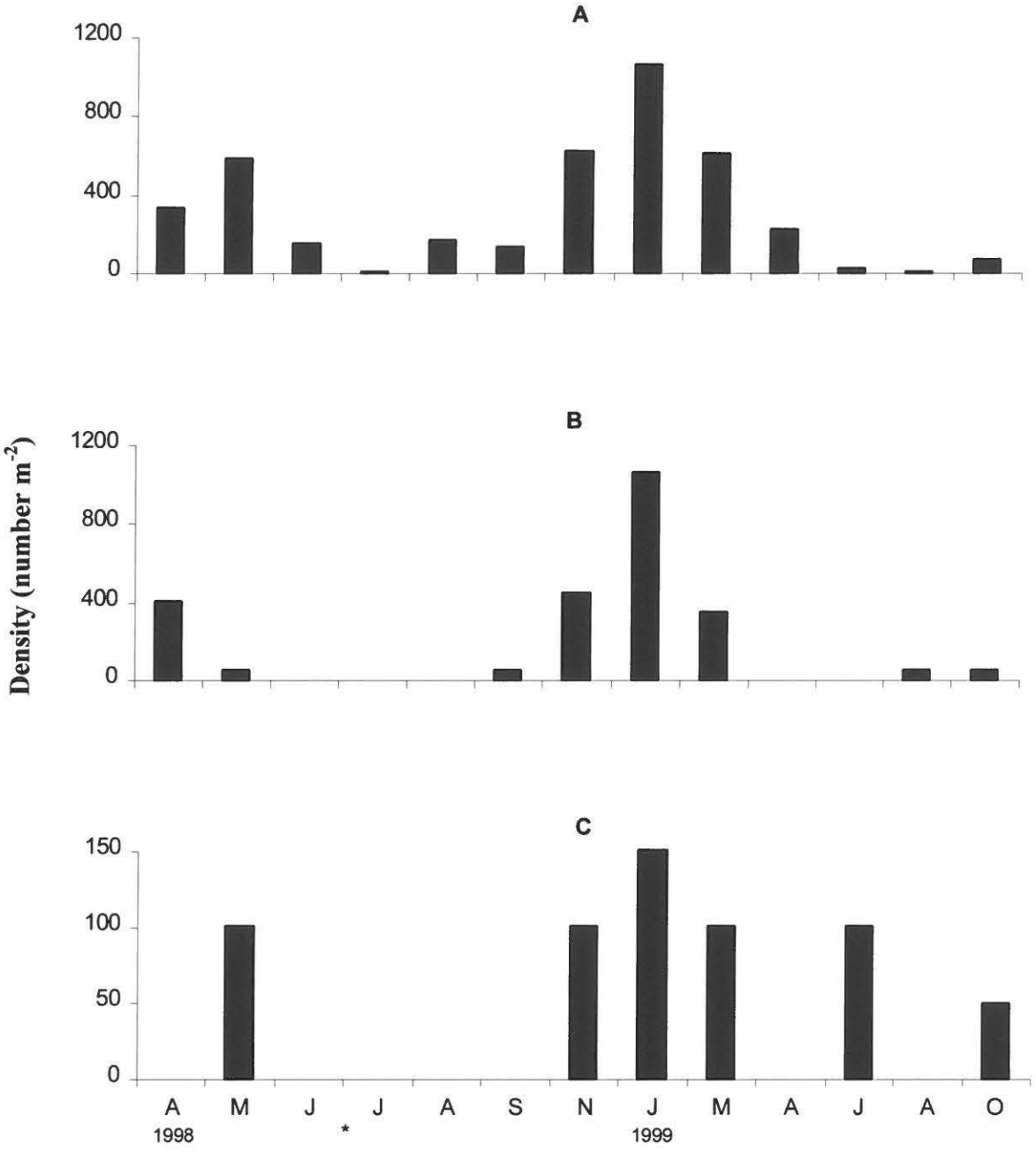


Figure 5-22. Mean density of *Polypedilum tritum* collected from three habitats in Tates Hell Swamp using corers. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

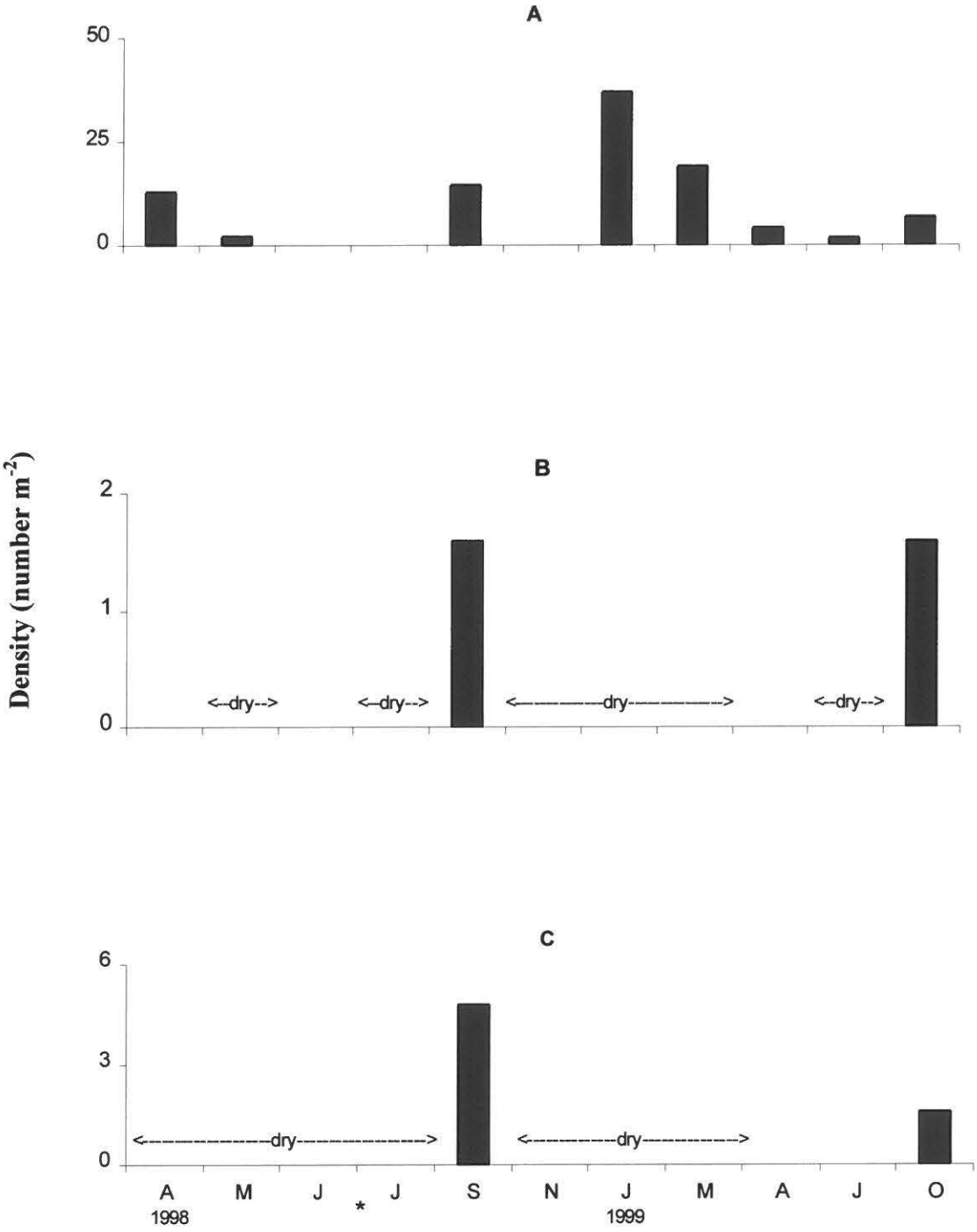


Figure 5-23. Mean density of *Polypedilum tritum* collected from three habitats in Tates Hell Swamp using sweep nets. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

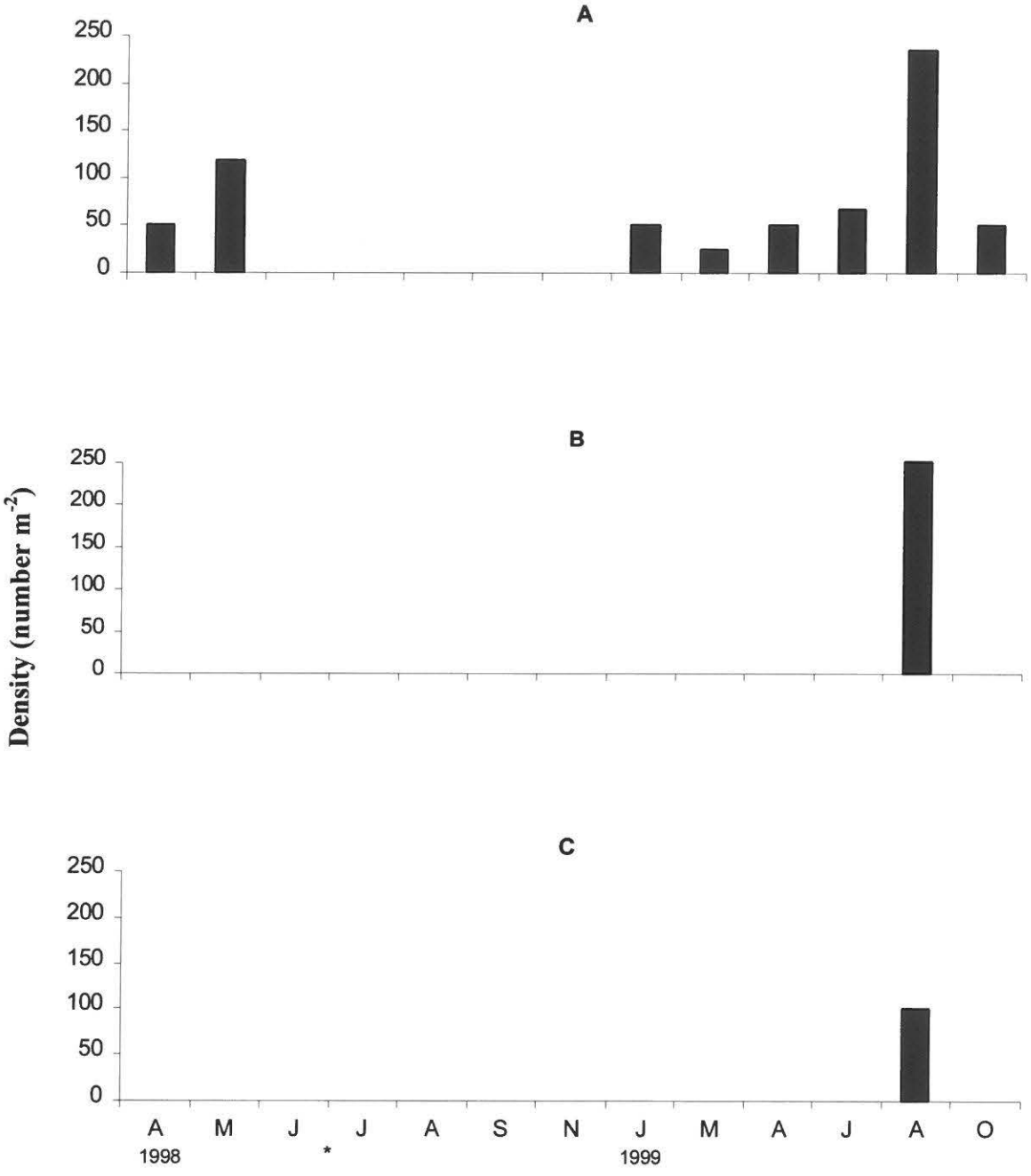


Figure 5-24. Mean density of *Chironomus* collected from three habitats in Tates Hell Swamp using corers. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

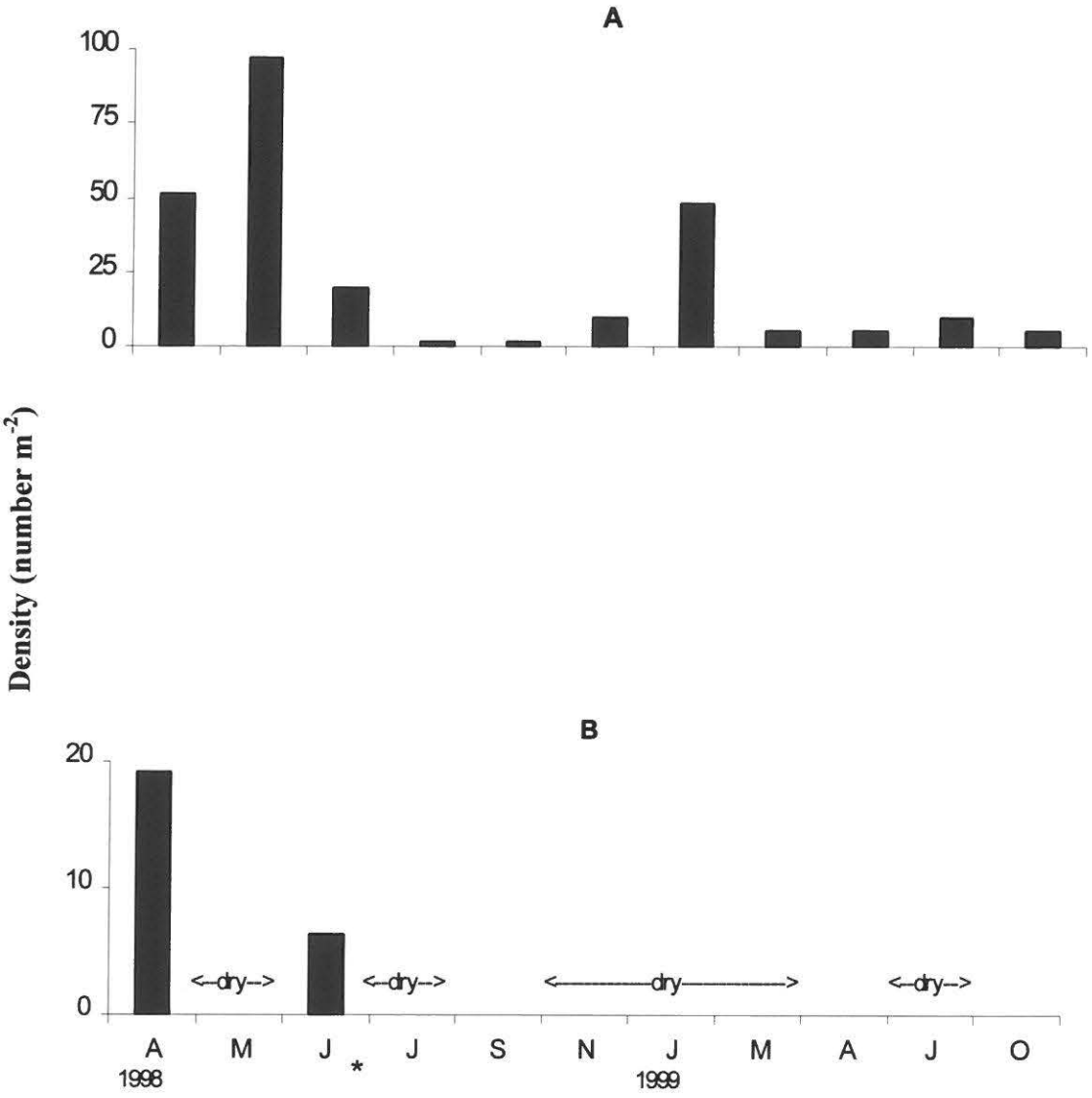


Figure 5-25. Mean density of *Chironomus* collected from two habitats in Tates Hell Swamp using sweep nets. A- demonstration wetlands and B- control wetland. No specimens were collected from the reference wetland. * indicates initiation of restoration.

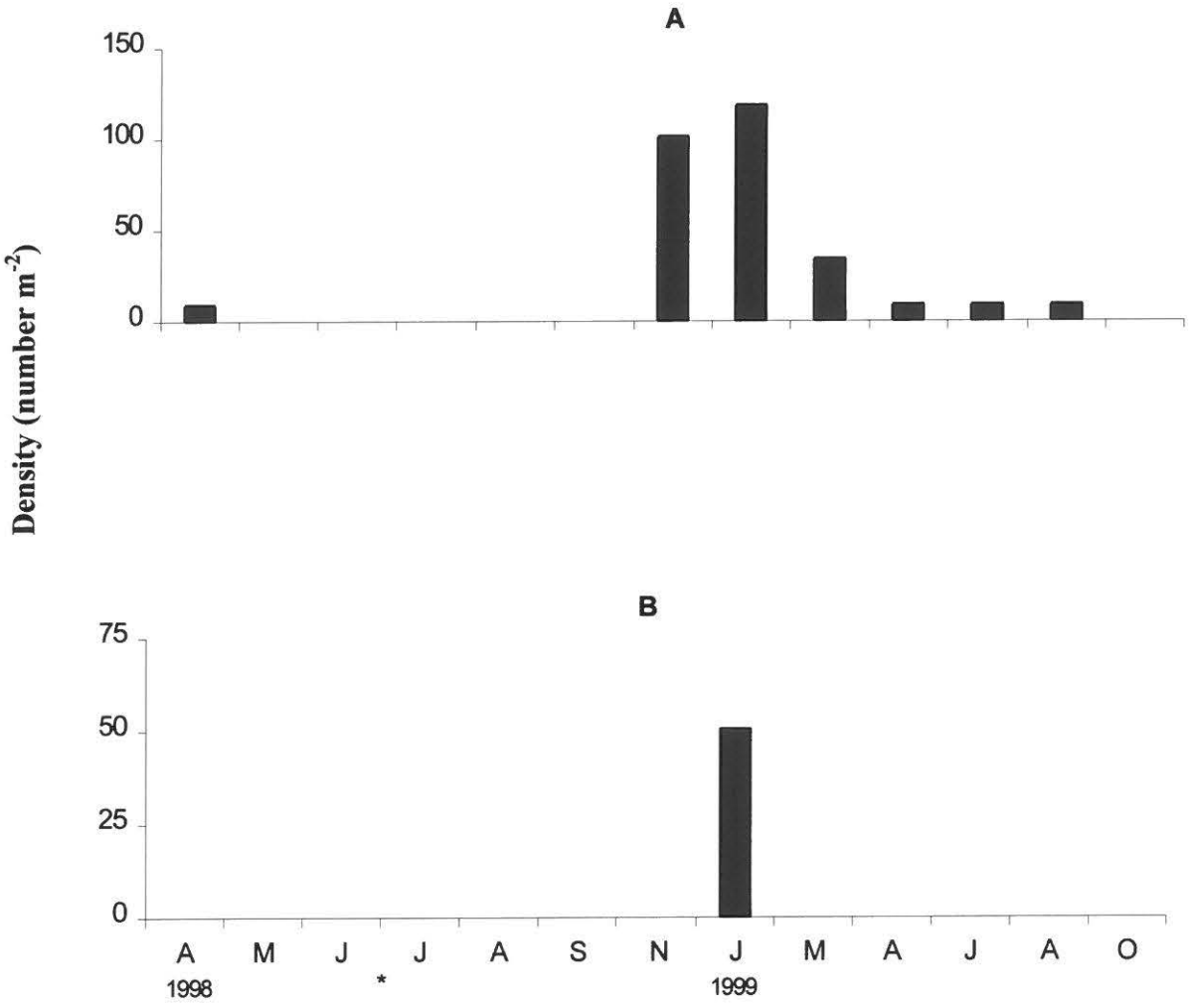


Figure 5-26. Mean density of *Tanypus* collected from two habitats in Tates Hell Swamp using corers. A- demonstration wetlands and C- control wetland. No specimens were collected from reference wetland. * indicates initiation of restoration.

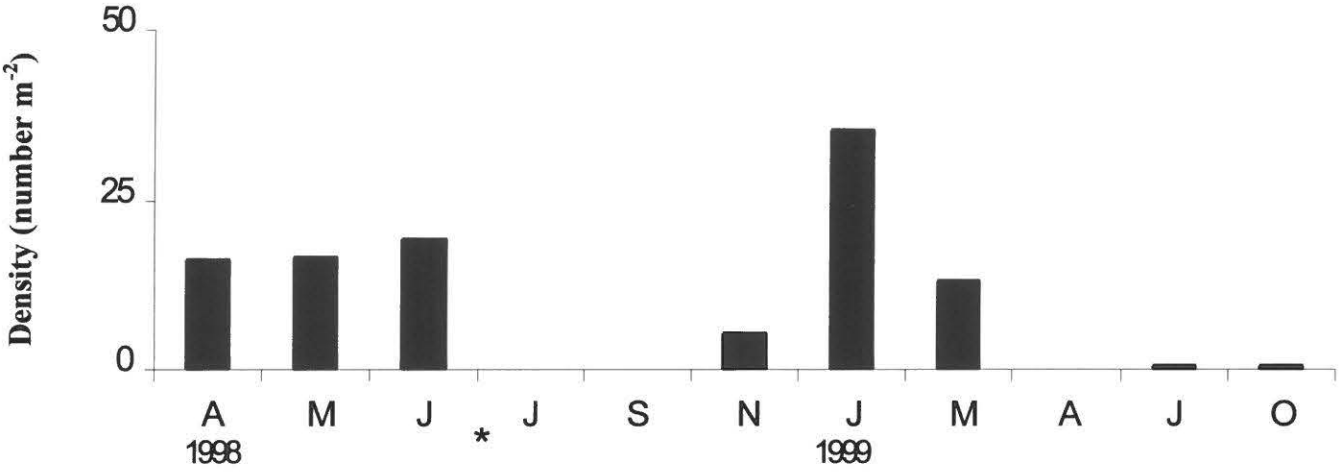


Figure 5-27. Mean density of *Tanypus* collected from demonstration wetlands in Tates Hell Swamp using sweep nets. No specimens were collected from control wetland or reference wetland. * indicates initiation of restoration..

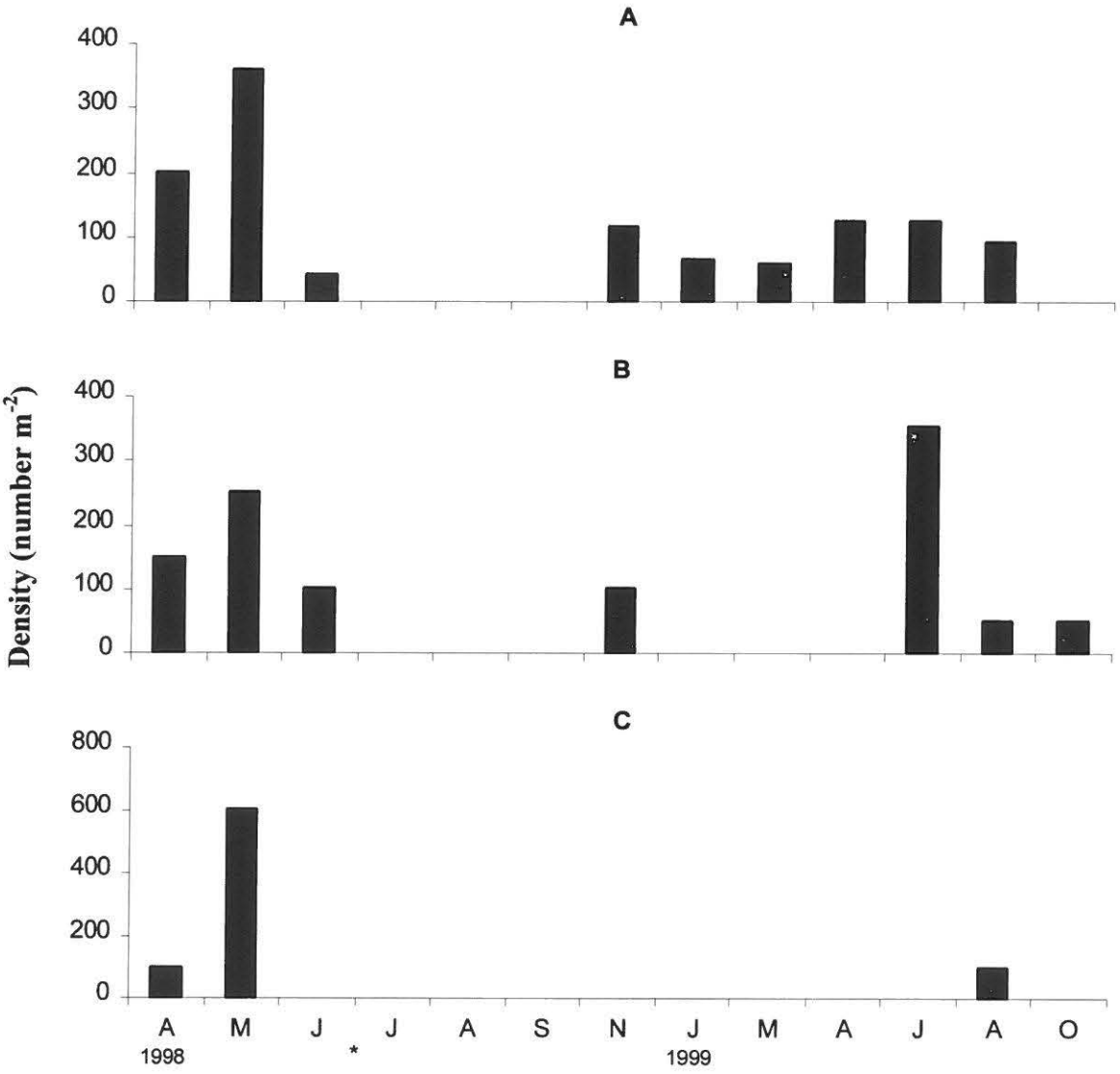


Figure 5-28. Mean density of *Procladius* collected from three habitats in Tates Hell Swamp using corers. A- demonstration wetlands and B- control wetland. No specimens were collected from reference wetland. * indicates initiation of restoration.

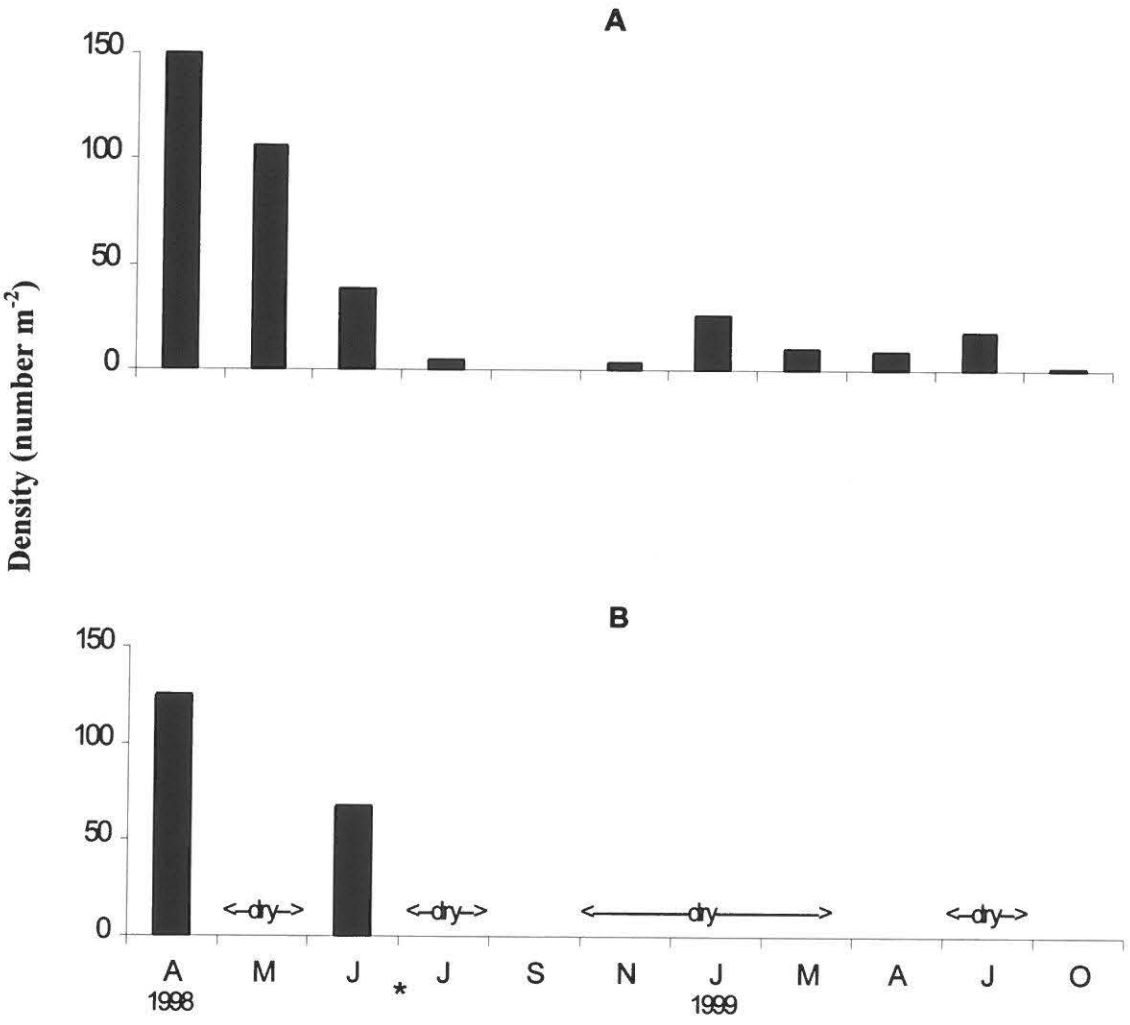


Figure 5-29. Mean density of *Procladius* collected from two habitats in Tates Hell Swamp using sweep nets. A- demonstration wetlands and B- control wetland. No specimens were collected from reference wetland. * indicates initiation of restoration.

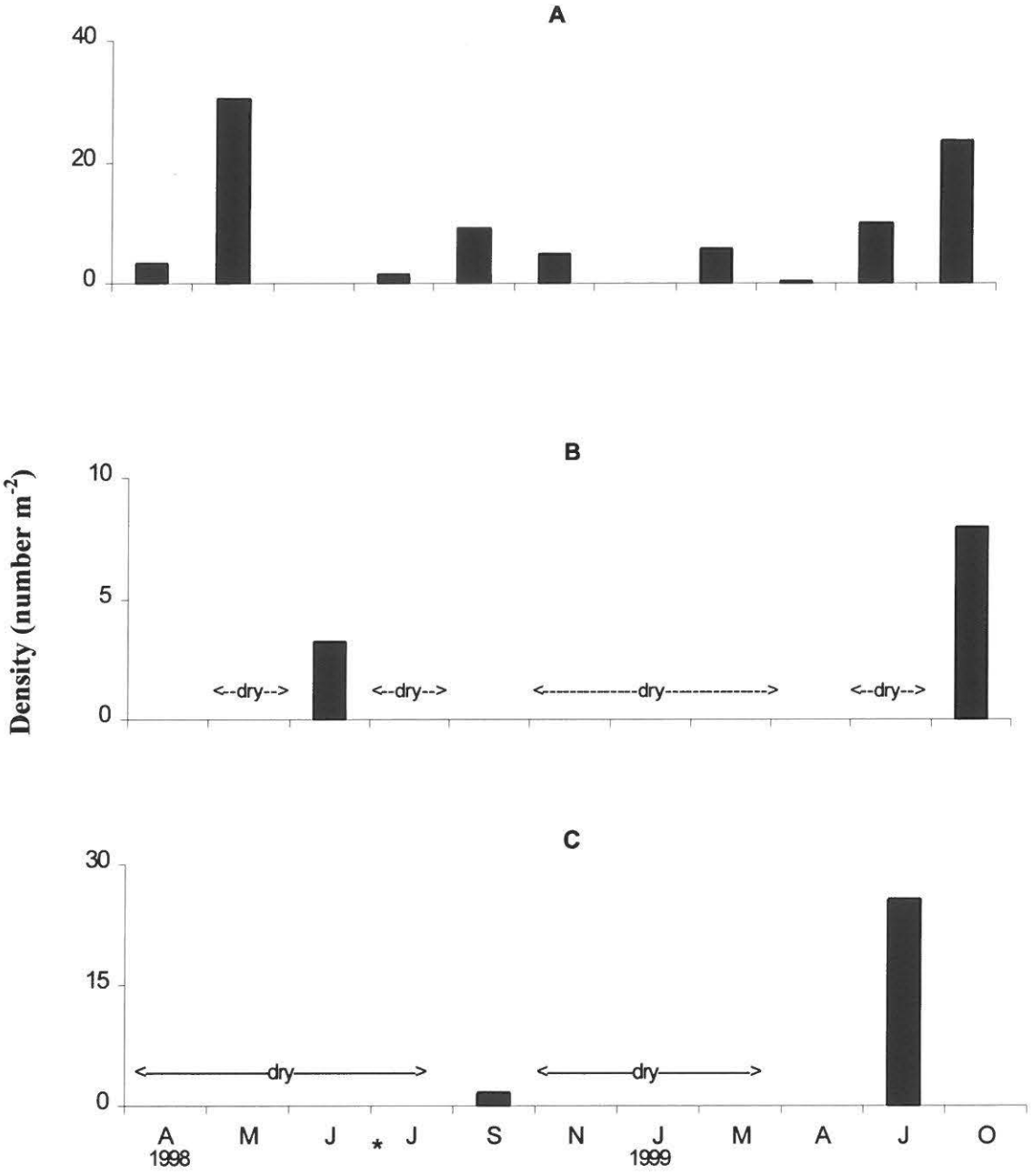


Figure 5-30. Mean density of Hemiptera collected from three habitats in Tates Hell Swamp using sweep nets. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

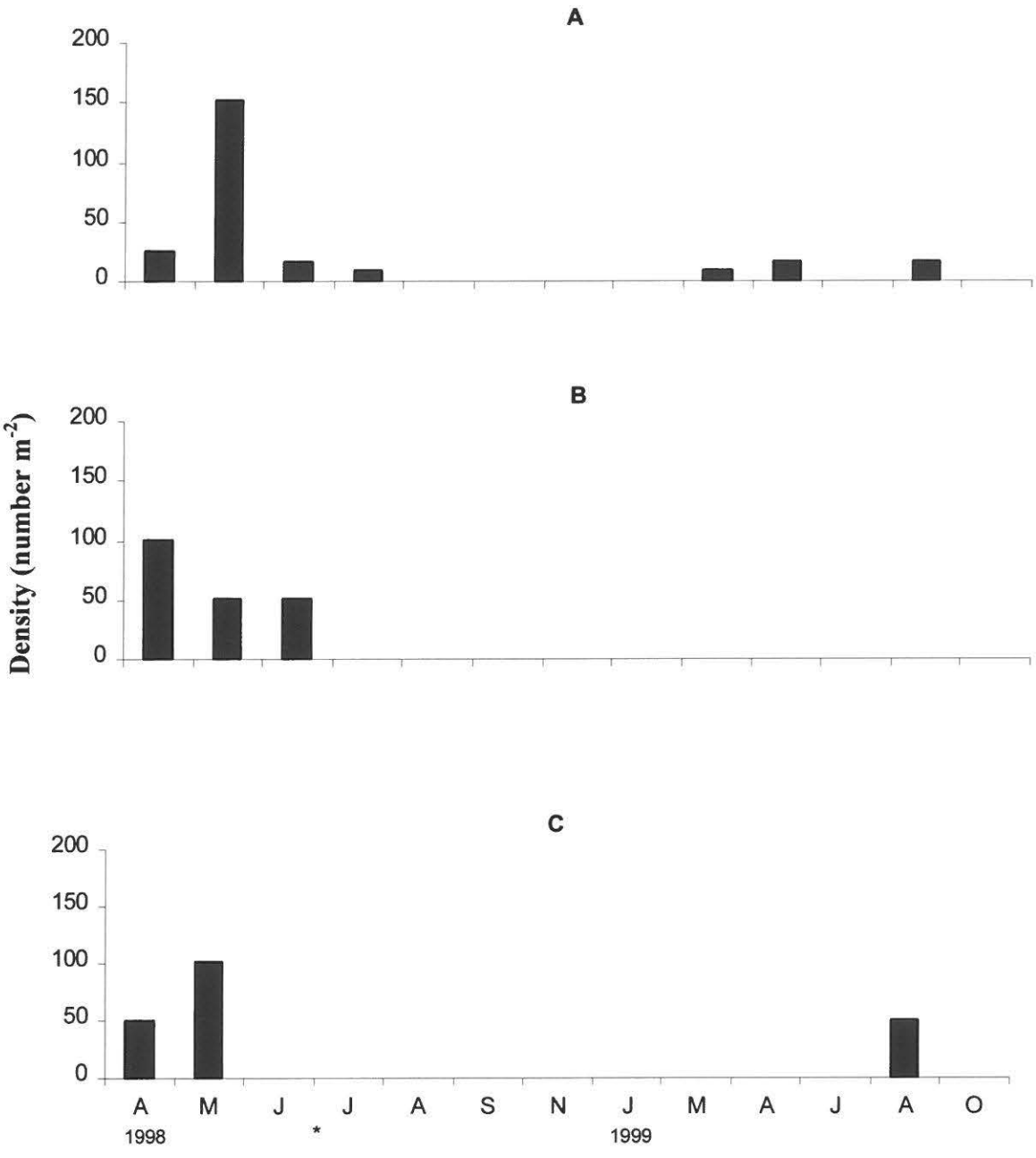


Figure 5-31. Mean density of Hemiptera collected from three habitats in Tates Hell Swamp using corers. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration.

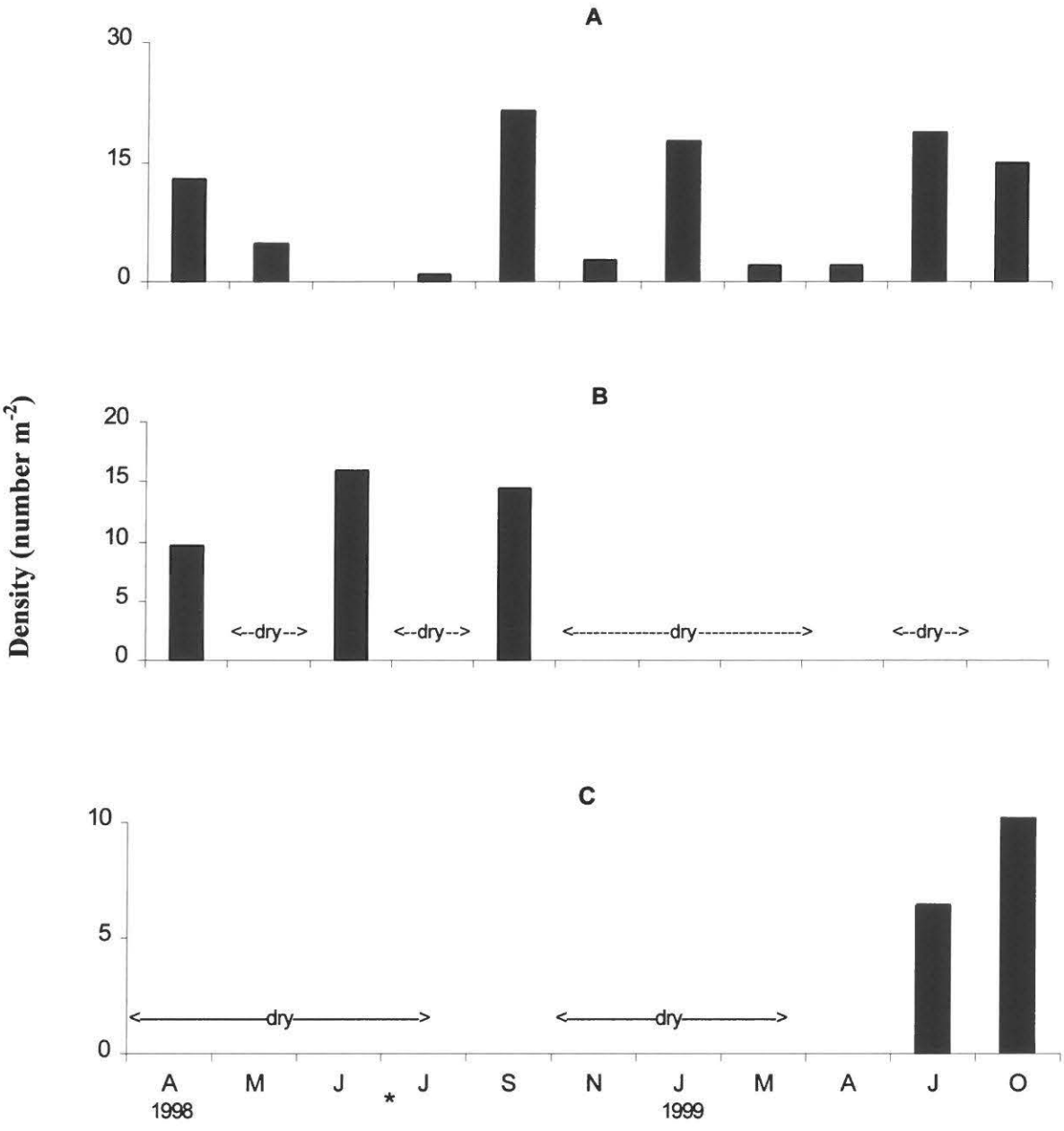


Figure 5-32. Mean density of Odonata collected from three habitats in Tates Hell Swamp using sweep nets. A- demonstration wetlands, B- control wetland and C- reference wetland. * indicates initiation of restoration

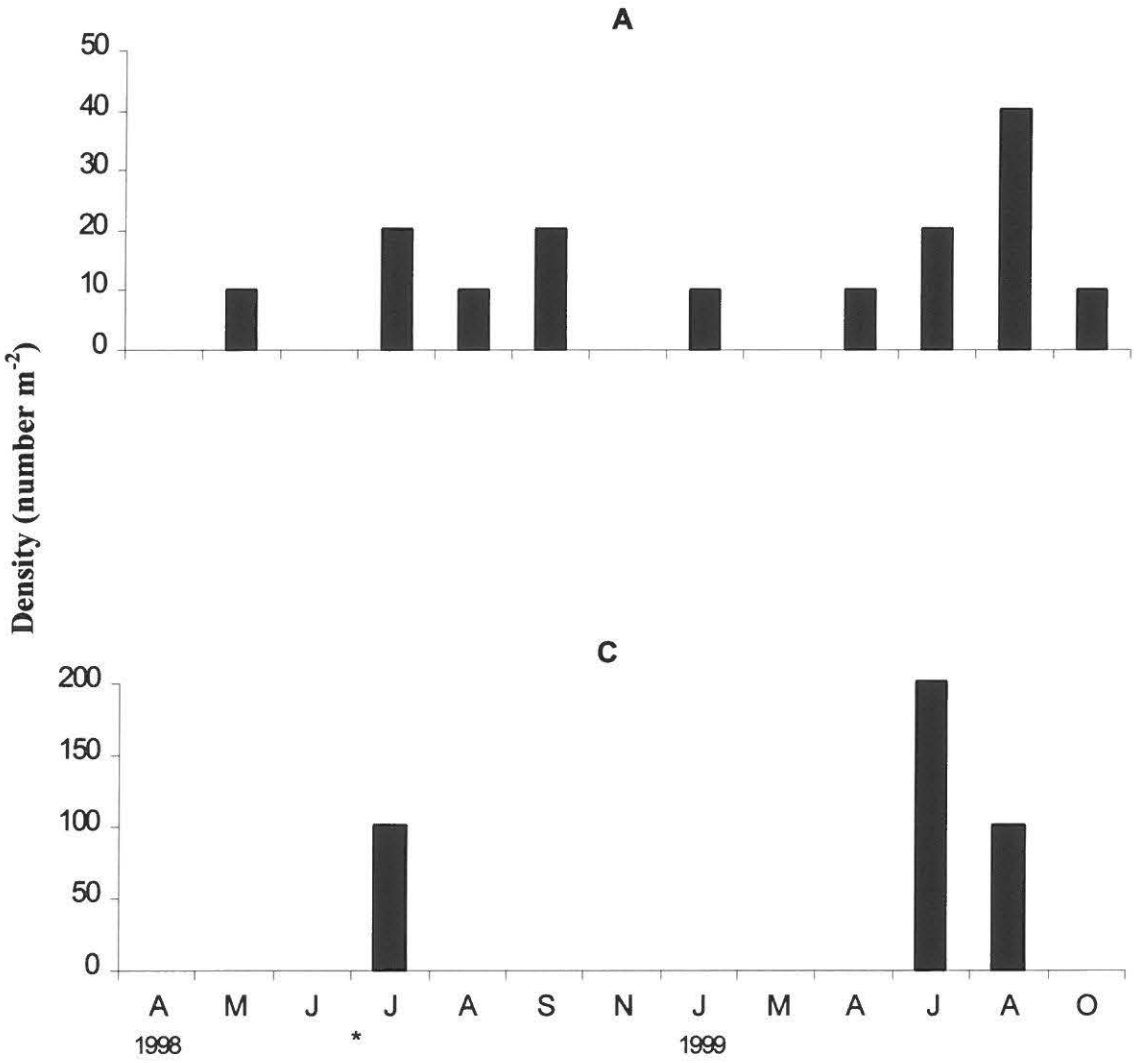


Figure 5-33. Mean density of Odonata collected from two habitats in Tates Hell Swamp using corers. A- demonstration wetlands and C- reference wetland. No specimens were collected from control wetland. * indicates initiation of restoration.

ZOOPLANKTON

Zooplankton are important links in the aquatic food chain that can reduce and modify phytoplankton assemblages (Elser and Mackay 1989; Timms and Moss 1984; Vanni 1987) and are an important food source for larval, juvenile, and adult fishes (Hanson and Riggs 1995; Pollard et al. 1983; Whiteside 1988; Whiteside et al. 1985). Zooplankton in many wetlands must have adaptations to withstand potentially dry conditions. There are sometimes refugia in which zooplankton may reside during dry periods, and, although horizontal migration is known for zooplankton (Walls et al. 1990), life cycles must be synchronized with the wet season, and reproductive propagules or resting phases facilitate dry-period survival (Kalk and Schulten-Senden 1977; McClachlan and Cantrell 1980; Rzoka 1961). This study examined zooplankton populations in both ditches and wetlands.

Description of Zooplankton Community

During the 18 months of sampling in Tates Hell Swamp, 13 taxa of cladocerans and five taxa of copepods were collected (Table 6-1). This richness is slightly lower than observed for zooplankton and meiobenthos in lakes in Florida. In an oligotrophic lake in north central Florida, Billets (1985) found 16 species of cladocerans and seven species of copepods, while in a series of mesotrophic lakes in central Florida, Blancher (1984) found 14 species of cladocerans and seven species of copepods. Anderson (1977) found 18 cladocerans and five species of copepods in Lake Drummond, a lake in the center of the Great Dismal Swamp, Virginia. In a temporary wetland in South Carolina, Taylor (1990) found at least 19 cladocera and 10 copepods, which has a higher species richness than any of the other southeastern systems.

One group of Cladocera, the chydorids, are typically very abundant and possess high species richness in most littoral habitats. Frey (1982) found an average of about 14 species per wetland system in the Florida panhandle, compared to the four species in Tates Hell Swamp reported here. Each of these systems had low pH and conductivity similar to Tates Hell Swamp, but they were not heavily colored, unlike Tates Hell Swamp.

Wetland/Ditch Interactions

Zooplankton and meiobenthos densities did not differ significantly between demonstration ditches and wetlands ($t = 1.35$; $P = 0.20$). After restoration, wetlands were more frequently flooded as a result of the creation of low water crossings and ditch backfilling, but this did not appear to affect zooplankton communities. However, the low number of samples in the pre-restoration period did not allow for adequate interpretation of the data. A longer pre-restoration time period for the collection of samples would have allowed detection of any restoration-related changes.

Neither Cladocera nor Copepoda were collected in significantly greater densities from the wetlands than they were from ditches (Cladocera, $t = -0.957$; $P = 0.361$; Copepoda, $t = -0.155$; $P = 0.88$). The two highest peaks of zooplankton were during December 1998 (19.4 mg L^{-1}) and September 1999 (24.5 mg L^{-1}) in the demonstration ditches of Tates Hell Swamp. These peaks corresponded with periods when water was absent in the wetlands. Cladocera were collected in significantly higher numbers in demonstration ditches ($t = 3.82$; $P = 0.004$) and demonstration wetlands ($t = 5.19$; 0.001).

The construction of the permanently-flooded low water crossings at Sites 2 and 3 could have provided refugia from drought different from those of the ditches. The shallow, open low water crossings can provide food resources in the form of epiphytic and epipelic algae which zooplankton such as chydorids, macrothricids, and some copepods consume (Dodson and Frey 1991; Soto and Hurlbert 1991; Whiteside *et al.* 1978). On a visual basis, ditches did not appear to provide nearly the same amount of epipelic algae as the low water crossings.

However, because zooplankton are, by definition, planktonic, the degree to which they could migrate horizontally to low water crossings or ditches as water levels fell is unknown. The term "plankton" typically infers varying degrees of control over vertical distributions, but limited control of horizontal movement (Wetzel 1983). Thus, when standing water begins to disappear, zooplankton must employ other means to survive (Taylor *et al.* 1999; Wyngaard *et al.* 1991). Cladocera produce resting eggs just prior to drought conditions, ensuring repopulation upon reflooding (Dodson and Frey 1991). Cyclopoid copepods can enter dormancy in copepodid instar IV in response to adverse conditions, and development will proceed once reflooding occurs (Williamson 1991). Calanoid copepods, like cladocerans, produce resting eggs, which hatch upon reflooding (Williamson 1991). Copepods were also found in moist soils during this study in benthos cores, suggesting that some adults can survive when no standing water is present.

Once reflooding occurs, it takes several days before resting eggs hatch (Wyngaard *et al.* 1991). Adults have short generation times, especially during warmer months, thus their densities can increase exponentially in just a few weeks. Under warmer conditions (20-25° C), generation times can be as short as one to two weeks (Anderson and Benke 1994; Anderson *et al.* 1998; Hann 1985; Taylor *et al.* 1999).

Rotifers, cladocerans, and copepods are the dominant zooplankton groups found in most freshwater systems. Some oligotrophic systems have rotifers as the dominant zooplankton (Stoneburner and Smock 1980). Others have seen Cladocera as the dominants (Hessen 1989; Schoenberg 1988). Copepods can be the dominant zooplankton in oligotrophic systems (Blancher 1984; Byron *et al.* 1984). Elmore (1984) found copepods to be the dominant members of the zooplankton in an oligotrophic Florida lake, but rotifers were also important. However, all of these studies only collected zooplankton in open-water areas, which may exclude certain taxa (e.g., chydorids) from being collected. Furthermore, zooplankton undergo pronounced diel vertical migration, with most zooplankton moving to deeper waters during the day, and moving towards the

surface at dusk (Wetzel 1983). The sampling methods in many of these studies are unable to account for this migration. Entire taxa can be missed if the sampling technique does not adequately sample strata where zooplankton may be found, especially if zooplankton are collected during the day.

In Tates Hell Swamp, this could have been the case in both the ditches and wetlands. The depth of the ditches may have been enough for zooplankton to migrate to the more hypoxic bottom layers; and wetlands, while not deep, may have had most of their zooplankton community in or on the sediments. At night, these zooplankton would move into the water column (Stenson and Oscarson 1985). The sampling technique we used only sampled the top 10-13 cm of water and was performed during the day. In the wetlands, our technique may have missed those taxa closely associated with the sediments, such as many chydorid and macrothricid cladocerans. However, other sampling techniques would be difficult to implement in the wetlands.

Oligotrophic waters such as those of Tates Hell Swamp are characterized by low standing crops of phytoplankton (Wetzel 1983). The highly-colored waters of oligotrophic Tates Hell Swamp may also decrease the phytoplankton standing crop by decreasing light transmission (Bienert 1982). Since many cladocerans, copepods, and some rotifers feed on phytoplankton, this resource can be limiting for zooplankton (Dodson and Frey 1991). Unless a zooplankter can utilize other resources such as bacterioplankton, organic matter (particulate or dissolved), other zooplankton, or periphyton, its presence in oligotrophic waters is likely to be in low numbers, if not absent altogether (Hessen 1989; Stoneburner and Smock 1980).

Size-selective predation by fish has a significant impact on zooplankton communities (Brooks and Dodson 1965; Eggers 1982; Hall *et al.* 1976). Larger zooplankton tend to dominate in systems with low or no zooplanktivores because they are more efficient at filtering phytoplankton from the water column (Vanni 1987). Fish tend to choose larger zooplankton in an effort to maximize the energy return in relation to the energy used feeding (Pyke *et al.* 1977). The small size of Florida zooplankton as compared to those in more northern latitudes has been attributed to such size-selective predation (Bays and Crisman 1983; Blancher 1984; Nordlie 1976; Wyngaard *et al.* 1982). Hence, in systems with an abundance of zooplanktivorous fishes, mean zooplankton sizes should be smaller than those systems without the zooplanktivorous fishes. In Tates Hell Swamp, the two most abundant zooplankters, *Alona guttata* and *Microcyclops varicans*, were the smallest cladoceran and copepod, respectively, suggesting the effects of size-selective predation.

Small zooplankton may also be dominant in Tates Hell Swamp for reasons other than predation. Small zooplankton predominate in waters with low food resources (Foran 1983) and phytoplankton density is very low in Tates Hell Swamp. Food resources would be expected to be especially low in the summer when elevated temperatures have an adverse effect on phytoplankton (Foran 1983). Small zooplankton have lower filtering rates, which makes them more efficient at obtaining food in phytoplankton limited systems.

Seasonality

In Tates Hell Swamp, the general seasonal pattern of abundance tends to follow that of Schoenberg (1988), whose Okefenokee Swamp study site is similar in latitude to that of Tates Hell Swamp. Cladocera in the demonstration ditches were generally more abundant in the late autumn-early spring (November-March) and were lowest during the spring and into summer (April-October), with the exception of May of both years and September 1999 (Figure 6-1). The large September 1999 abundance may reflect the abnormally low water levels at that time which dried the wetlands, leaving water only in the ditches and low water crossings. Zooplankton may have been concentrated in the ditches at this time, leading to the large abundance. The reason for the May peaks is less obvious. While water levels were low in May 1998, perhaps suggesting a concentration effect, water levels were high in May 1999. Another possible mechanism for high zooplankton densities during low water periods is that particulate and dissolved organic carbon may move out of the wetlands and into the ditches. The movement of water into the ditches during this time may also concentrate carbon sources, which can be utilized as a zooplankton food source (Arruda *et al.* 1983; McCabe and O'Brien 1983).

Copepods displayed less pronounced seasonality, though in general, the lowest abundance was during the summer and the peak abundance in the winter. Similar peaks as with the Cladocera also appeared in May of both years and September 1999. These trends are most evident in the demonstration ditches (Figure 6-1) and demonstration wetland (Figure 6-2), and to a lesser extent in the control ditch (Figure 6-3) and control wetland (Figure 6-4).

In temperate lakes and wetlands, zooplankton peak in the late spring-early summer when phytoplankton are most abundant (e.g. Hann and Zrum 1997; Murkin *et al.* 1992; Whiteside 1988). Fish predation, specifically by juvenile fishes, in concert with diminishing food resources, eventually may act to decrease zooplankton abundance through the summer (Bohanan and Johnson 1983; Whiteside 1988), although invertebrate predation, specifically the dipteran *Chaoborus*, may also play a role in some systems (e.g. Goulden 1971; Hanazato and Yasuno 1989; Havens 1991b; Neill 1981). There is often a smaller zooplankton peak in the autumn which corresponds to an increase in phytoplankton, possibly as a result of nutrient regeneration from macrophyte senesence (Wetzel 1983).

In warm temperate and subtropical areas, the seasonal dynamics are a bit different. Beaver (1990) observed in oligotrophic Florida lakes that zooplankton reach peak abundance in the late winter-early spring with dominance in autumn as well. However, in a lake in the Okefenokee Swamp of south Georgia/north Florida, Schoenberg (1988) found peak abundance of copepods in late autumn and cladocerans in late summer-early autumn. The peak abundance of both groups in Tates Hell Swamp in the late winter through early spring also agrees with those results found by Bays (1983). In the case of these southern systems, temperature may be the major determinant of seasonal zooplankton abundance. However, increased summer vertebrate predation and an increase in inedible algae may also play a role (Bays 1983).

Species richness showed little trend seasonally in the demonstration ditches (Figure 6-5), and demonstration wetlands (Figure 6-6), control ditches (Figure 6-7), and control wetlands (Figure 6-8).

The control ditch showed the same general trend of peak winter density and low summer density (Figure 6-3). However, seasonal abundance in the control ditch was poorly correlated with the demonstration ditches for both cladocerans and copepods. Since there was only one control site, it is possible that sampling did not accurately reflect the true zooplankton community. Alternatively, there may be differences between the demonstration and control sites which may affect zooplankton assemblages. Zooplankton are known to form swarms (Butorina 1986; Tessier 1983), and this spatial clumping may bias results. If a swarm happens to be sampled, the resulting density may not reflect the true abundance found in that body of water. Similarly, if zooplankton are swarming and sampling misses these swarms, the measured density may be artificially low. Additional control sites would have provided greater confidence in this respect but were beyond the budget of this project. May peaks were not apparent as in the demonstration ditches, but the September 1999 peak was apparent.

Since wetlands were dry at times during the study, their true seasonality is difficult to infer. The zooplankton density in demonstration wetlands (Figure 6-2) deviated somewhat from the trend seen in the ditches. Cladocera followed the general trend of peak density in the winter and lowest abundance in the summer, although density was high throughout much of the autumn and spring in both years. Copepods reached their peak density in the autumn and late spring of both years. The lowest density occurred during the winter.

Only during nine of the 16 months of sampling was standing water present in the control wetland, so delineating trends is difficult (Figure 6-4). However, for those months where standing water was present, the abundances were in line with those of the demonstration wetlands in that a peak density occurred during autumn. Standing water was absent during late spring and much of the winter, so little of these data can be used to compare the copepod trends to that of the demonstration wetlands.

The three most common factors governing copepod populations are food limitation, predation, and temperature (Williamson 1991). The dominant copepod in Tates Hell Swamp, *Microcyclops varicans*, feeds on periphyton (Soto and Hurlbert 1991), and with the abundance of periphyton in the Tates Hell Swamp cypress strands (see Chapter 8), this resource would not appear to be limiting, even in winter. Surface temperatures in the wetlands reached a low of 11 °C during January 1999. However, these temperatures were similar to those of the ditches at the same time (10 °C). Temperature is recognized as being significant in controlling reproduction, development, and growth in copepods (e.g. Cooley 1978; Herzig 1984; Jamieson 1980; Vijverberg 1980).

The spring-early summer decline of zooplankton also corresponds to an increase of bladderwort, *Utricularia* spp., which feeds on zooplankton as an additional nutrient

source in oligotrophic systems such as Tates Hell Swamp (Havens 1991b; Ulanowicz 1995). *Utricularia* is present throughout the year, but during the course of this study, it appeared to be far more abundant from spring-early summer in four of the demonstration ditches (Sites 2-3). It is possible that *Utricularia* may also be contributing to the decrease in zooplankton during this period. *Utricularia* is also found in the wetlands as well, but it was not as abundant as in the ditches.

Rotifers were found throughout much of the sampling period in low numbers, but their abundance was highest before and immediately after restoration (Figure 6-9). The majority of rotifers were collected in ditches, though they were also collected in wetlands. So few rotifers were encountered that no identification was made below the level of Rotifera. Densities were not different between ditches and wetlands.

Overall, zooplankton showed considerable seasonality, with peak abundance occurring during the colder months of the year. The small number of samples collected prior to restoration made it impossible to make meaningful comparisons of pre- and post-restoration zooplankton communities.

Table 6-1: Zooplankton and meiobenthos collected from Tates Hell Swamp, Florida, from May 1998-October 1999, their incidence in habitat type (ditch vs. wetland), and frequency of occurrence (A = abundant, C = common, U = uncommon, R = rare).

Scientific name	Habitat		Frequency of encounter
	Ditch	Wetland	
Cladocera			
Bosminidae			
Eubosmina tubicen	*x*	*x*	*U*
Chydoridae			
Acroperus harpae	*x*	*x*	*U*
Alona guttata	*x*	*x*	*A*
Alona setulosa	*x*	*x*	*C*
Eurycercus lamellatus	*x*		*U*
Daphnidae			
Scapholebris mucronata	*x*	*x*	*R*
Ceriodaphnia reticulata	*x*	*x*	*U*
Macrothricidae			
Acantholebris curvirostris	*x*	*x*	*U*
Ilyocryptus spinifer	*x*	*x*	*C*
Macrothrix laticornis	*x*	*x*	*U*
Moinidae			
Moina affinis		*x*	*R*
Polyphemidae			
Polyphemus pediculus	*x*	*x*	*U*
Sididae			
Diaphanosoma brachyurum	*x*	*x*	*A*
Copepoda			
Diaptomidae			
Diaptomus sp.	x	x	U
Cyclopidae			
Ectocyclops phaleratus	*x*	*x*	*U*
Eucyclops speratus	*x*	*x*	*U*
Microcyclops varicans	x	x	A
Orthocyclops modestus	x		R

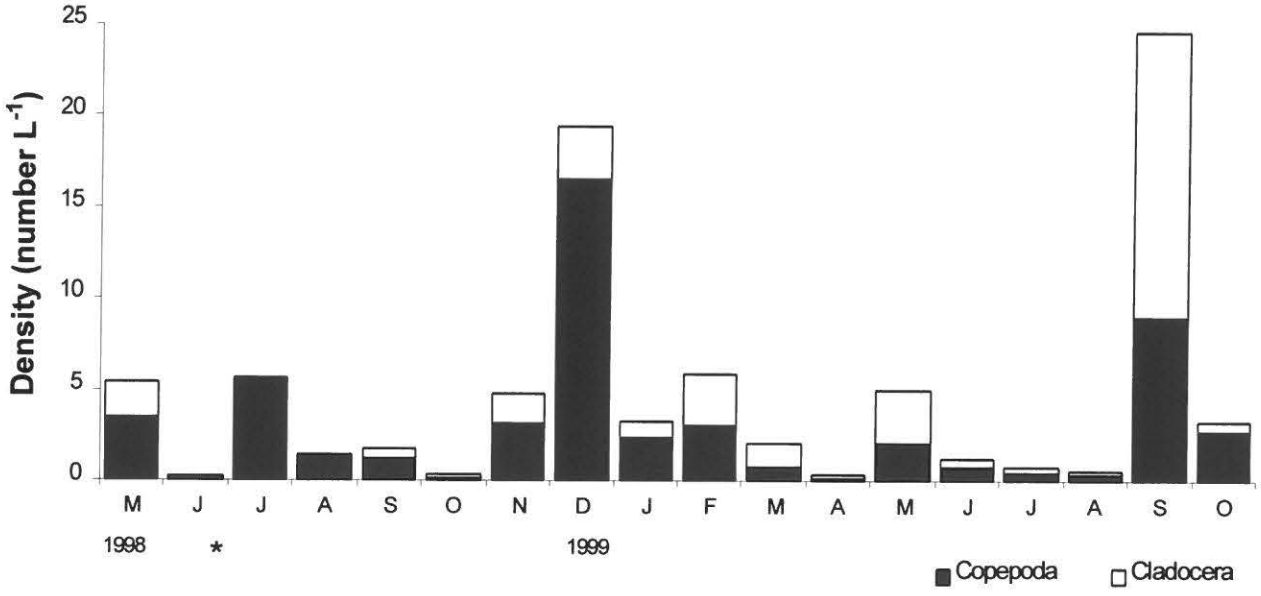


Figure 6-1. Mean density of Cladocera and Copepoda collected from demonstration ditches in Tates Hell Swamp. * indicates initiation of restoration.

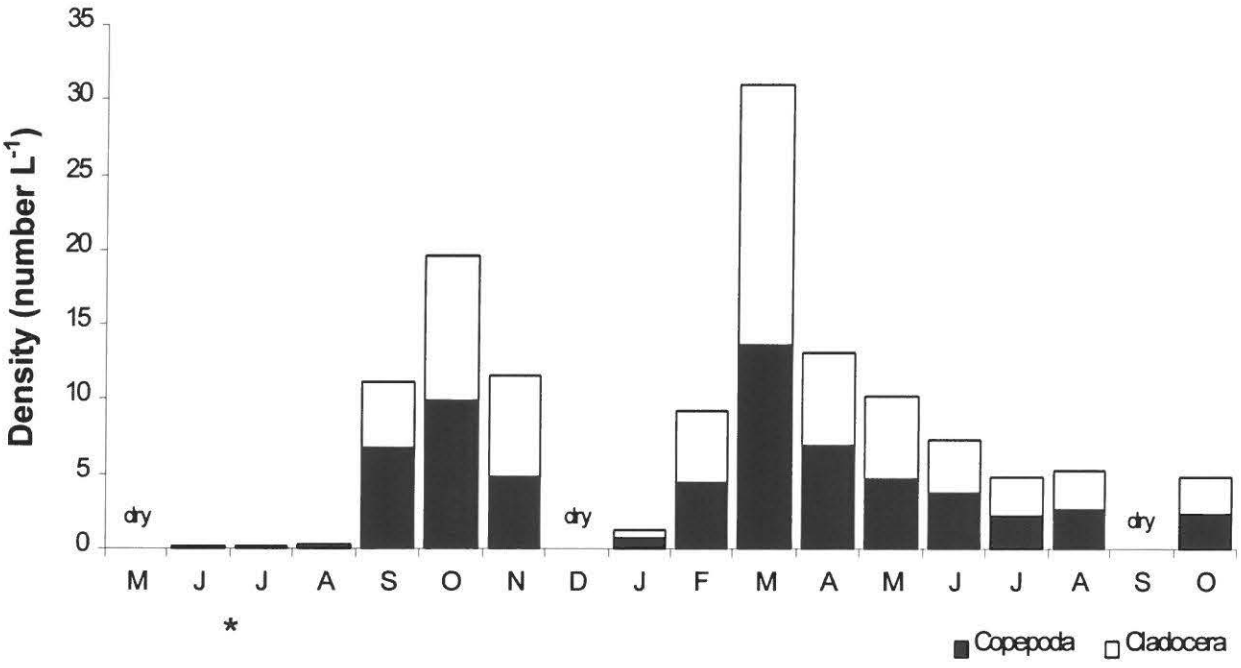


Figure 6-2. Mean density of Cladocera and Copepoda collected from the demonstration wetlands in Tates Hell Swamp. * indicates initiation of restoration.

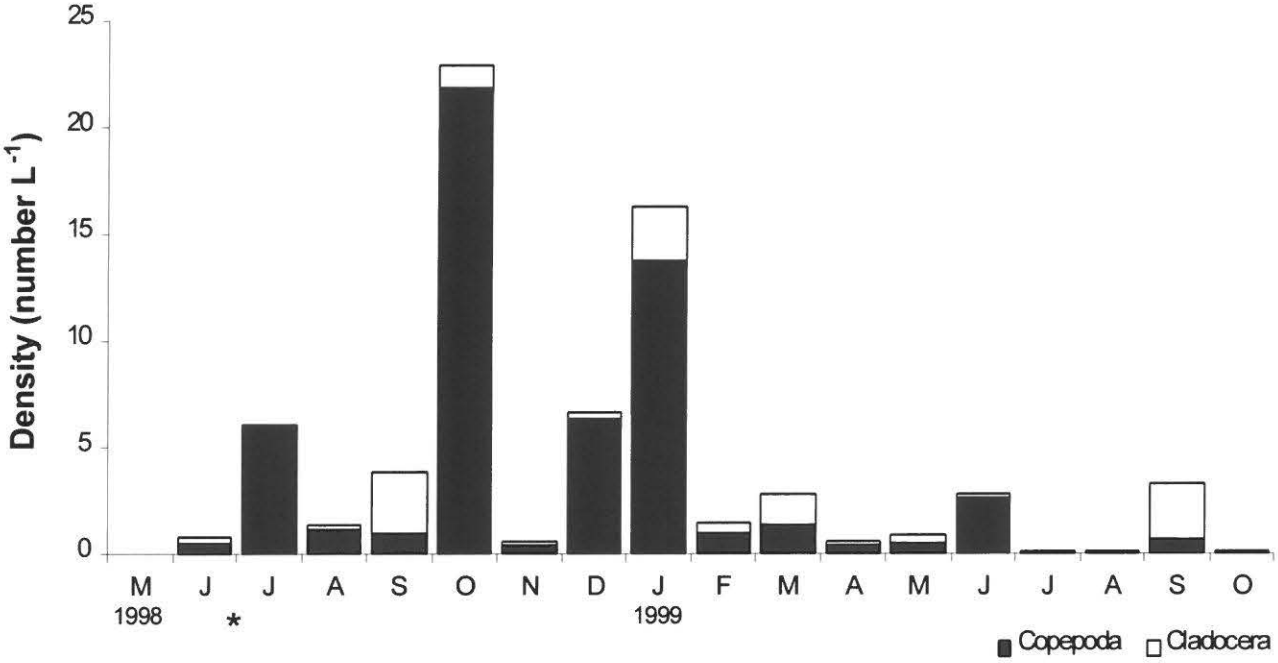


Figure 6-3. Mean density of Cladocera and Copepoda collected from the control ditch in Tates Hell Swamp. * indicates initiation of restoration.

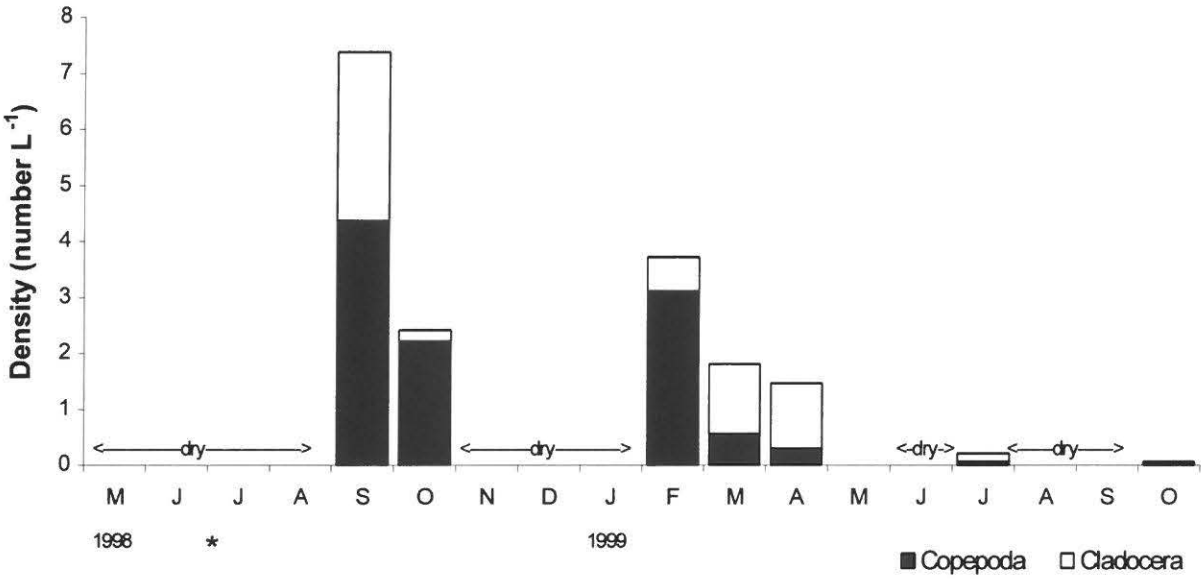


Figure 6-4. Mean density of Cladocera and Copepoda collected from control wetland in Tates Hell Swamp. * indicates initiation of restoration.

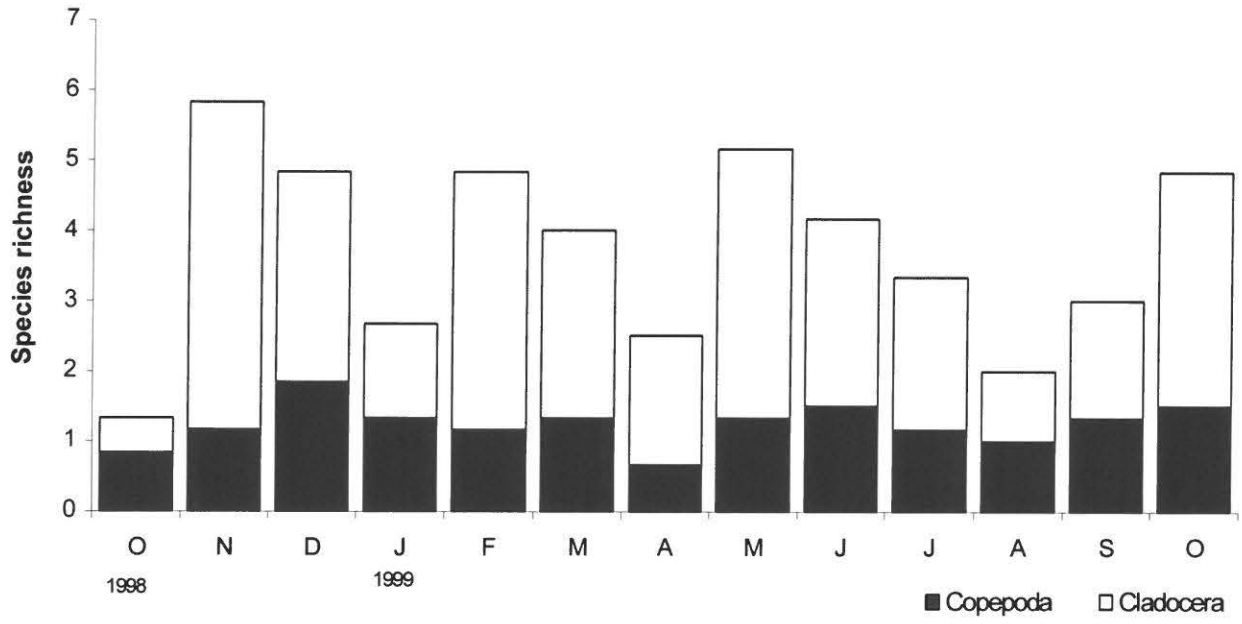


Figure 6-5. Species richness of Cladocera and Copepoda collected from the demonstration ditches in Tates Hell Swamp.

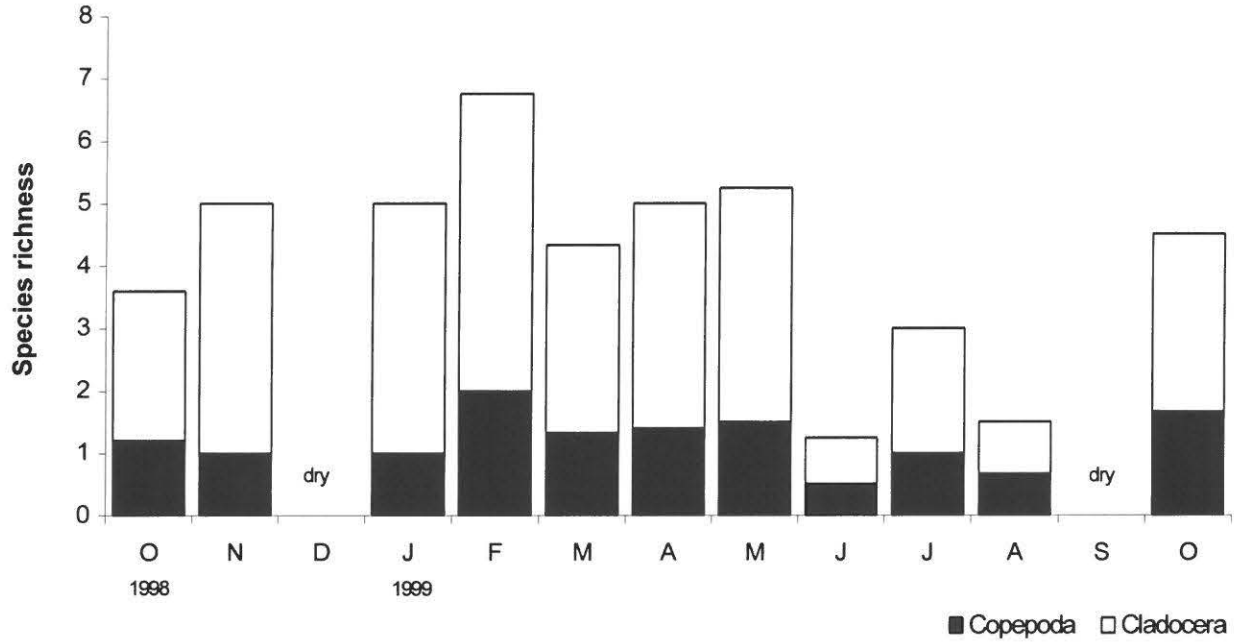


Figure 6-6. Species richness of Cladocera and Copepoda collected from the demonstration wetlands in Tates Hell Swamp.

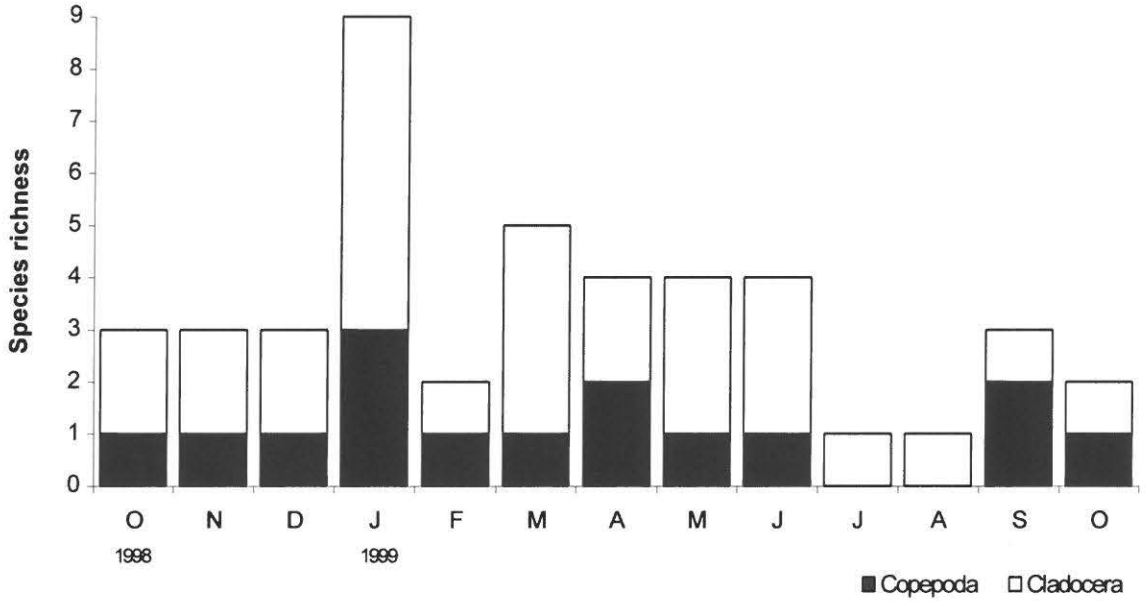


Figure 6-7. Species richness of Cladocera and Copepoda collected from the the control ditch in Tates Hell Swamp.

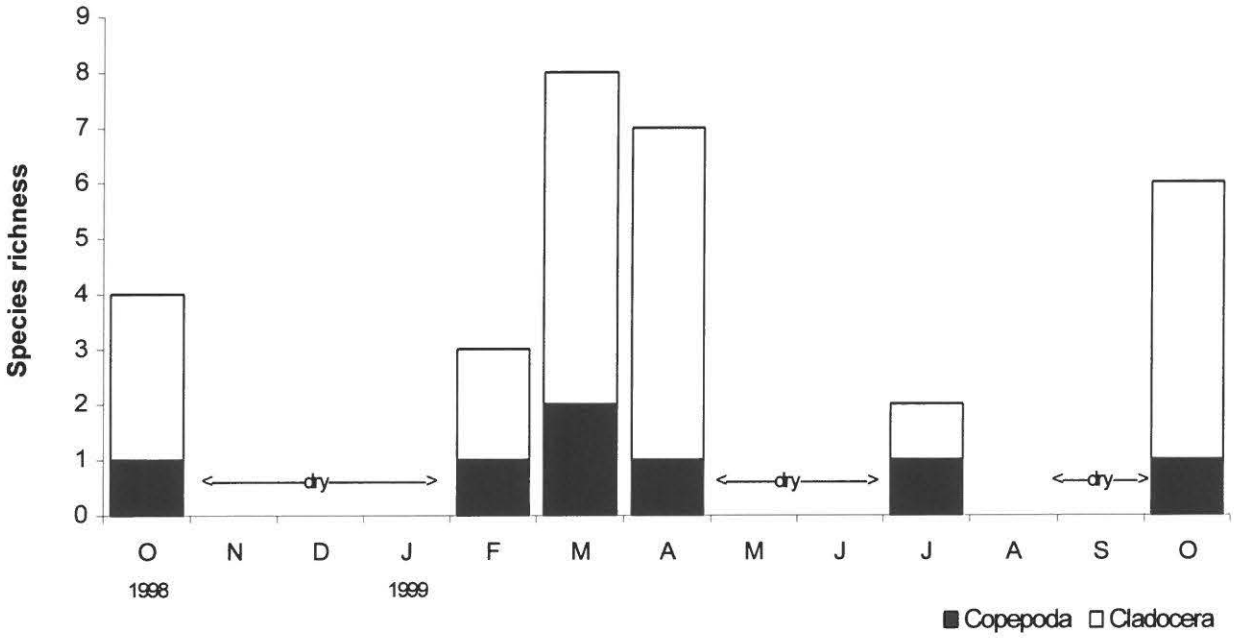


Figure 6-8. Species richness of Cladocera and Copepoda collected from the control wetland in Tates Hell Swamp.

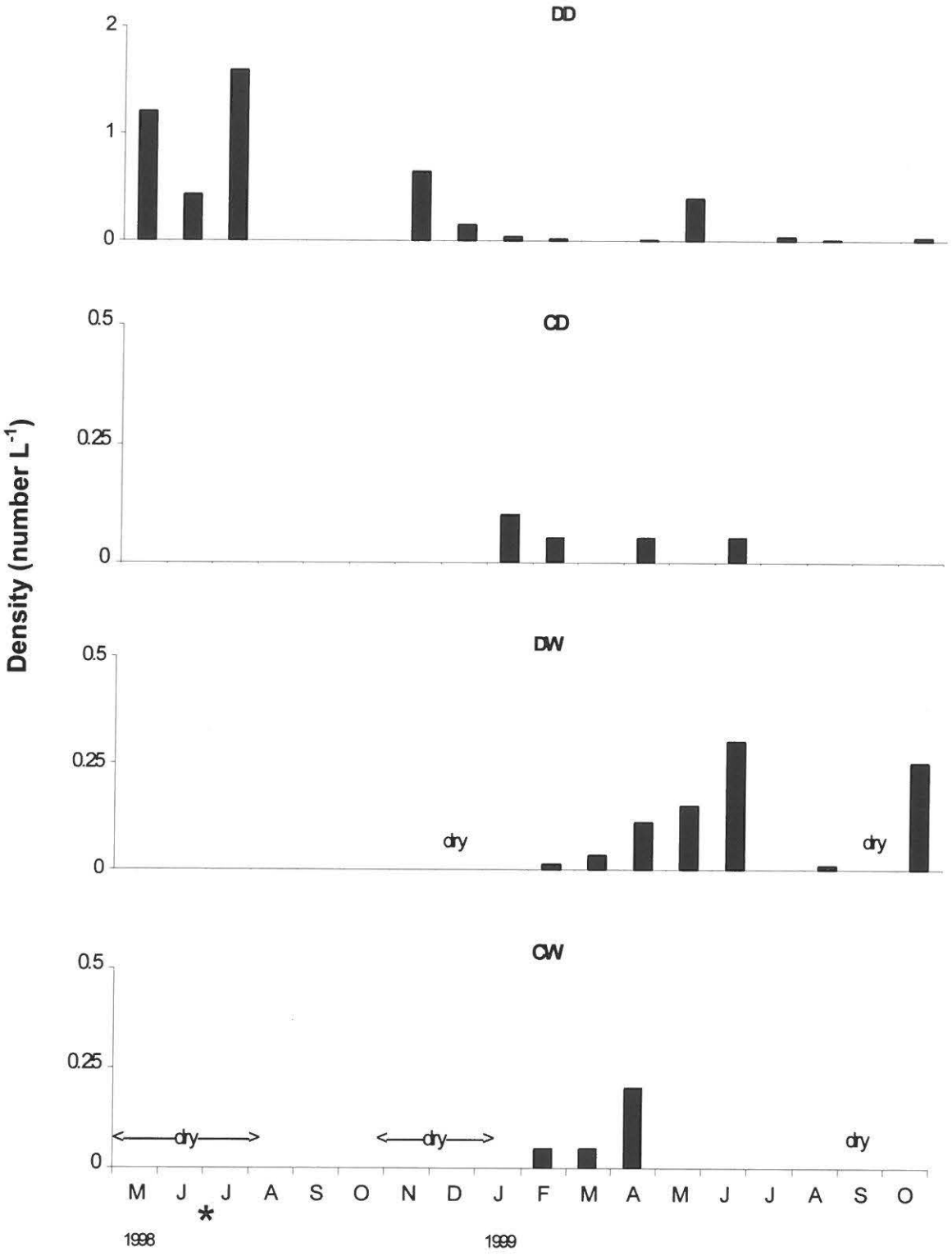


Figure 6-9. Mean abundance of rotifers collected from four habitats in Tates Hell Swamp. DD- demonstration ditches, DW- demonstration wetlands, CD- control ditch, and CW- control wetland. * indicates initiation of restoration.

FISH

Fishes have been studied fairly extensively in wetland systems such as marshes (Brazner and Beals 1997; Jordan et al. 1998; Streever and Crisman 1993) and bottomwood hardwood swamps (Killgore and Baker 1996; Knight and Bain 1996; Pezold 1998), but the fish literature for cypress swamps is limited (Browder 1976; Carlson and Duever 1977; Kahl 1964). Much of the literature on cypress wetlands focuses on cypress domes, which, because of their annual drying cycle and lack of overland flow (Dierberg and Brezonik 1984), rarely possess fish communities.

Wetlands can be important for feeding and reproduction of fish. During flood stage, fishes of such neotropical rivers as the Orinoco (Rodriguez and Lewis 1997), Parana (Agostinho and Zalewski 1995) and Amazon (Fernandes 1997; Tejerina-Garro et al. 1998) spread out from the main channel into floodplain wetlands. There they find greater food resources, especially fruits, and cover- and most taxa are unable to complete their life cycles without the flooding (Welcomme 1979). In more temperate regions, floodplain wetlands are also important for many species of fishes. As with the neotropics, temperate wetlands are important for feeding and reproduction and as a nursery (Pollard et al. 1983) (Killgore and Baker 1996; Turner et al. 1994) (Tibbs and Galat 1998). Wetlands may also act as refugia from predation when the predator is not as well-adapted as prey species to low dissolved oxygen levels in the wetland (Chapman et al. 1996).

Description of the Fish Community

During the 18 months of fish sampling, 13 species belonging to nine families of fishes were collected in Tates Hell Swamp. Of these, 6 species were collected in gill net sampling, and ten species were collected in fish traps (Table 7-1). Redfin pickerel (*Esox americanus*), yellow bullhead (*Ameiurus natalis*), and warmouth (*Lepomis gulosus*), were the only species captured using both methods. Ditches (11 species) possessed a greater species richness than both low water crossings (eight species) and wetlands (eight species) (Table 7-1). It should be noted, however, that gill nets were used only in ditches, and thus there exists some bias in these results. Taking only into account the fish trap data, only seven species were collected from ditches (Table 7-1).

The fish species richness in Tates Hell Swamp is consistent with other wetlands in the southeastern United States (Figure 7-1). Generally, Tates Hell Swamp species richness was less than that of both riparian wetlands and marshes (8-37), but greater than other cypress wetlands (8-9). Of the other studies listed in Figure 7-1, the assemblage of fishes in Tates Hell Swamp is most similar to that of a study performed in South Florida in a wetland with a diversity of habitats, including seasonally flooded wet prairie and deeper slough areas that are analogous to the cypress strands and ditches of Tates Hell Swamp (Jordan *et al.* 1998). The Tates Hell Swamp assemblage is also similar to that of Knight

(1996), with eight of the 14 fishes collected in Tates Hell Swamp (57%) also found in the floodplain of the Pea River in southeastern Alabama. Both these studies had areas which served as habitat for fishes when the wetlands were not flooded, similar to the role of ditches in Tates Hell Swamp. Fish abundance does not appear to be correlated with conductivity, pH, dissolved oxygen, or temperature (Figure 7-2).

Fish Trap Data

The three most frequently collected species from Tates Hell Swamp were mosquitofish (*Gambusia holbrooki*), pygmy killifish (*Leptolucania ommata*), and warmouth, (*Lepomis gulosus*) (Figure 7-3). *Leptolucania* dominated the fish trap collections prior to restoration/drought, while afterwards, *Gambusia* was the most abundant fish. Abundance of *L. gulosus* increased during spring 1999, and it was the second most abundant fish after *Gambusia*. These *L. gulosus* collected in fish traps represent one year old and young-of-the-year (YOY) fishes. All other species collected in fish traps, exclusive of redfin pickerel (*Esox americanus*) and yellow bullhead (*Ameiurus natalis*), represented a range in sizes from juvenile to adult.

Fliers (*Centrarchus macropterus*), and spotted sunfish (*Lepomis punctatus*), were represented by single specimens collected in gill nets during September 1999. Swamp darters (*Etheostoma fusiforme*) (Figure 7-4), lined topminnows (*Fundulus lineolatus*) (Figure 7-5), and yellow bullheads (*Ameiurus natalis*) (Figure 7-6), were the least abundant species in fish traps, represented by two, two, and one individuals, respectively. *Etheostoma* and *Ameiurus* are bottom-dwelling fishes, and the low catch in fish traps may be a reflection of equipment bias, as fish traps are not designed to collect bottom-dwellers. *Fundulus lineolatus* was seen commonly in the control ditch but was never collected there.

Pirate perch, (*Aphredoderus sayanus*), pygmy sunfish (*Elassoma evergladei*), banded topminnows, (*F. cingulatus*), and redfin pickerel (*Esox americanus*) were collected throughout the study, but were never a numerically abundant part of the catch in fish traps. *Esox* adults were also commonly collected in gill nets. *Aphredoderus* was relatively numerous ($n \geq 15$) in April 1998 and in September 1999 (Figure 7-7). The reason for its abundance is unknown, though the September 1999 collections did occur while the fish traps were completely submerged due to heavy rains from a tropical depression during the night. Whether this favored the collection of *Aphredoderus* is unknown. *Elassoma* was collected more frequently from ditches than wetlands and low water crossings (Figure 7-8), but was never an important part of the catch. *Fundulus cingulatus* was collected in low numbers except for a substantial number ($n = 20$) collected in the control wetland in August 1999. Half the *F. cingulatus* collected during the course of the study were from a single sampling day at a single site, the control wetland (Figure 7-9). It is unclear as to why such a large number were collected at that site at that time. *Esox* was collected throughout the study in low numbers (Figure 7-10).

Warmouth (*Lepomis gulosus*) was the only species that was collected in greater numbers in wetlands than in either ditches or low water crossings (Figure 7-11), although this difference was not statistically significant. Wetlands provide greater structural complexity in the form of woody debris, emergent and submersed macrophytes, and trees than do either ditches or low water crossings. Ditches contained woody debris, and low water crossings contained little woody debris, but do have some submersed macrophytes. This structure can provide a refugium from predation, and with the wetlands being the most structurally complex, they should provide the best refugium (Orth and White 1993). Associated with these woody debris, leaf litter, trees, and macrophytes are macroinvertebrates (Orth and White 1993). These are important parts of the diet of every fish in Tates Hell Swamp with the exception of *Esox*, which is the swamp's only true piscivore. As such, *L. gulosus* may be favoring the wetlands because of both the protection afforded by structure and the food resources available. The preference for shallow, well-vegetated areas by *L. gulosus* has been seen by other researchers as well (Guillory 1978).

Gill Net Data

Gill net data were rather limited, as almost 60% of those fishes collected had empty stomachs. Only three species had food items in their stomachs, *L. gulosus*, *Esox*, and *Ameiurus*. Diet analysis showed that *L. gulosus* and *Ameiurus* were omnivorous, while *Esox* was piscivorous (Figure 7-12). Odonate larvae were most common component in *L. gulosus* stomachs (27% occurrence), followed by unidentified constituents (26% occurrence) and adult coleopterans (13% occurrence). *Ameiurus* fed predominantly on fish (32% occurrence), followed by odonates (17% occurrence), coleopterans (17% occurrence), and dipterans (11% occurrence). *Esox* consumed only fishes, but detritus also occurred frequently in the stomachs (33% occurrence). Detritus was probably accidentally ingested when the fishes engulfed prey. These findings agree with diets of these fishes elsewhere within their ranges, although *L. gulosus* is also known to eat crayfish and small fishes (Guillory 1978; Hoyer and Canfield 1994).

Gill nets and fish traps are both size selective. This is why both were utilized, but also why there is such a discrepancy in the taxa collected between the two methods. Gill nets can only collect those fish small enough to swim into the mesh and large enough to get their gills caught in the mesh. As a result, fish such as *Gambusia*, *Leptolucania*, *Fundulus spp.*, *Aphredoderus*, and *Elassoma* could not be collected in gill nets. Similarly, young-of-the-year centrarchids and young of other species with larger adults could not be collected. The three mesh sizes attempted to maximize the chances of collecting fishes of a range of sizes.

In contrast to gill nets, fish traps are designed to collect smaller species. Any organism larger in diameter than 2.2 cm cannot enter the fish trap. This excludes large adults, though juveniles potentially could enter the traps. Fish traps do not appear to collect all taxa capable of entering the trap uniformly. In the control ditch, a large number of lined topminnows (*F. lineolatus*) could be seen throughout the study at the water surface, yet

not a single specimen was collected from this locality. Similarly, *F. cingulatus* was frequently seen in many of the ditches, but only 40 individuals were collected during the course of the study, half of those coming from one sample date. Because these two species, and many cyprinodontids in general, spend most of their time at the water surface (Page and Burr 1991), fish traps that have their funnels a bit below the water surface may not favor their collection.

Gill nets were utilized only on four sampling dates in 1998 and not at all in 1999. Extensive damage was done to several of the nets by wildlife, most likely river otters (*Lutra canadensis*), or alligators (*Alligator mississippiensisis*). In June 1998, much of the gill net placed into Ditch 2 was shredded and the catch of fish was destroyed. Shredding also occurred with gill nets in other ditches during July 1998, September 1998, and November 1998 as well. Otters were occasionally seen in Tates Hell Swamp during the course of the study and following restoration in July 1998. A dead otter was found in the ditch at Site 2. Otters are known to consume fishes (Burt and Grossenheider 1980), and dead fish suspended in the water column in a gill net should be rather inviting for a number of predators and scavengers, including otters. Alligators are also known to consume fishes (Conant and Collins 1991), and numerous alligators were observed in Tates Hell Swamp throughout the course of the study.

In addition to the damage to the nets, catches decreased to almost zero during the drought of summer 1998. Also, as a result of the restorations, depths of the demonstration ditches decreased and more brush fell into the ditches. These factors made setting gill nets extremely difficult. As a result, gill net sampling was stopped in November 1998.

Wetland/Ditch Interactions

The ditch/wetland layout of the Tates Hell Swamp study sites can be viewed from two different perspectives. One perspective is that the wetland acts as a floodplain for the ditch, which is thus analogous to a river. The obvious difference is that the ditches in Tates Hell Swamp have almost imperceptible flow, except at Site 3 during extremely flooded conditions. However, the cypress strands bordering the ditches serve very similar functions to those of a riparian floodplain.

Floodplains are known to be of great importance for the feeding and reproduction of adult fishes and as a nursery for juvenile fishes (e.g., Copp 1989; Finger and Stewart 1987; Reimer 1991; Turner *et al.* 1994; Welcomme 1979). Floodplains are very productive ecosystems (Mitsch and Gosselink 1993), and much of this high production can be attributed to the periodic flooding of the riparian forests. This "flood pulse concept" (Junk *et al.* 1989) states that the exchange of water between the floodplain and river and nutrient cycling within the floodplain are the most important factors controlling fishes and other organisms in floodplains.

When a floodplain receives water, soils release nutrients, especially phosphorus, generally the limiting nutrient in freshwaters (Olila *et al.* 1997), and this nutrient is then available for uptake by plants and algae (Fabre 1988; Qiu and McComb 1994;

Schoenberg and Oliver 1988). With nutrients available for photosynthesis, and consequently transfer to higher trophic levels, fishes find an ample nutrition in floodplains (Holland and Huston 1985). The structural complexity of floodplains also may provide cover for fish species from predators (Paller 1987). All of this is governed by flooding, however, and if either the extent, timing, or duration of flooding is altered, spawning failures (Starrett 1951) and decreased utilization of floodplains by adults may occur (Kwak 1988).

Another way to view the Tates Hell Swamp ditch/wetland system is by considering the ditch as a refugium from decreasing water levels. Carlson (1977) observed that fishes were able to survive decreasing water levels in a South Florida cypress strand by moving into deeper pockets within the wetland. Similarly, Kushlan (1979) studied the use of "alligator ponds", (deep holes formed by digging of alligators) by fishes in the Everglades as water levels decreased and found that they utilize alligator ponds as refugia. Both of these refugia are analagous to the ditches of Tates Hell Swamp in that they provide an area for fishes to survive when water levels drop below the surface of the wetlands.

Such systems of ditches and wetlands are also present in South Florida, and Carlson (1977) suggested that these ditches are serving the same purpose as the "alligator ponds" as refugia from decreasing water levels. He also stated that the permanence of these ditches may be altering the fish assemblages by favoring the dominance of drought-intolerant predatory species. This, in turn, may affect the communities of smaller prey fishes through predation. Permanent and semipermanent waterbodies typically support a biomass comprised primarily of piscivorous fishes (Wegener *et al.* 1973). However, periodic drought or draw-down tends to support a fish biomass dominated by smaller prey species, and this appears mediated by predators such as piscivorous birds that preferentially feed on larger, piscivorous fishes (Carlson and Duever 1977; Ogden *et al.* 1976). In Tates Hell Swamp, the ditches may also be providing predator species with permanent refugia.

If the wetlands provide such advantages for fishes, why then are catches greater in ditches? As water levels rise and flood the cypress strands, smaller fishes move out into the wetlands. This could act to decrease predation pressure and to increase access to good food resources. This acts as a dilution effect, as the number of fishes in the small relative area of the ditches spread out into the larger area of the wetlands. Conversely, as water levels fall and fishes from the wetlands must come back to the permanent waters of the ditches, they are concentrated. Carlson (1977) described the same effect in a cypress strand in South Florida. There, fishes were concentrated into predator-rich depressions and alligator ponds. Similarly, Jordan (1998) found a negative correlation between water level and fish abundance in a marsh at the headwaters of the St. John's River, Florida. When water was high, fishes spread out into the wet prairie, but as the prairie dried out, fishes were concentrated into permanent sloughs. Thus, the negative correlation seen between water level and fish catch per unit effort reflects dilution and concentration effects rather than changes in fish population levels.

Turbidity

After restoration in June 1998, and over the course of the drought that occurred during the summer of 1998, water in the ditches became noticeably more turbid. Without turbidity measurements, it is unknown whether the restoration played any role in the increased turbidity, though the fact that high turbidity was observed in the demonstration and control ditches suggests that the earth-moving involved in the restoration was not the driving force. Prior to the restoration and drought, and during any other high water periods since then, the water had displayed very low turbidity or suspended solids. Furthermore, phytoplankton was never an important part of either the ditches or wetlands, as evidenced by low chlorophyll *a* concentrations, so biogenic turbidity was not a factor. Throughout the study, the waters in both ditches and wetlands had been highly colored, but only during periods of extreme low water did the ditches become turbid.

The wetlands were not turbid until September 1998, after two hurricanes added a great deal of water to the Florida panhandle, including Tates Hell Swamp. With the rapid increase in water levels as a result of these two hurricanes, the wetlands were quickly flooded. High turbidity in these wetlands may have resulted from sediment-laden water from the ditches flowing into the wetlands with the rapid rise in water levels. Typically, the decline or increase in water level occurs on the order of months, and slower increases in water level appear to decrease turbidity in the ditches and wetlands.

Turbidity can affect fishes in a number of ways. One way is to decrease the reactive distance of a fish. Reactive distance refers to the maximum distance from which a fish is able to detect prey visually (Vinyard and O'Brien 1976). High turbidity may affect reactive distance by attenuating or scattering light in the water column (Breitburg 1988; Lythgoe 1979). If the reactive distance is decreased, fishes may have more difficulty feeding (Benfield and Minello 1996), and this could lead to reduced growth rate, poor recruitment, or even death. However, effects on feeding in turbid systems are generally only seen at low light intensities, such as those at dusk and dawn, as well as during extremely overcast days (Benfield and Minello 1996; Miner and Stein 1993; Vinyard and O'Brien 1976). Under these conditions, fishes that are not well-adapted to low light conditions may locate near the water's surface, where the effects of turbidity-induced light attenuation are smallest (Miner and Stein 1993). While prey such as zooplankton are more likely to be found here (Zettler and Carter 1986), fishes are also potentially more susceptible to predation. Those species that are non-aquatic piscivores may be better able to detect prey that are near the surface, and under conditions of high light intensity, aquatic piscivores may be able to detect prey because of enhanced prey contrast against the background (Hinshaw 1985) and decreased contrast of the predator against the background (Muntz 1982).

In highly turbid conditions, gill lamellae can become irritated, and this can decrease the efficiency of oxygen uptake (Berg and Northcote 1985; Lalancette 1984; Ryan 1991) and possibly increase the incidence of disease (Bellerud *et al.* 1995). This is especially a concern for species, such as many of those found in Tates Hell Swamp, that are not adapted to living in turbid conditions. Furthermore, decreased efficiency of oxygen

uptake can be a problem in areas such as Tates Hell Swamp which periodically experience hypoxic conditions.

An indirect manner by which fishes could be affected by turbidity is through a decrease of cover in the form of aquatic vegetation. Turbidity shades aquatic plants, and this decreases the photosynthetic ability of the plants (Fletcher *et al.* 1985; Giesen *et al.* 1990). The plants can die off, eliminating cover for those species often associated with vegetation. In Tates Hell Swamp, aquatic vegetation was very limited throughout the entire study. Floating-leafed plants such as *Nymphaea odorata* and *Nuphar lutea* were commonly found in many of the ditches, but they would not be as likely to be affected by turbidity as submersed plants. However, bladderworts, *Utricularia purpurea* and *U. vulgaris*, the two most common submersed plants, tended to disappear during turbid conditions, but whether this relates to turbidity or the decreased water level is unknown.

Leptolucania-Gambusia Dominance Shift

Turbidity may have had an impact on the apparent shift in dominance within small fishes prior to, and after the restoration/drought. Pygmy killifish (*Leptolucania ommata*), was the dominant small fish found in Tates Hell Swamp (Figure 7-13). However, after the restoration, drought, and subsequent increase in turbidity, mosquitofish (*Gambusia holbrooki*), became the numerical dominant (Figure 7-14). Only during the second to the last sampling date were substantial numbers of *Leptolucania* collected from Tates Hell Swamp, and those were collected from the control ditch. This is significant, because *Gambusia* was rarely collected from the control site, whereas it was dominant in the demonstration sites.

In Florida, *Leptolucania* are found in oligotrophic, clear, soft, colored waters with an abundance of aquatic vegetation (Hoyer and Canfield 1994). These conditions were present in Tates Hell Swamp prior to restoration/drought and during high water periods afterwards. During the severe drought of summer 1998, aquatic vegetation, specifically *Utricularia* spp., disappeared from the ditches of Tates Hell Swamp, and this was accompanied by very turbid conditions. *Leptolucania* abundance may have been affected by one or both of these factors. Feeding may have been impaired, or *Leptolucania* may have been more susceptible to predation because of lack of cover and/or greater visibility. Smith (1992) and Belk (1994) suggested that cover segregated the small poeciliid *Heterandria formosa* from *Gambusia* in areas where they co-occur, with *Heterandria* inhabiting areas with dense vegetation while *Gambusia* inhabit less vegetated areas. In the absence of this vegetation *Heterandria* loses its refugium.

Greater competition from and predation by *Gambusia* may have also had an effect. Predation is one manner in which *Gambusia* are able to negatively affect fish populations (Meffe *et al.* 1983). This can occur not only on juvenile fish, but also on adults and eggs of small taxa. Agonistic behavior also appears to decrease the fitness of the fishes that *Gambusia* attack (Barrier and Hicks 1994; Howe *et al.* 1997), though this behavior is not always displayed by *Gambusia* (Pen and Potter 1991). Agonistic behavior can suppress reproduction (Schoenherr 1981) and cause wounds that are subject to infection

(Arthrington 1991). Competition for food also appears to be a possibility, as both *Gambusia* and *Leptolucania* feed predominantly on macroinvertebrates (Arthrington 1987; Hoyer and Canfield 1994; Pen and Potter 1991). However, studies with *Gambusia* and the least killifish, *Heterandria formosa*, suggests that the interaction between the two is governed more by predation than by competition (Belk and Lydeard 1994; Schaefer *et al.* 1994).

The apparent reestablishment of *Leptolucania* in the control ditch late in the study is possibly a result of low predation from *Gambusia*. Turbid conditions did seem to affect the population of *Leptolucania* present in the control ditch, but *Gambusia* never became the dominant fish there, despite the fact that it occurred in the demonstration sites. Physical alterations of the morphology of the ditches may have led to the dominance of *Gambusia* by providing preferred habitat. Site 1 and the low water crossings specifically provided broad, relatively shallow waters, and these sites accounted for 84% of the total *Gambusia* collected during the course of the study. The control ditch did not undergo any physical alterations, and the morphology of the ditches (i.e., deep and steep-sided) may not have provided preferred habitat for *Gambusia*.

Once conditions in Tates Hell Swamp reverted to higher water and non-turbid conditions in autumn 1998, why then did *Leptolucania* not reestablish itself as the dominant small fish at the demonstration sites? Aquatic plants, specifically *Utricularia* spp., did not reappear in the ditches of Tates Hell Swamp until the following spring, although with the wetlands flooded, emergent aquatic vegetation was present. There simply may not have been dense vegetation available for *Leptolucania*. Furthermore, *Leptolucania* spawns from early April to late August in aquatic vegetation (Hoyer and Canfield 1994), so the remaining fish would not have been able to produce young until April 1999. It is also possible that the morphological alterations to the ditches and the creation of permanently-flooded low water crossings did not provide *Leptolucania* a competitive advantage over *Gambusia*, though the mechanism for this is unknown.

In the meantime, *Gambusia* was able to build up a substantial population. *Gambusia* are viviparous (live-bearing) fishes, and they are able to reproduce rapidly through multiple broods each breeding season. Gestation takes approximately 24 days (Baensch and Riehl 1997), and a new brood can be produced every 5-8 weeks (Riehl and Baensch 1997). Anywhere from 10-60 fry are produced (Baensch and Riehl 1997; Riehl and Baensch 1997). So, rapid increases in abundance are possible for *Gambusia* in a relatively short period of time.

Belk (1994) hypothesized that *Gambusia* was able to control *Heterandria* populations only when large populations existed. In Tates Hell Swamp, it was only after the restoration/drought that *Gambusia* populations increased in numbers. This suggests that after the drought/restoration, these large populations of *Gambusia*, through predation, were able to effectively suppress *Leptolucania*, even in the presence of aquatic vegetation. The exception to this was in the control ditch, where *Gambusia* never was collected in substantial numbers. Only at this site did *Leptolucania* appear to be returning in considerable numbers.

Table 7-1: Fishes collected from Tates Hell Swamp, Florida, from April 1998-October 1999, their incidence in habitat type (ditch vs.wetland vs. low water crossing (LWC)), and gear type utilized for collection (Gill net vs. fish trap).

Common name	Scientific name	Habitat			Sampling Method	
		Ditch	Wetland	LWC	Gill Net	Fish Trap
Esocidae						
Esox americanus	redfin pickerel	x	x	x	x	x
Catostomidae						
Erimyzon sucetta	lake chubsucker	x			x	
Ictaluridae						
Ameiurus natalis	yellow bullhead	x	x		x	
Aphredoderida						
Aphredoderus sayanus	pirate perch	x	x	x		x
Cyprinodontidae						
Fundus cingulatus	banded topminnow	x	x	x		x
Fundulus lineolatus	lined topminnow			x		x
Leptolucania ommata	pygmy killifish	x	x	x		x
Poeciliidae						
Gambusia holbrooki	Eastern mosquitofish	x	x	x		x
Elassomatidae						
Elassoma evergladei	Everglades pygmy sunfish	x	x	x		x
Centrarchidae						
Centrarchus macropterus	flier	x			x	
Lepomis gulosus	warmouth	x	x	x	x	x
Lepomis punctatus	spotted sunfish	x			x	
Percidae						
Etheostoma fusiforme	swamp darter			x		x

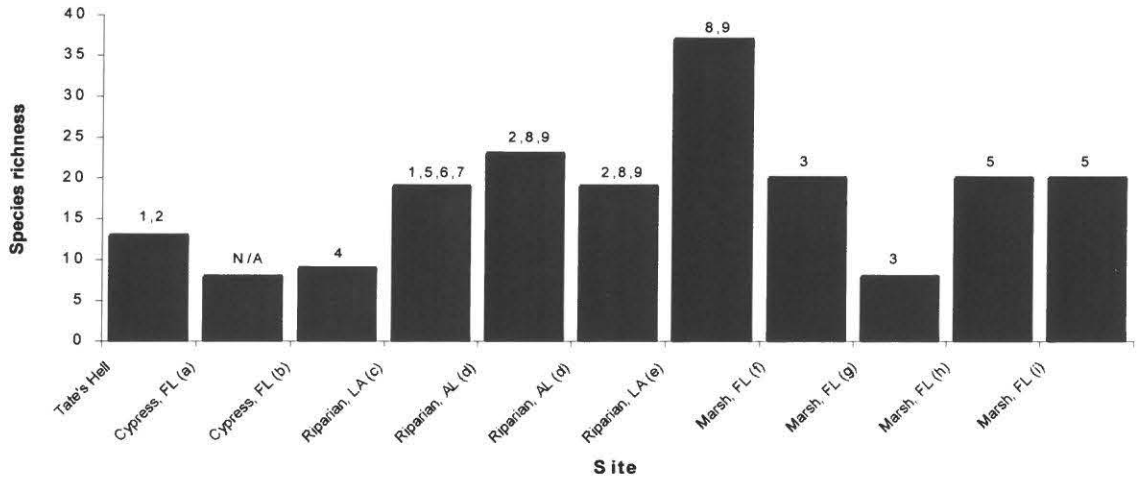


Figure 7-1. Fish species richness in wetlands of the southeastern United States. Numbers indicate method of capture (1, fish trap; 2, gill net; 3, throw trap; 4, Wegener ring; 5, dip net; 6, flag net; 7, seine; 8, electrofishing; 9, trap net). Data sources are: a (Weller 1995), b (Carlson and Duever 1977), c (Pezold 1998), d (Knight and Bain 1996), e (Pollard *et al.* 1983), f (Jordan *et al.* 1998), g (Streever and Crisman 1993), h (Dunson *et al.* 1998), i (Dunson *et al.* 1998).

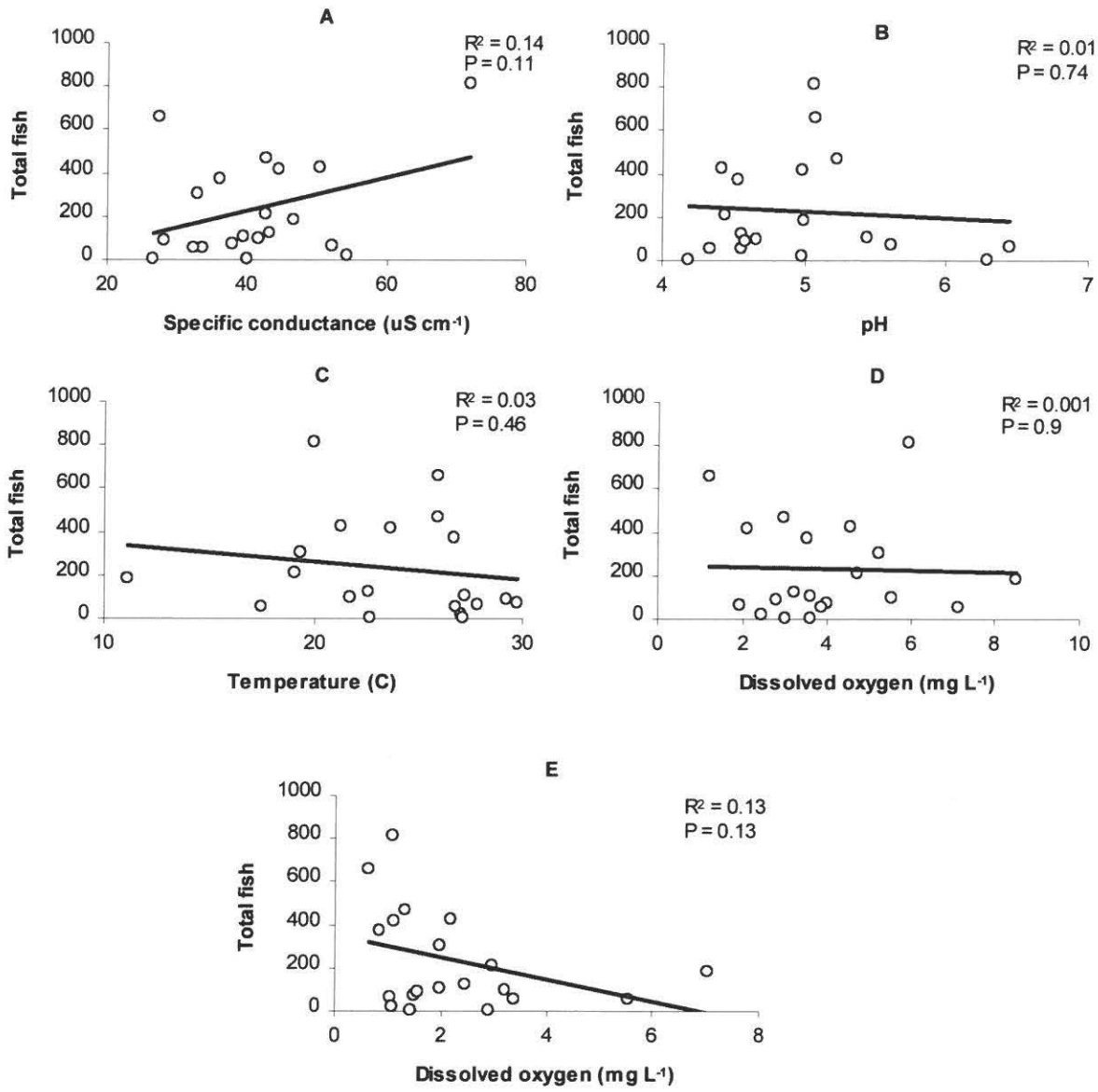


Figure 7-2. Total fish abundance versus environmental variables. A, conductivity; B, pH; C, temperature; D, ditch surface and wetland dissolved oxygen; E, ditch bottom dissolved oxygen.

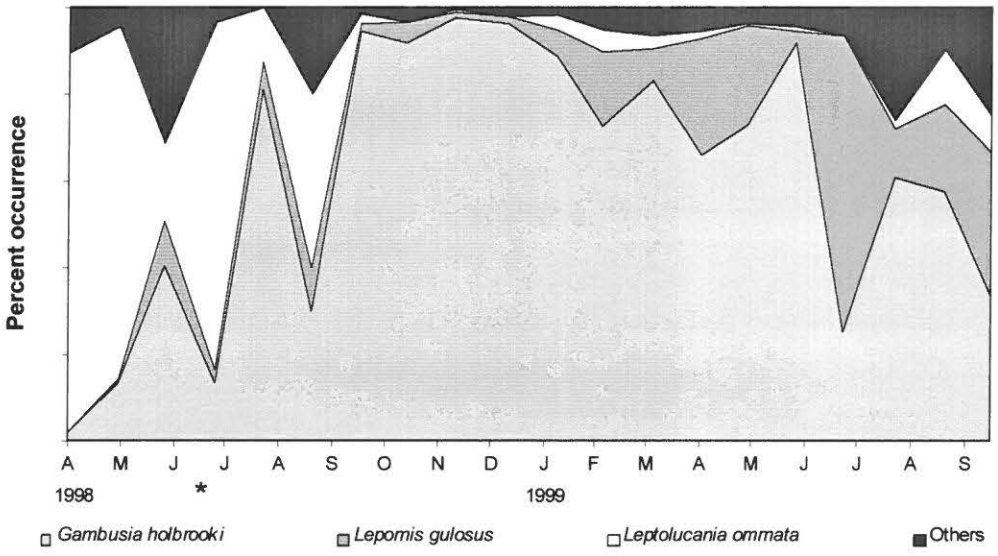


Figure 7-3. Dominance of three fish species collected using fish traps at all sites in Tates Hell Swamp.* indicates initiation of restoration.

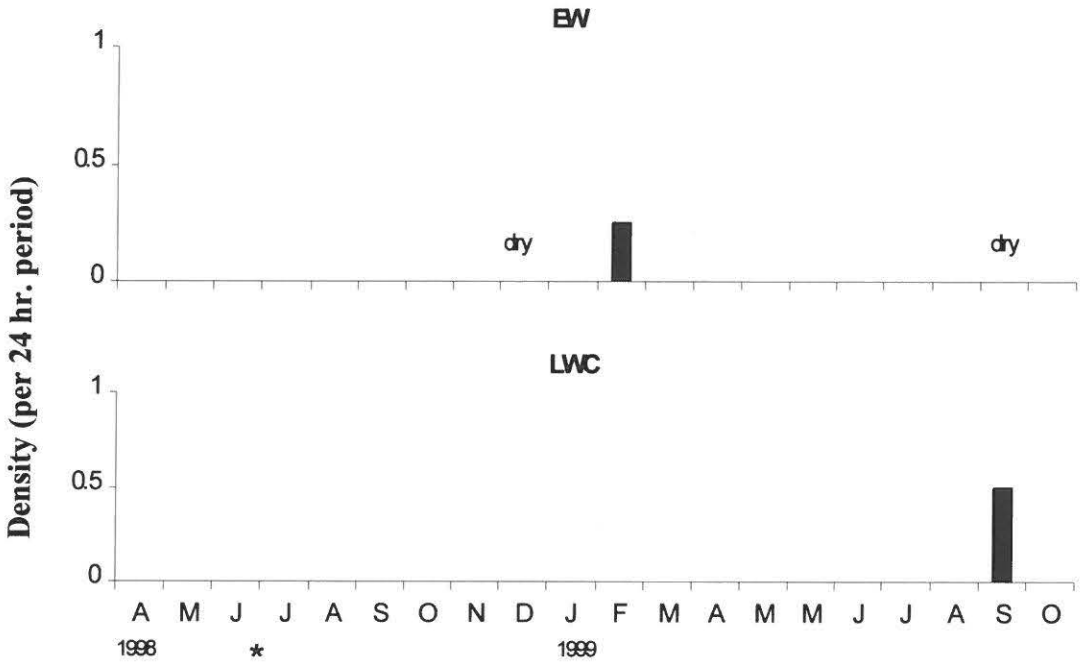


Figure 7-4. Mean density of *Etheostoma fusiforme* collected using fish traps from two habitats in Tates Hell Swamp. Fish traps were set for a 24 hour period. EW- demonstration wetlands and LWC- low water crossings. No specimens were collected from demonstration ditches, control ditches, or control wetlands. * indicates initiation of restoration.

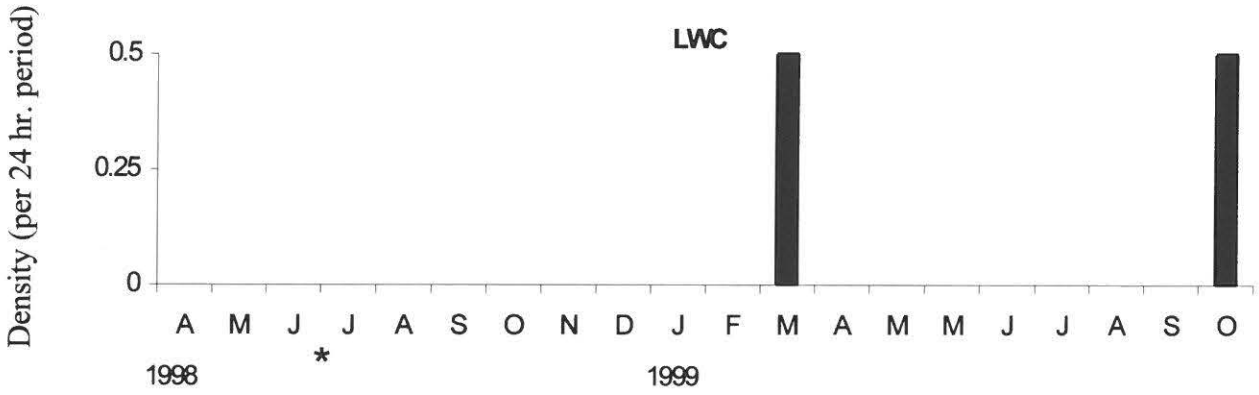


Figure 7-5. Mean density of *Fundulus lineolatus* collected using fish traps from one habitat in Tates Hell Swamp. Fish traps were set for a 24 hour period. LWC- low water crossings. No specimens were collected from demonstration ditches, demonstration wetlands, control ditches, or control wetlands. * indicates initiation of restoration.

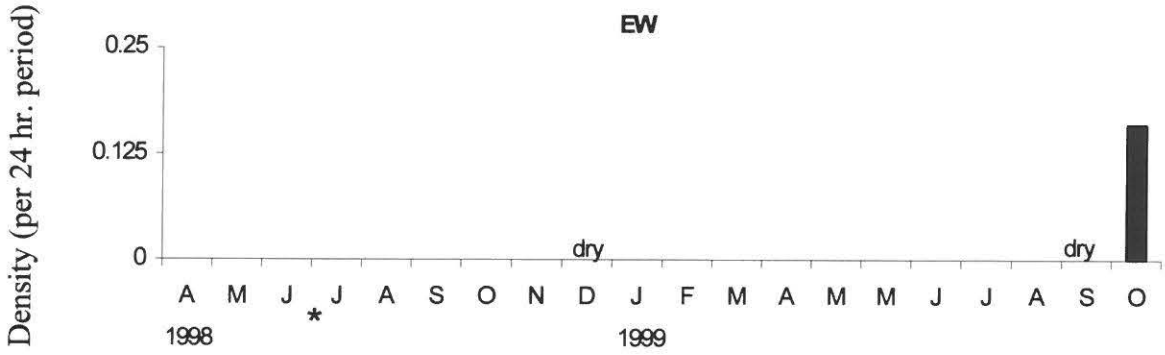


Figure 7-6. Mean density of *Ameiurus natalis* collected using fish traps from one habitat in Tates Hell Swamp. Fish traps were set for a 24 hour period. EW- demonstration wetlands No specimens were collected from demonstration ditches, control ditches, control wetlands, or low water crossings. * indicates initiation of restoration.

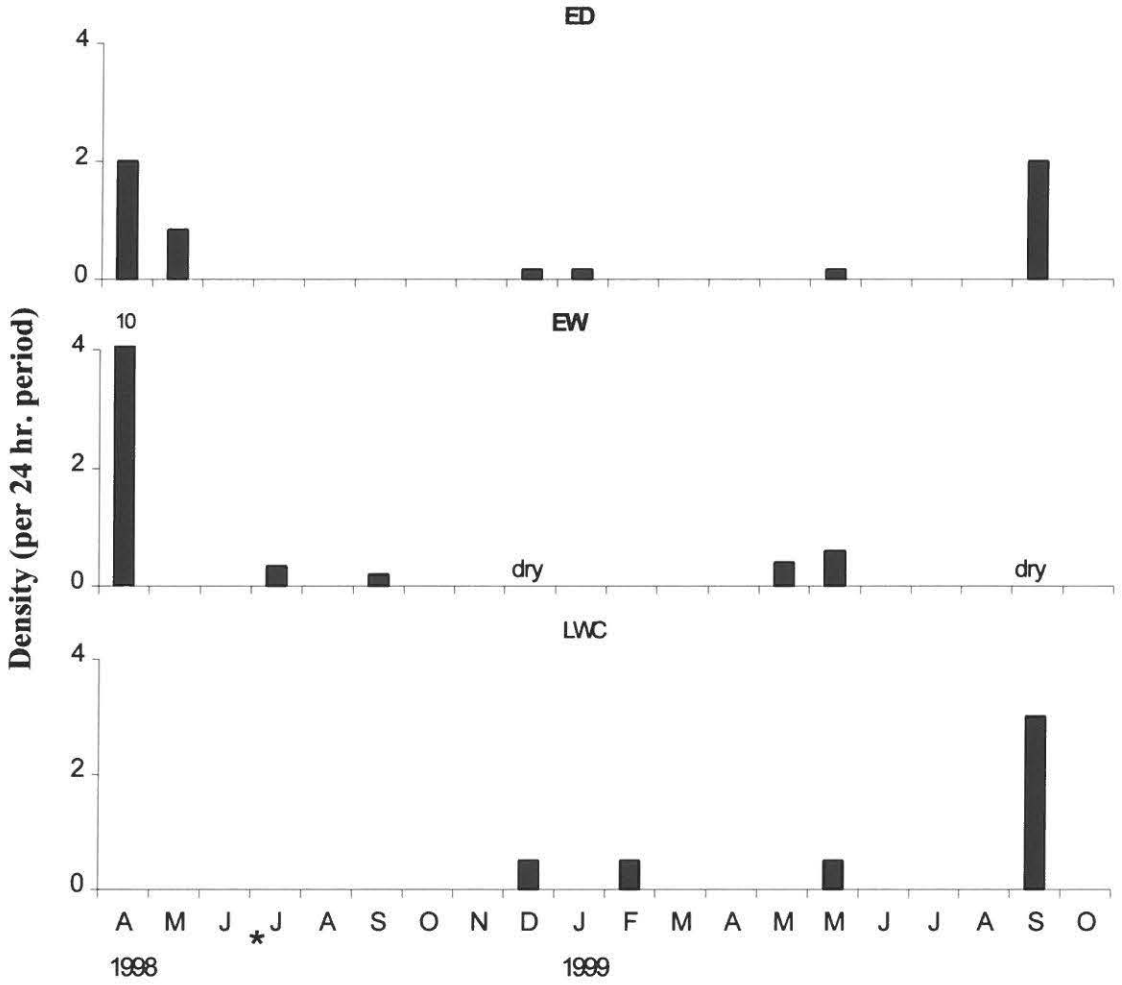


Figure 7-7. Mean density of *Aphredoderus sayanus* collected using fish traps from three habitats in Tates Hell Swamp. Fish traps were set for a 24 hour period. ED- demonstration ditches, EW- demonstration wetlands and LWC- low water crossings. No specimens were collected from the control ditch of control wetland. * indicates initiation of restoration.

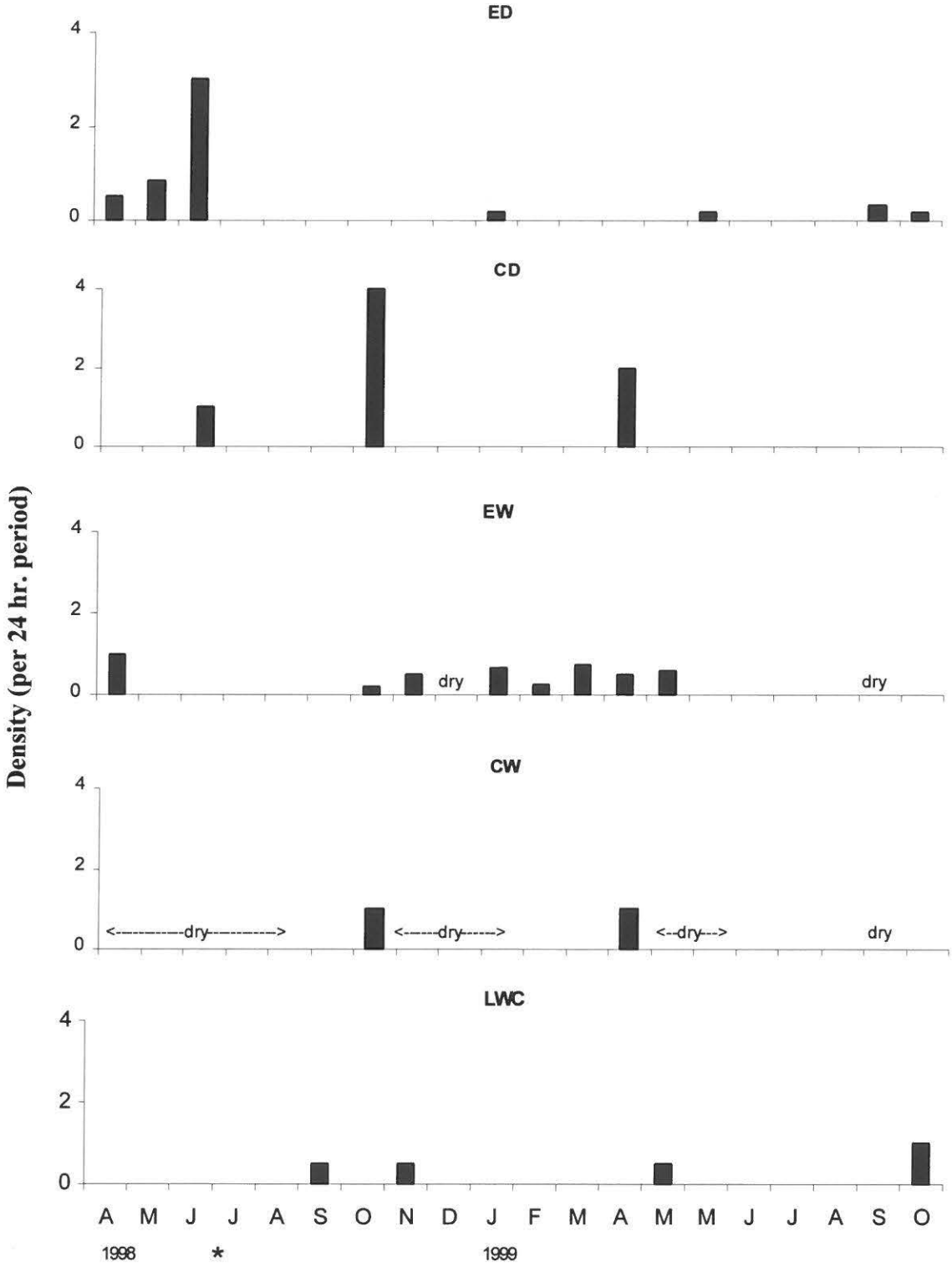


Figure 7-8. Mean density of *Elassoma evergladei* collected using fish traps from five habitats in Tates Hell Swamp. Fish traps were set for a 24 hour period. ED- demonstration ditches, EW- demonstration wetlands, CD- control ditch, CW- control wetland, and LWC- low water crossings. * indicates initiation of restoration.

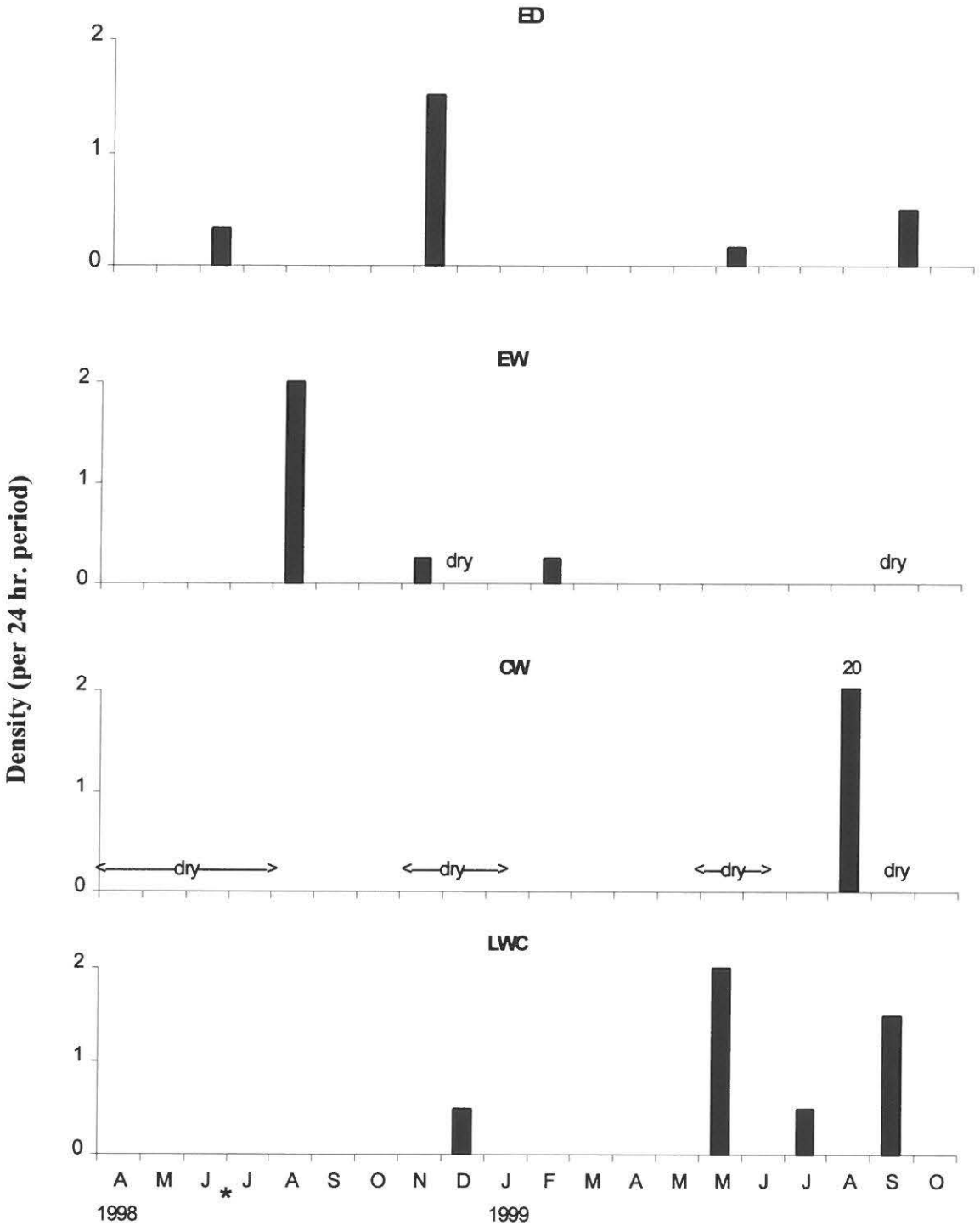


Figure 7-9. Mean density of *Fundulus cingulatus* collected using fish traps from four habitats in Tates Hell Swamp. Fish traps were set for a 24 hour period. ED- demonstration ditches, EW- demonstration wetlands, CW- control wetland and LWC- low water crossings. No specimens were collected from the control ditch. * indicates initiation of restoration.

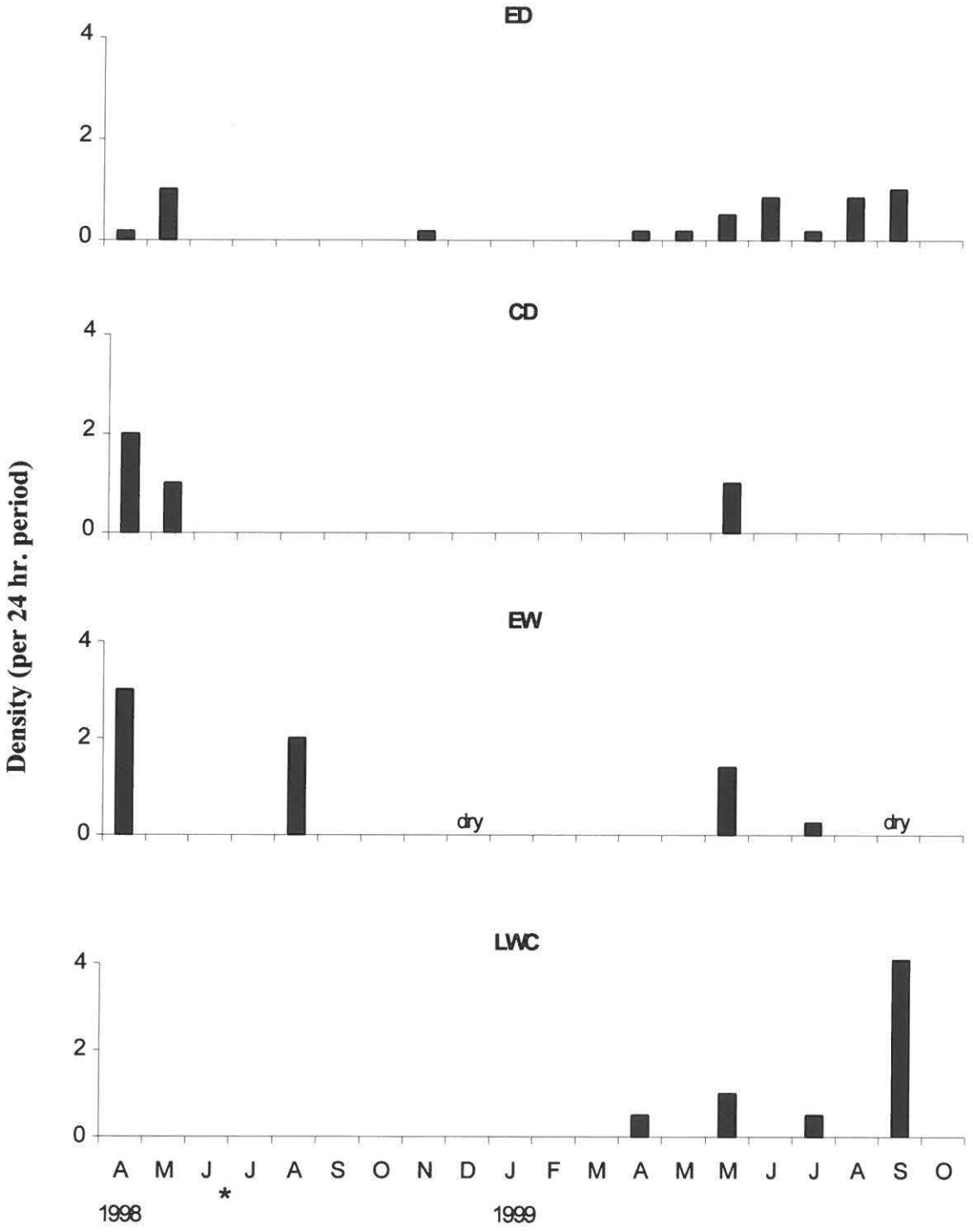


Figure 7-10. Mean density of *Esox americanus* collected using fish traps from four habitats in Tates Hell Swamp. Fish traps were set for a 24 hour period. ED- demonstration ditches, EW- demonstration wetlands, CD- control ditch, and LWC- low water crossings. No specimens were collected from the control wetland. * indicates initiation of restoration.

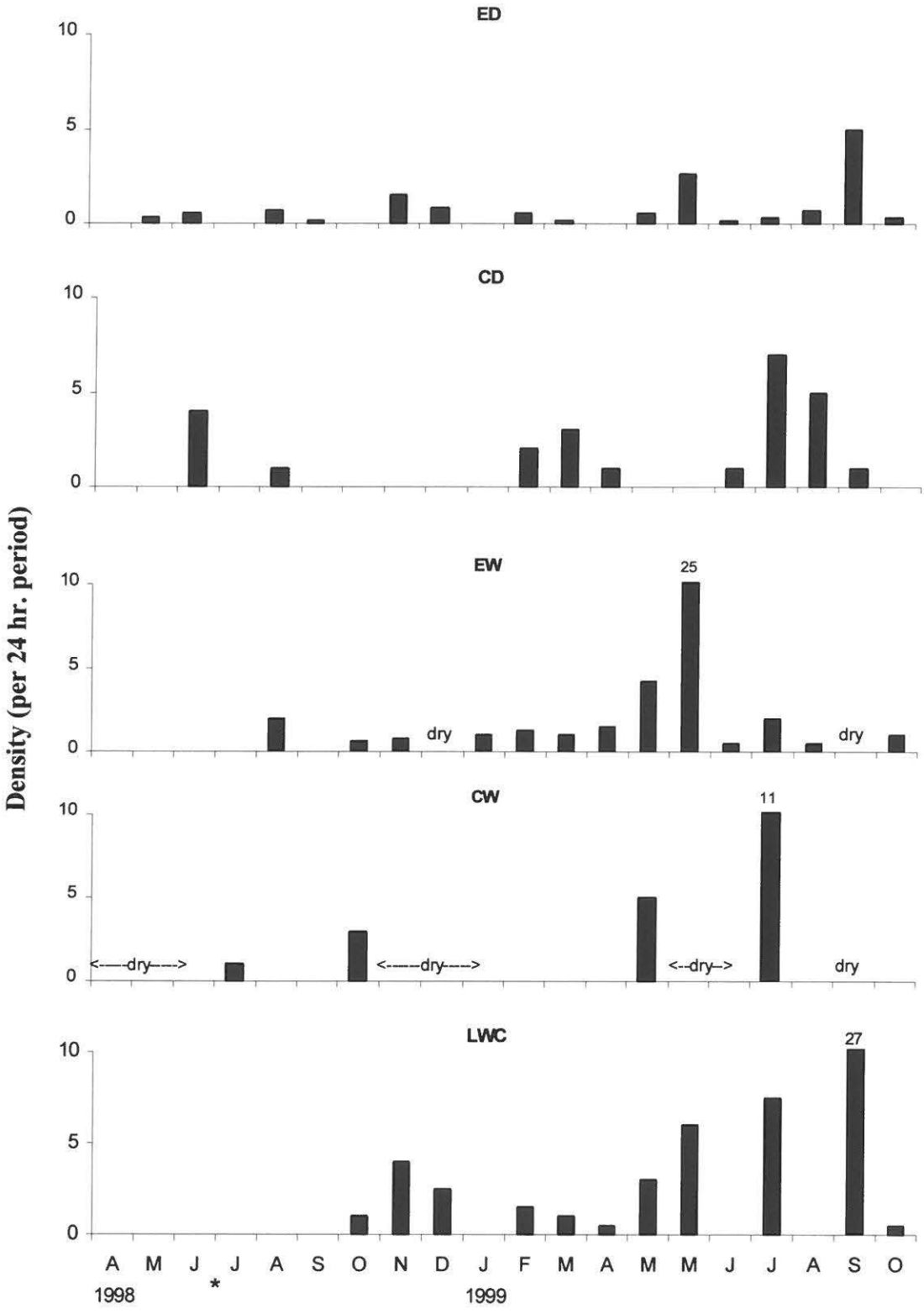


Figure 7-11. Mean density of *Lepomis gulosus* collected using fish traps from five habitats in Tates Hell Swamp. Fish traps were set for a 24 hour period. ED- demonstration ditches, EW- demonstration wetlands, CD- control ditch, CW- control wetland, and LWC- low water crossings. * indicates initiation of restoration.

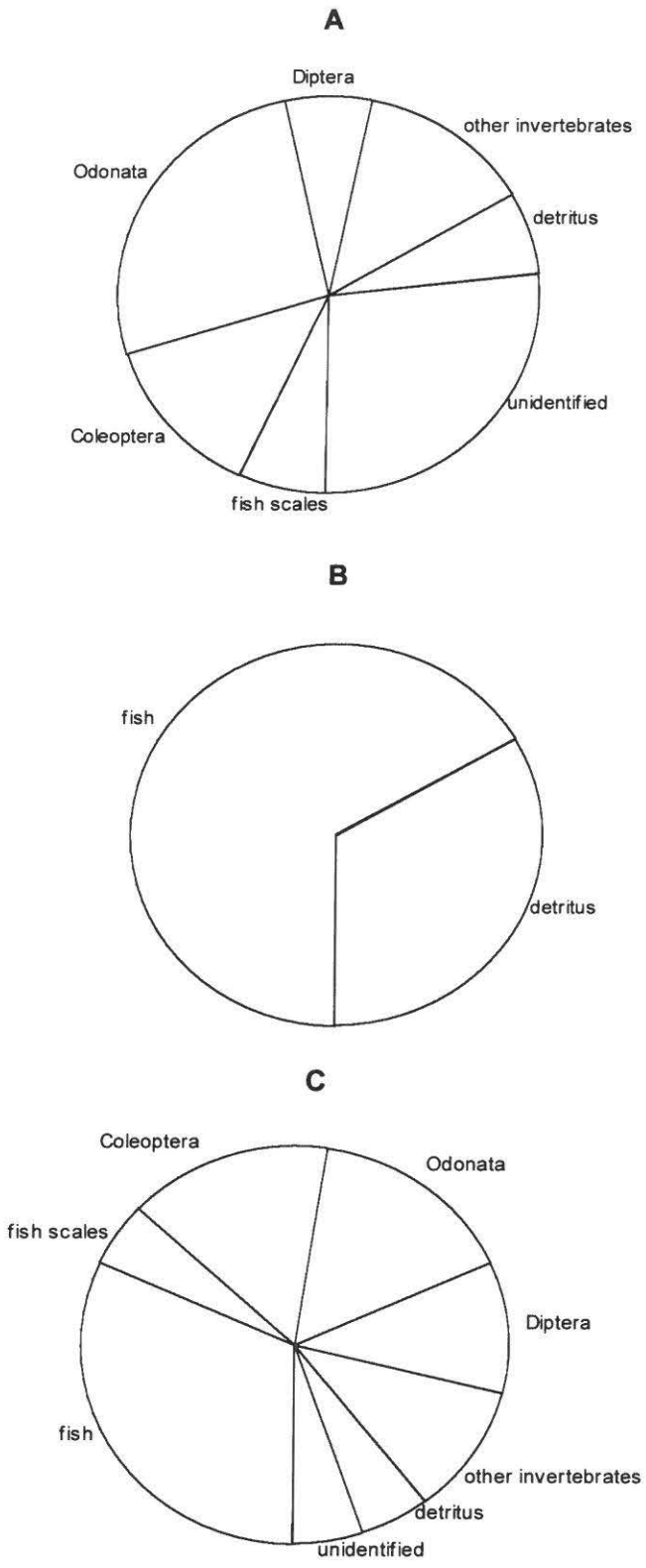


Figure 7-12. Diet of three large, predatory fish collected from ditches in Tates Hell Swamp using gill nets. A, warmouth, *Lepomis gulosus*; B, redfin pickerel, *Esox americanus*; C, yellow bullhead, *Ameiurus natalis*.

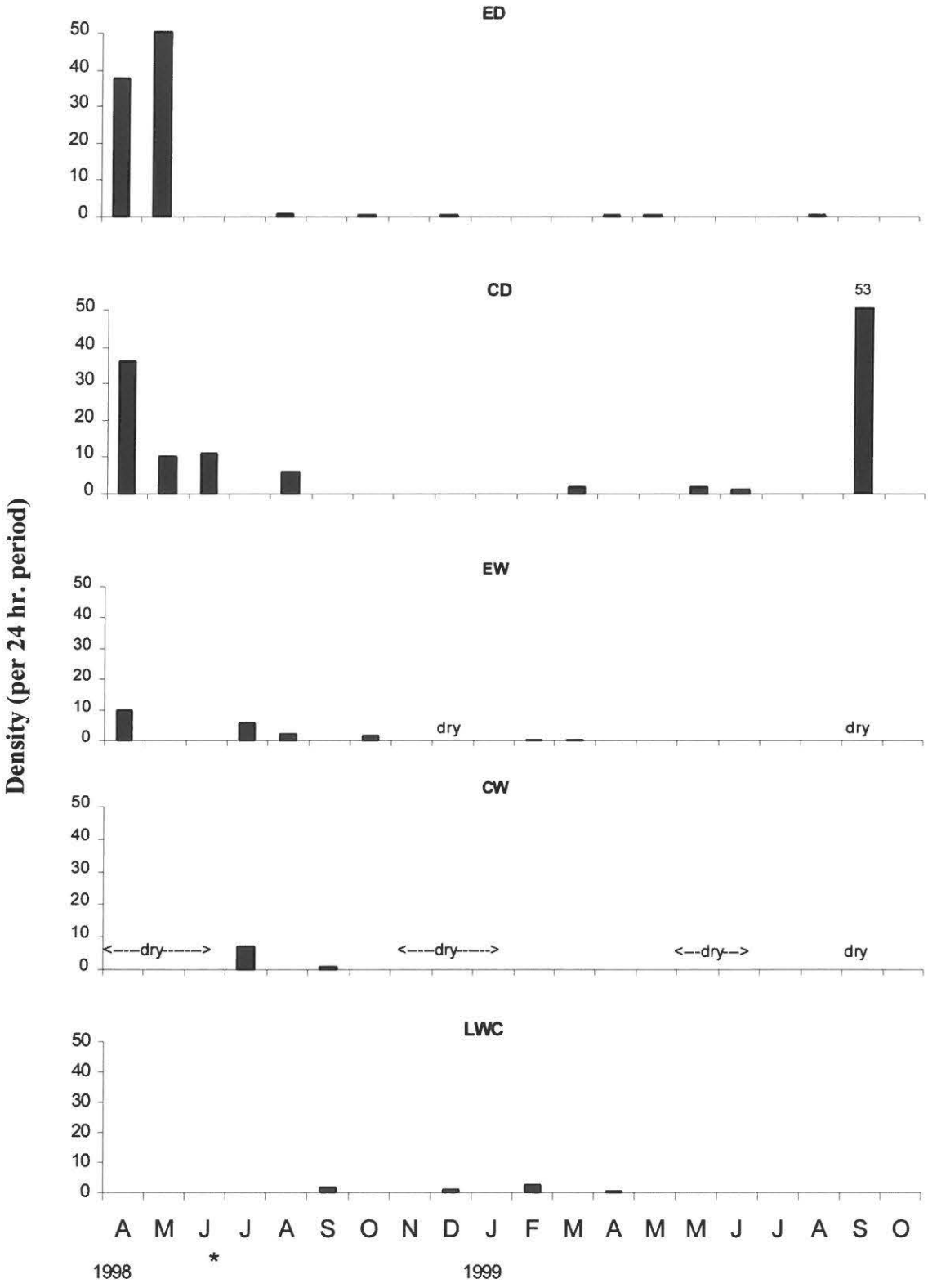


Figure 7-13. Mean density of *Leptolucania ommata* collected using fish traps from five habitats in Tates Hell Swamp. Fish traps were set for a 24 hour period. ED- demonstration ditches, EW- demonstration wetlands, CD- control ditch, CW- control wetland, and LWC- low water crossings. * indicates initiation of restoration.

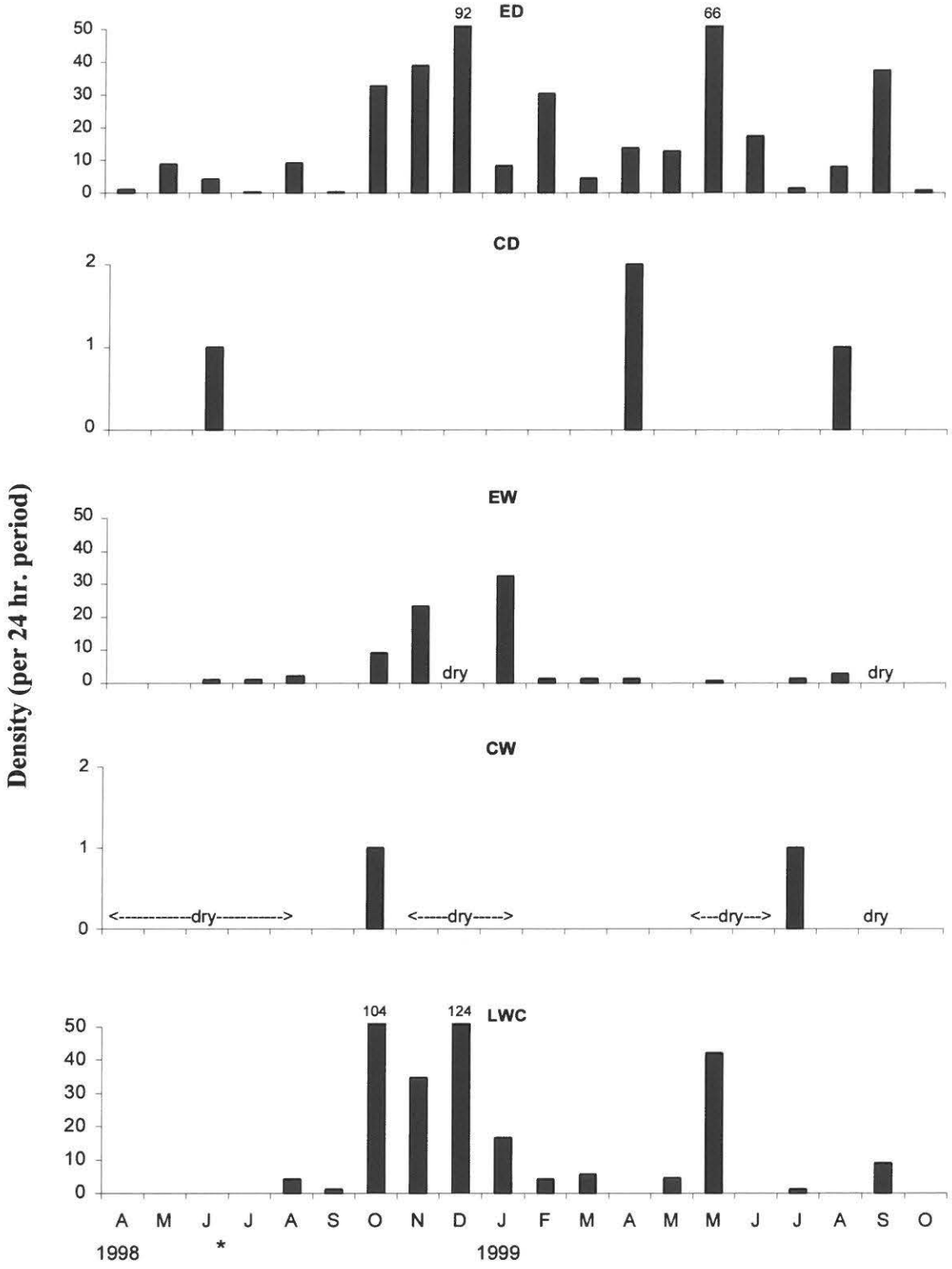


Figure 7-14. Mean density of *Gambusia holbrooki* collected using fish traps from five habitats in Tates Hell Swamp. Fish traps were set for a 24 hour period. ED- demonstration ditches, EW- demonstration wetlands, CD- control ditch, CW- control wetland, and LWC- low water crossings. * indicates initiation of restoration.

COMMUNITY METABOLISM

Community primary productivity and respiration—which together constitute community metabolism—reflect the integrated biological activity of natural systems. Little is known about what constitutes "normal" community metabolism in wetland systems, and the effects of disturbances on community metabolism are also poorly understood (Adamus and Brandt 1990). Heavy nutrient or organic loadings tend to increase metabolism, and toxic pollutants can reduce metabolism, but the effects, if any, of subtler disturbances such as hydrologic alteration cannot be predicted. The metabolism data collected in this study are therefore intended to provide a reference framework for wetland streams of the type found in Tates Hell.

Measurement of community metabolism by the diel DO method is notoriously difficult and subject to error (Greeson 1985). A simpler surrogate for *potential* gross primary productivity can be obtained by measuring rates of algal growth. Algal productivity in wetlands is typically dominated by periphyton rather than phytoplankton (Kadlec and Knight, 1995). Low phytoplankton chlorophyll a levels found in the present study (all 1.0 μg/L or lower) suggest that this holds true for Tates Hell. Therefore, potential algal productivity was estimated by measuring periphyton growth. Periphyton growth data do not estimate absolute primary productivity, but provide relative estimates of potential productivity that can be used to make comparisons among different sites and times.

A number of problems were encountered in obtaining diel dissolved oxygen curves. Membrane fouling or other equipment failure resulted in loss of many data sets. Others were invalidated by rain or other weather effects—sunny, relatively stable weather is required. As a result, only three usable curves were obtained: August 1998 at the demonstration site, October 1998 at the control site, and November 1998 at the demonstration site. Calculated respiration and productivity values are presented in Table 8-1. Both respiration and GPP showed declining trends from summer through late fall, as would be predicted from seasonal day length, solar irradiance, and temperature patterns. Periphyton chl-a declined proportionally to GPP, suggesting both that the GPP data are reasonable, and that chl-a production can serve as a surrogate for GPP.

Calculated NPP was negative in all three cases. This result may appear counter-intuitive, since long-term negative NPP ultimately will result in complete depletion of organic matter in the system. The negative NPP values are in fact quite reasonable because these data only reflect processes occurring within the water column. In most wetland systems, emergent vegetation accounts for abundant above-water productivity. Litter fall in turn constitutes a large carbon subsidy to the water column, with subsequent litter decomposition fueling water column respiration. Thus, negative water column NPP is the norm for systems with extensive emergent plant growth.

Periphyton chl-a data was not collected prior to restoration, but sufficient information was collected to allow comparisons between the control and demonstration sites.. Both

sites showed strong seasonal patterns, but not as strong as those shown in Table 8-1. The demonstration site was consistently higher in periphyton production than the control site (Figure 8-1). While this suggests greater GPP in the demonstration site, it cannot be determined if this effect is attributable to the restoration.

The metabolism data obtained in this study are consistent with those seen in stable, mature aquatic systems (Odum, 1956). The modest increase in productivity seen in the demonstration site, if real, suggests that restoration activities have had at most a minor impact on community metabolism. Given the vagaries of community metabolism measurement, further work in this area would likely yield few new insights.

Table 8-1. Respiration, productivity, and periphyton chlorophyll-a production at demonstration and control sites.

Site	Date	Respiration	Gross Primary Productivity	Net Primary Productivity	P/R	Avg. T	Periphyton Chl-a*
		----------g/m^2-day----------				--°C--	--mg/m^2-
Demonstration	8/20/98	-6.4	5.2	-1.2	0.81	28.4	7.2
Control	10/15/98	-4.2	2.9	-1.4	0.68	23.0	—
Demonstration	11/15/98	-2.2	1.2	-1.0	0.56	20.3.	1.4

*Demonstration site chl-a values are for station 1 only.

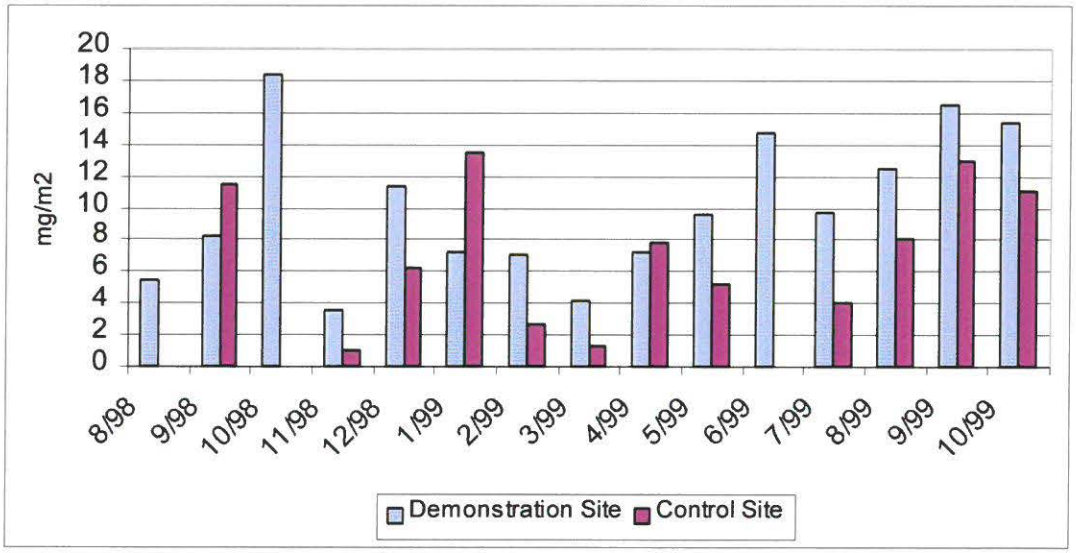


Figure 8-1. Mean monthly periphyton chlorophyll-a production in the demonstration and control sites.

VASCULAR PLANTS

The Tates Hell region is extremely interesting phytogeographically with a very large number of vascular plant species present—several of which are endemics (Anderson 1987, Livingston 1983, Meyer and Ewel 1990, Muller et al. 1989). Hydrologic restoration is expected to reverse ongoing encroachment of upland and weedy species into wetland habitats, and to expand .

Vascular plant work, from May 1998 through October 1999, consisted of field observations and collections to provide an extensive inventory of the vascular plants around the demonstration sites along Big Slough Branch of upper Whiskey George Creek, and at the nearby reference site in Apalachicola National Forest (Figure 2-1). Based on this field work, plant community types in the vicinity of the demonstration site were delineated. In addition, four permanent quadrats were established within the restoration sites in October 1999 to allow an intensive survey of the local plant community structure to provide a basis for evaluating future changes.

Plant Communities

Plant communities at the study sites are wetlands that have been altered by past human activities (Figure 9-1). The seemingly least disturbed plant community is the floodplain swamp (area 1; Figure 9-1). Floodplain swamp encompasses Whiskey George Creek and its Big Slough Branch. This plant community is comprised of woody species such as pond cypress, tupelo or black gum, titi, pop ash, myrtle-leaf holly, gallberry, bayberry, sweetbay, and limited amounts of pine. Herbaceous species are fairly sparse in occurrence; they are mostly "graminoids" (i.e., grasses, sedges and rushes) plus pipewort, yellow-eyed grass, redroot, and clubmoss. Near the confluence of the upper two channels of the creek there are sloughs (shallow, open bodies of water, hence the name "Big Slough Branch") within the cypress swamp; these sloughs have submersed and floating species of bladderwort and emergent species, such as yellow-eyed grass.

A unique plant community straddles North Boundary Road on the north flank of Big Slough Branch that could be called a cypress savanna (area 2). South of the road, this community contains dwarfed pond cypress in a more open setting than that of the main floodplain swamp; associated woody species include slash pine, dwarf blackgum, small-leaved titi, buckwheat titi, St. John's wort, corkwood, and bayberry. The herbaceous flora is more extensive than in the floodplain swamp with swamp tickseed, pitcher plants, yellow flax, water-dropwort, gerardia, and yellow-eyed grass in addition to the graminoid taxa. The dwarf pond cypress-dwarf blackgum "sub" community is the most botanically interesting area (both structurally and in species diversity) in the Whiskey George Creek system. The dwarf blackgum (*Nyssa ursina*) is notable, and West's yellow flax (an endangered species) occurs here. North of the road the cypress savanna gives way to shrubby savanna with little or no cypress, but more pine, titi, hollies, and St. John's wort. The herbaceous flora is similar to that south of the road, but bluestem grass and rush-featherling are frequent here.

The wet flatwood plant communities, bordering the floodplain swamp on all sides, are all secondary and considerably altered. Extensive logging has created patches of prairie or savanna-

like areas in a mosaic among the secondary, open pinewoods (mostly slash pine). An isolated area of wet flatwood is found within the floodplain (area 3).

A fourth plant community, mixed hardwood swamp/shrub bog, is located in the southeast portion of the Big Slough Branch site (area 4). This community consists of a mosaic of forested and shrub/herbaceous wetlands. Forested areas are dominated by sweetbay,swamp bay, and scattered cypress. Large gallberry, titi, and scattered slash pine are also present. Shrub/herbaceous areas are characterized by St. John's Wort, yellow-eyed grass, redroot, and a variety of beak-rushes and grasses.

Intensive Survey

Four permanent quadrats were laid out for intensive study of plant community structure and possible responses to restorative activities (Figure 9-1). Each quadrat was marked with metal border stakes and flagging. Quadrats, each measuring 2 x 15 meters, were established on the north and south sides of North Boundary Road near and are identified as Quadrat 2-N and Quadrat 2-S, respectively. Similarly, quadrats, 2 x 15 meters, were established east and west of the road near Low Water Crossing #5 and are identified as Quadrat 5-E and Quadrat 5-W.

Quadrats 2-N and 2-S both are both located within the cypress savanna community shown in Figure 9-1. Quadrat 2-N could be described more specifically as a savanna-swamp community and contained 24 species (Table 9-1). Quadrat 2-S is an open, dwarf blackgum-dwarf cypress swamp. It had the greatest diversity among the four quadrats with 31 species present (Table 9-2).

Quadrats 5-E and 5-W are located within the general floodplain swamp community. Quadrat 5-E is a more dense cypress swamp. It had a higher density of woody plants than the other three quadrats, but it also had the lowest number of species (i.e., 15; Table 9-3). Quadrat 5-W is a wet savanna with some woody plant intrusion; 23 species were encountered here (Table 9-4).

It should be noted that the plant quadrats in this study are located in relatively wet portions of the demonstration site. These locations were selected because plant communites at most of the drier portions of the site had been severely disturbed by logging activities, and because these wet sites still retain some natural characteristics.

While the plant assemblages in the quadrats generally indicate high-quality, relatively intact swamp/savanna communities, encroachment by upland or weedy species was observed in all quadrats. The abundance of young slash pine (*Pinus elliottii*) in Quadrat 2-N is likely a reflection of the dry conditions that resulted from historic drainage activities. The presence of slash pine—even at low frequencies—in Quadrat 5-E also suggests excessive drainage, as this quadrat is located in the heart of a cypress strand. Hydrologic restoration may not eliminate slash pine in these locations, but is expected to effectively curb further encroachment. The weedy species broom sedge (*Andropogon virginicus*) has encroached in Quadrats 2-N and 5-W, likely as a result of both drainage and past soil disturbance. Hydrologic restoration and cessation of soil disturbance are expected to reduce or eliminate this species. A number of species (e.g., *Agalinis linifolia*, *Linum westii*, *Fuirena breviseta*, *Ludwigia linifolia*) that are normally found at the edges of wetlands are also expected to disappear from the quadrats.

In addition to hydrologic restoration, the long-term management plan for Tates Hell incorporates a prescribed burning program. Several woody species encountered in the intensive quadrats are expected to diminish in abundance with periodic burning. These include the aggressive, weedy titi (*Cyrilla racemiflora*) and buckwheat tree (*Cliftonia monophylla*), as well as less agressive species such as the various St. John's worts (*Hypericum sp.*).

Extensive Survey

Extensive site survey of plant species in and around Big Slough Branch of Whiskey George Creek was conducted in the field, and voucher specimens for most species were filed in the Florida State University Herbarium. In Table 9-5, species are listed in major groups (i.e., ferns, gymnosperms, monocots, and dicots) and then by family. Species names are in bold letters followed by authorship. Identifications generally follow nomenclature found in Clewell (1985) unless more recent studies with taxonomical revisions have been published. Common names are also listed in capitals letters. Most occurrences were at or near the low water crossings shown on Figure 3-1.

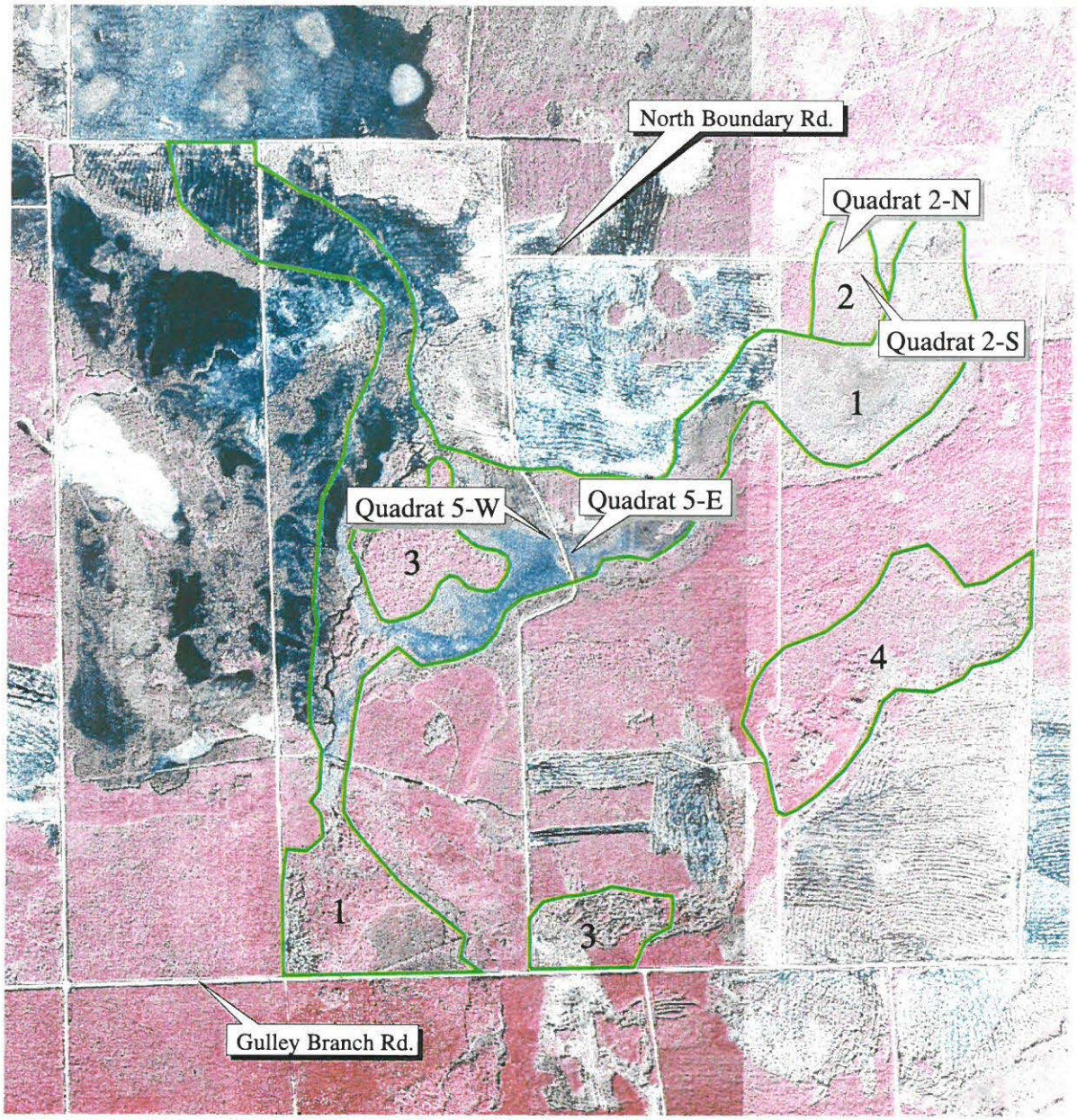
A total of 295 observed taxa are listed in Table 9-5. Families that had good representation included Poaceae with 47 taxa, Cyperaceae with 42, and Asteraceae with 30. Surprisingly, the legumes (Fabaceae), which usually rank fourth in species richnessin area surveys in the southeastern United States, were represented by only four species (one of which, scareweed, was only seen near Tower Road west of the study area). Genera that were well represented included *Panicum* (16), *Rhynchospora* (15), *Xyris* (11), *Juncus* (9), and *Hypericum* and *Rhexia*, each with eight species.

Three species listed in Florida as endangered (Coile 1998) were found in the study sites: *Linum westii, Hymenocallis henryae,* and *Rhexia parviflora.* The following species, listed as threatened in Florida, were found: *Cleistes divaricata, Nolina atopocarpa, Pogonia ophioglossoides,* and *Sarracenia psitticina.*

Extensive survey of vascular plant species was also conducted at the relatively undisturbed reference site (Figure 2-1) a few miles SSW of the demosntration site in the Apalachicola National Forest. Table 9-6 lists a total of 200 taxa that were found at this site which is an open, wet savanna in pine flatwoods bordered by a cypress stringer. This site, though much smaller in area than the demonstration site, contains several state-listed threatened or endangered species, such as *Asclepias viridula, Hymenocallis henryae, Justicia crassifolia, Lilium castesbaei, Parnassia caroliniana, Physostegia godfreyi, Pinguicula lutea, Pinguicula planifolia, Pogonia ophioglossoides, Sarracenia psitticina,* and *Verbesina chapmanii.*

Many of the rich variety of wetland plant species found in the demonstration site are present only in relict populations. Native plant communities throughout Tates Hell have been disturbed to varying degrees by altered hydrology, fire exclusion, and destructive logging practices. These disturbances have been well-documented by the Florida Natural Areas Inventory (Kindell 1997).

Figure 9-1: Plant communities and intensive plant monitoring quadrats.



Plant Community:

1 - Floodplain swamp
2 - Cypress swamp
3 - Wet flatwoods
4 - Mixed bay swamp / shrub bog

0 0.5 1 1.5 Miles

N

Table 9-1. Vascular plant species found at Quadrat 2-N.

Taxon	Frequency
Andropogon virginicus var. virginicus	F
Aristida palustris	C
Cliftonia monophylla	F
Cyrilla racemiflora var. parvifolia	F
Drosera capillaris	F
Hypericum brachyphyllum	I
Hypericum chapmanii	F
Ilex glabra	I
Ilex myrtifolia	I
Juncus trigonocarpus	C
Lachnanthes caroliniana	C
Ludwigia pilosa	F
Nyssa ursina	F
Pinus elliottii	F
Pleea tenuifolia	C
Rhynchospora cephalantha	F
Rhynchospora sp.	I
Sarracenia flava	F
Schoenolirion elliottii	F
Scleria baldwinii	C
Smilax laurifolia	I
Utricularia juncea	F
Utricularia purpurea	F
Xyris stricta	F

C = common at site (20-40% cover), F = frequent (5-20%), and I = infrequent (<5%)

Table 9-2. Vascular plant species found at Quadrat 2-S.

Taxon	Frequency
Agalinis linifolia	F
Aristida palustris	C
Bartonia paniculata	F
Cliftonia monophylla	F
Coreopsis nudata	I
Cyrilla racemiflora var. parvifolia	F
Drosera capillaris	F
Eriocaulon compressum	C
Fuirena breviseta	I
Hypericum brachyphyllum	F
Hypericum chapmanii	C
Ilex myrtifolia	I
Linum westii	F
Ludwigia linifolia	F
Myrica heterophylla	F
Nyssa ursina	C
Oxypolis filiformis	F
Rhynchospora careyana	F
Rhynchospora cephalantha	F
Rhynchospora filifolia	F
Rhynchospora harperi	C
Sarracenia flava	F
Sarracenia psittacina	F
Schoenolirion elliottii	F
Scleria baldwinii	F
Smilax laurifolia	I
Stillingia aquatica	I
Taxodium ascendens	F
Utricularia purpurea	C
Xyris serotina	I
Xyris stricta	F

C = common at site (20-40% cover), F = frequent (5-20%), and I = infrequent (<5%)

Table 9-3. Vascular plant species found at Quadrat 5-E.

Taxon	Frequency
Aristida palustris	C
Coreopsis sp.	I
Cyrilla racemiflora var. parvifolia	F
Eriocaulon decangulare	C
Fraxinus caroliniana	F
Hypericum brachyphyllum	F
Ilex myrtifolia	F
Lycopodium prostratum	I
Pinus elliottii	I
Rhynchospora careyana	C
Rhynchospora cephalanthus	F
Rhynchospora harperi	C
Smilax laurifolia	I
Taxodium ascendens	F
Xyris stricta	I

C = common at site (20-40% cover), F = frequent (5-20%), and I = infrequent (<5%)

Table 9-4. Vascular plant species found at Quadrat 5-W.

Taxon	Frequency
Andropogon virginicus	F
Aristida palustris	C
Aristida stricta	A
Cliftonia monophylla	F
Cyrilla racemiflora var. parvifolia	F
Eupatorium mohrii	I
Hypericum brachyphyllum	F
Hypericum nitidum	F
Hypericum reductum	I
Ilex myrtifolia	F
Lycopodium alopecuroides	I
Panicum sp.	I
Pinus elliottii	I
Rhexia alifanus	F
Rhexia mariana	I
Rhexia lutea	I
Rhynchospora careyana	I
Rhynchospora chapmanii	F
Rhynchospora oligantha	I
Scleria triglomerata	F
Smilax laurifolia	I
Taxodium ascendens	F
Xyris sp.	I

A = abundant at site (>40% cover), C = common (20-40%), F = frequent (5-20%), I = infrequent (<5%)

Table 9-5. Vascular plant taxa found in or near the Big Slough Branch demonstration site.

FERNS AND FERN ALLIES
Lycopodium alopecuroides L. FOXTAIL CLUBMOSS.
Lycopodium prostratum Harper. FEATHER-STEM CLUBMOSS.
Lycopodium carolinianum L. SLENDER CLUBMOSS.
Lygodium japonicum (Thunb.) Sw. CLIMBING FERN.
Osmunda regalis L. ROYAL FERN.
Polypodium polypodioides (L.) Watt. RESURRECTION FERN.
Pteridium aquilinum (L.) Kuhn BRACKEN.
Thelypteris hispidula (Decne.) Reed var. **versicolor** (R.St.John) Lellinger. HAIRY MAIDEN FERN.
Woodwardia areolata (L.) Moore. NETTED CHAIN-FERN.
Woodwardia virginica (L.) J. E. Smith. VIRGINIA CHAIN-FERN.

GYMNOSPERMS
CUPRESSACEAE
Chamaecyparis thyoides (L.) BSP. ATLANTIC WHITE-CEDAR.
PINACEAE
Pinus elliottii Engelm. SLASH PINE.
Pinus palustris Mill. LONGLEAF PINE.
Pinus serotina Michx. POND PINE.
TAXODIACEAE
Taxodium ascendens Brongn. PONDCYPRESS.

ANGIOSPERMS: MONOCOTYLEDONS
AGAVACEAE
Nolina atopocarpa Bartlett. FLORIDA BEARGRASS.
ALISMATACEAE
Sagittaria graminea Michx. var. **chapmanii.**
Sagittaria graminea Michx. var. **graminea** GRASS-LEAVED ARROWHEAD.
AMARYLLIDACEAE
Hymenocallis henryae Traub. GREEN SPIDER LILY.
ARECACEAE (Palmae)
Serenoa repens (Bartr.) Small. SAW-PALMETTO.
BROMELIACEAE
Tillandsia usneoides (L.) L. SPANISH MOSS.
CYPERACEAE
Bulbostylis ciliatifolia (Ell.) Fern. var. **ciliatifolia.** HAIRSEDGE.
Bulbostylis ciliatifolia (Ell.) Fern. var. **coarctata** (Ell.) Kral.
Carex glaucescens Ell. WAXY SEDGE.
Carex striata Michx. WALTER'S SEDGE.
Carex verrucosa Muhl. WARTY SEDGE.
Cyperus compressus POORLAND FLATSEDGE.
Cyperus croceus Vahl. FLATSEDGE.
Cyperus haspan L. SOFT-STEM FLATSEDGE.
Cyperus lecontei Torr. LECONTE'S FLATSEDGE.
Cyperus retrorsus Chapm. RETROSE FLATSEDGE.
Dulichium arundinaceum (L.) Britt. SHEATHED GALINGALE.
Eleocharis baldwinii (Torr.) Chapm. ROADGRASS.
Eleocharis geniculata (L.) R.& S. CLUSTERED SPIKEMOSS.
Eleocharis microcarpa Torr. SMALL-FRUIT SPIKERUSH.

Table 9-5 (continued)

Eleocharis obtusa (Willd.) Schultes in R. & S. BLUNT SPIKERUSH.
Eleocharis tuberculosa (Michx.) R. & S. BIG-CAP SPIKERUSH.
Fimbristylis autumnalis (L.) R. & S. SLENDER FIMBRY.
Fimbristylis tomentosa Vahl. WOOLLY FIMBRY.
Fuirena breviseta (Cov.) Cov. in Harper. SALTMARSH UMBRELLA-SEDGE.
Fuirena pumila (Torr.) Spreng. DWARF UMBRELLA-GRASS.
Lipocarpha micrantha (Vahl) G.Webster. DWARF BULLRUSH.
Psilocarya nitens (Vahl) Wood. BALD-RUSH.
Rhynchospora baldwinii Gray. BALDWIN'S BEAK-RUSH.
Rhynchospora careyana Fern. HORNED-RUSH.
Rhynchospora cephalantha Gray var. **pleiocephala** Fern. & Gale.
Rhynchospora chapmanii Curtis. CHAPMAN'S BEAK-RUSH
Rhynchospora ciliaris (Michx.) Mohr.
Rhynchospora divergens Chapm. ex M. A. Curtis SPREADING BEAK-RUSH.
Rhynchospora fascicularis (Michx.) Vahl BEAK-RUSH.
Rhynchospora filifolia Gray. THREAD-LEAF BEAK-RUSH.
Rhynchospora gracilenta Gray. SLENDER BEAK-RUSH.
Rhynchospora harperi Small. CYPRESS-SWAMP BEAK-RUSH.
Rhynchospora latifolia Baldw. (Dichromena) WHITE-TOP SEDGE.
Rhynchospora oligantha Gray. FEW-FLOWER BEAK-RUSH.
Rhynchospora plumosa Ell. PLUMED BEAK-RUSH.
Rhynchospora rariflora (Michx.) Ell. UNCOMMON-FLOWER BEAK-RUSH.
Rhynchospora wrightiana Boeckl. WRIGHT'S BEAK-RUSH.
Scirpus cyperinus (L.) Kunth. WOOL-GRASS.
Scleria baldwinii (Torr.) Steud. BALDWIN'S NUTRUSH.
Scleria georgiana Core. GEORGIA NUTRUSH.
Scleria reticularis Michx. NETTED NUTRUSH.
Scleria triglomerata Michx. TALL NUT-RUSH.

ERIOCAULACEAE

Eriocaulon compressum Lam. HAT-PINS.
Eriocaulon decangulare L. COMMON PIPEWORT.
Syngonanthus flavidulus (Michx.) Ruhl. SHOE-BUTTONS.

HAEMODORACEAE

Lachnanthes caroliniana (Lam.) Dundy. REDROOT.
Lophiola americana (Pursh) Wood. GOLDCREST.

HYPOXIDACEAE

Hypoxis juncea J.E.Smith. YELLOW STAR-GRASS.

JUNCACEAE

Juncus abortivus Chapm. PINEBARREN RUSH.
Juncus coriaceus Mack. LEATHERY RUSH.
Juncus effusus L. SOFT RUSH.
Juncus marginatus Rostk. SHORE RUSH.
Juncus polycephalus Michx. MANY-HEAD RUSH.
Juncus repens Michx. CREEPING RUSH.
Juncus scirpoides Lam. GLOBE RUSH.
Juncus trigonocarpus Steud. RED-TOP RUSH.
Juncus validus Cov. ROUND-HEAD RUSH.

Table 9-5 (continued)

LILIACEAE

Aletris lutea Small. YELLOW COLIC-ROOT.
Pleea tenuifolia Michx. RUSH-FEATHERING.
Schoenolirion elliottii Gray. WHITE SUNNYBELL.
Tolfieldia racemosa (Walt.) BSP. ASPHODEL.

ORCHIDACEAE

Cleistes bifaria (Fern.) Catling & Gregg. ROSEBUD ORCHID.
Pogonia ophioglossoides (L.) Ker. ROSE POGONIA.

POACEAE (Gramineae)

Andropogon glomeratus (Walt.) BSP. var. **glaucopsis** (Ell.) Mohr
Andropogon glomeratus (Walt.) BSP. var. **pumilus** Vasey BUSHY BEARDGRASS
Andropogon gyrans Ashe var. **stenophyllus** (Hack.) Campbell. NARROWLEAF BLUESTEM.
Angropogon virginicus L. var. **glaucus** Hack. LITTLE CHALKY BLUESTEM.
Angropogon virginicus L. var. **virginicus.** BROOM-SEDGE BLUESTEM.
Aristida beyrichiana Trin & Rupr. [*A. stricta*]. WIREGRASS.
Aristida longespica Poir. var. **geniculata** (Raf.) Fern. SLIM THREE-AWN GRASS.
Aristida palustris (Chapm.) Vasey. MARSH THREE-AWN GRASS.
Aristida purpurescens Poir. var. **tenuispica** (Hitchc.) Allred. NARROW ARROW-FEATHER.
Aristida purpurescens Poir. var. **virgata** (Trin.) Allred. ARROW-FEATHER.
Aristida spiciformis Ell. BOTTLEBRUSH THREEAWN.
Axonopus fissifolius (Raddi) Kuhlm. COMMON CARPETGRASS.
Ctenium aromaticum (Walt.) Wood. TOOTHACHE GRASS.
Dactyloctenium aegypticum (L.) Beauv. CROWFOOT GRASS.
Digitaria ciliaris (Retz.) Koeler. SOUTHERN CRABGRASS.
Digitaria ischaemum (Schreb. ex Schw.) Muhl. CRABGRASS.
Eragrostis bahiensis Schrad. ex Schult. BAHIA LOVEGRASS.
Eragrostis elliottii S. Wats. ELLIOTT LOVEGRASS.
Eragrostis refracta (Muhl.) Scribn. COASTAL LOVEGRASS.
Eragrostis secundiflora J.Presl ssp. **oxylepis** (Torr.) Koch. RED LOVEGRASS.
Eremochloa ophiuroides (Munro) Hackel CENTIPEDE GRASS.
Eustachys glauca Chapm. BIG FINGERGRASS.
Eustachys petraea (Sw.) Desv. FINGERGRASS.
Panicum aciculare Desv. ex Poir. PANICGRASS.
Panicum acuminatum Swartz var. **acuminatum.** POINT-TIP PANICGRASS.
Panicum acuminatum Swartz var. **leucothrix** (Nash) Lelong
Panicum acuminatum Swartz var. **longiligulatum** (Nash) Lelong
Panicum chamaelonche Trin.
Panicum commutatum Schultes. VARIABLE PANICGRASS.
Panicum dichotomiflorum Michx. FALL PANICUM.
Panicum erectifolium Nash. MARSH PANICGRASS.
Panicum rigidulum Nees var. **rigidulum**. REDTOP PANICGRASS.
Panicum scabriusculum Ell. TALL SWAMP PANICGRASS.
Panicum scoparium Lam. BROOM PANIC GRASS.
Panicum strigosum Muhl. var. **leucoblepharis** (Trin.) Lelong. SHORT-BRISTLE PANICGRASS.
Panicum tenerum Beyr. ex Trin. BLUE-JOINT PANICGRASS.
Panicum tenue Muhl. WHITE-EDGE PANICGRSS.
Panicum verrucosum Muhl. WARTY PANICGRASS.
Panicum wrightianum Scribn. WRIGHT'S PANICGRASS.
Paspalum dilatatum Poir. in Lam. DALLASGRASS.

Table 9-5 (continued)

Paspalum notatum Flügge. BAHIA GRASS.
Paspalum setaceum Michx. var. **ciliatifolium** (Michx.) Vasey.
Paspalum setaceum Michx. var. **setaceum**. THIN PASPALUM.
Paspalum urvillei Steud. VASEYGRASS.
Saccharum coarctatum (Fern.) R.Webster. SHORT-BEARD PLUMEGRASS.
Saccharum giganteus (Walt.) Pers. SUGARCANE PLUMEGRASS.
Urochloa ramosa (L.) Nguyen. BROWN-TOP MILLET.

PONTEDERIACEAE

Pontederia cordata L. var. **cordata**. PICKERELWEED.

SMILACACEAE

Smilax auriculata Walt. GREENBRIER WILD BAMBOO.
Smilax glauca Walt. WILD SARSAPARILLA.
Smilax laurifolia L. BAMBOO-VINE CATBRIER.
Smilax walteri Pursh. CORAL GREENBRIER.

XYRIDACEAE

Xyris ambigua Beyr. YELLOW-EYED GRASS.
Xyris brevifolia Michx. SHORT-LEAF YELLOW-EYED GRASS.
Xyris baldwiniana Schult. ST. MARY'S GRASS.
Xyris caroliniana Walt. PINELAND YELLOW-EYED GRASS.
Xyris fimbriata Ell. FRINGED YELLOW-EYED GRASS
Xyris flabelliformis Chapm. FAN-LEAF YELLOW-EYED GRASS.
Xyris jupicai L. Rich. COMMON YELLOW-EYED GRASS.
Xyris laxifolia C.Mart. var. **iridifolia** (Chapm.) Kral. IRIS-LEAVED YELLOW-EYED GRASS.
Xyris serotina Chapm. ACID-SWAMP YELLOW-EYED GRASS.
Xyris smalliana Nash. SMALL'S YELLOW-EYED GRASS.
Xyris stricta

ANGIOSPERMS: DICOTYLEDONS

ACERACEAE

Acer rubrum L. ssp. **rubrum**. RED MAPLE.

APIACEAE (Umbellifrae)

Centella erecta (L.f.) Fern. COINWORT SPADELEAF.
Eryngium yuccifolium Michx. var. **synchaetum** (Gray) C.& R. RATTLESNAKE MASTER.
Oxypolis filiformis (Walt.) Britt. COMMON WATER-DROPWORT.

AQUIFOLIACEAE

Ilex cassine L. DAHOON.
Ilex coriacea (Pursh) Chapm. LARGE GALLBERRY SWEET GALLBERRY.
Ilex glabra (L.) Gray. GALLBERRY INKBERRY.
Ilex myrtifolia Walt. MYRTLE-LEAF HOLLY.
Ilex vomitoria Ait. YAUPON.

ASTERACEAE (Compositae)

Ambrosia artemisiifolia L. COMMON RAGWEED.
Aster chapmanii T. & G. SAVANNAH ASTER.
Aster eryngiifolius Torr. & Gray. COYOTE-THISTLE ASTER.
Aster subulatus Michx. var. **elongatus** Bosserdet. ANNUAL SALT-MARSH ASTER.
Aster tortifolius Michx. WHITE-TOPPED ASTER.
Baccharis halimifolia L. SALT BUSH SEA MYRTLE.
Balduina uniflora
Bidens alba

Table 9-5 (continued)

Bidens mitis (Michx.) Sherff. MARSH BEGGAR-TICKS.
Bigelowia nudata (Michx.) DC. RAYLESS GOLDENROD.
Carphephorus pseudoliatris Cass. BRISTLE-LEAF CHAFFHEAD.
Chaptalia tomentosa Vent. SUN-BONNETS PINELAND DAISY.
Cirsium lecontei T. & G. LECONTE'S THISTLE.
Conyza canadensis (L.) Cronq. var. **pusilla** (Nutt.) Cronq. LITTLE HORSEWEED.
Coreopsis linifolia Nutt. NARROW-LEAF TICKSEED.
Coreopsis nudata Nutt. SWAMP TICKSEED.
Erechtites hieracifolia (L.) Raf. FIREWEED.
Eupatorium capillifolium (Lam.) Small. FILIFORM-LEAVED DOG FENNEL.
Eupatorium compositifolium Walt. NARROW-LEAVED DOG FENNEL.
Eupatorium mohrii Greene. MOHR'S THOROUGHWORT.
Eupatorium semiserratum DC. SMALL-FLOWER THOROUGHWORT.
Euthamia tenuifolia (Pursh) Nutt. BUSH GOLDENROD.
Helenium pinnatifidum (Nutt.) Rydb. SOUTHEASTERN SNEEZEWEED.
Iva microcephala Nutt. PIEDMONT MARSH ELDER.
Liatris gracilis Pursh. COMMON BLAZING STAR.
Liatris spicata (L.) Willd. TALL BLAZING-STAR.
Pityopsis graminifolia (Michx.) Nutt. var. **graminifolia**. GOLDEN ASTER.
Pluchea foetida (L.) DC. STINKING CAMPHOR-WEED.
Pluchea rosea Godfrey. ROSY CAMPHORWEED.
Solidago fistulosa Mill. SWAMP GOLDENROD.

CAMPANULACEAE

Lobelia glandulosa Walt. GLADES LOBELIA.
Lobelia paludosa Nutt. PALE LOBELIA.

CARYOPHYLLACEAE

Paronychia baldwinii (T. & G.) Fenzl. ssp. **baldwinii.** WHITLOW-WORT.

CHRYSOBALANACEAE

Licania michauxii Prance. GOPHER APPLE GROUND OAK.

CLETHRACEAE

Clethra alnifolia L. SWEET PEPPERBUSH.

CLUSIACEAE (Guttiferae)

Hypericum brachyphyllum (Spach) Steud. COASTAL PLAIN ST. JOHN'S WORT.
Hypericum chapmanii Adams. CHAPMAN'S ST. JOHN'S WORT.
Hypericum exile Adams SLENDER ST. JOHN'S-WORT.
Hypericum fasciculatum Lam. SANDWEED.
Hypericum gentianoides (L.) BSP. PINEWEED.
Hypericum microsepalum (T. & G.) Gray ex S. Wats. EARLY ST. JOHN'S WORT.
Hypericum nitidum Lam. CAROLINA ST. JOHN'S-WORT.
Hypericum reductum Adams. ATLANTIC ST. JOHN'S-WORT.

CONVOLVULACEAE

Cuscuta indecora Choisy. LARGE-SEED DODDER.
Ipomoea lacunosa L. WHITE MORNING-GLORY.

CYRILLACEAE

Cliftonia monophylla (Lam.) Sarg. BLACK TITI BUCKWHEAT EE.
Cyrilla racemiflora L. var. **racemiflora.** TITI LEATHERWOOD.
Cyrilla racemiflora L. var. **parvifolia** (Raf.) Sarg. SMALL-LEAVED TITI.

Table 9-5 (continued)

DROSERACEAE
Drosera brevifolia Nutt. DWARF SUNDEW
Drosera capillaris Poir. PINK SUNDEW.
ERICACEAE
Gaylussacia mosieri Small. WOOLLY BERRY.
Gaylussacia tomentosa (A. Gray) Small. DANGLEBERRY.
Kalmia hirsuta Walt. WICKY HAIRY LAUREL
Leucothoe racemosa (L.) Gray. FETTERBUSH.
Lyonia ferruginea (Walt.) Nutt. STAGGERBUSH RUSTY LYONIA.
Lyonia lucida (Lam.) K.Koch. FETTERBUSH.
Pieris phillyreifolia (Hook.) DC. VINE-WICKY.
Vaccinium corymbosum L. var. fuscatum (Ait.) Hook. HIGHBUSH BLUEBERRY.
Vaccinium darrowi Camp. GLAUCOUS BLUEBERRY.
EUPHORBIACEAE
Chamaesyce maculata (L.) Small. MILK PURSELANE.
Euphorbia inundata Torr. ex Chapm. FLORIDA PINE SPURGE.
Stillingia aquatica Chapm. CORKWOOD.
FABACEAE (Leguminosae)
Baptisia simplicifolia Croom. SCARE-WEED.
Chamaecrista nictitans (L.) Moench. WILD SENSITIVE PLANT LITTLE PATRIDGE-PEA.
Sesbania vesicaria (Jacq.) Ell. BLADDERPOD.
Tephrosia hispidula (Michx.) Pers. BRISTLY GOATS-RUE.
FAGACEAE
Quercus geminata Small SAND-LIVE OAK SCRUB LIVE OAK.
Quercus minima (Sarg.) Small. DWARF OAK.
Quercus pumila Walt. RUNNING OAK.
GENTIANACEAE
Bartonia paniculata
Bartonia verna (Michx.) Muhl. WHITE SCREW-STEM.
Bartonia virginica (L.) BSP. YELLOW SCREW-STEM.
Sabatia bartramii Wilbur. BARTRAM MARSH PINK.
Sabatia brevifolia Raf. SHORT-LEAF ROSE GENTIAN.
Sabatia macrophylla Hook. LARGE-LEAF ROSE GENTIAN.
HALORAGACEAE
Proserpinaca pectinata Lam. MERMAID-WEED.
LAMIACEAE (Labiatae)
Physostegia godfreyi Cantino. OBEDIENT FLOWER.
LAURACEAE
Persea palustris (Raf.) Sarg. SWAMPBAY
LENTIBULARIACEAE
Pinguicula sp. BUTERWORT. (not in bloom)
Utricularia cornuta Michx. HORNED BLADDERWORT.
Utricularia inflata Poir. FLOATING BLADDERWORT.
Utricularia juncea Vahl. BLADDERWORT.
Utricularia olivacea Wright ex Griseb. PYGMY BLADDERWORT.
Utricularia purpurea Walt. PURPLE BLADDERWORT.
Utricularia subulata L. ZIGZAG BLADDERWORT.

Table 9-5 (continued)

LINACEAE
Linum medium (Planch.) Britt. var. **texanum** (Planch.) Fern. YELLOW FLAX.
Linum westii C.M.Rogers. WEST'S FLAX.
LOGANIACEAE
Gelsemium rankinii Small. ODORLESS JESSAMINE.
Polypremum procumbens L. RUSTWEED COPPERWEED.
MAGNOLIACEAE
Magnolia grandiflora L. SOUTHERN MAGNOLIA.
Magnolia virginiana L. SWEETBAY.
MELASTOMATACEAE
Rhexia alifanus Walt. ROSE MEADOW BEUATY.
Rhexia cubensis Griseb. WEST INDIAN MEADOW BEAUTY.
Rhexia lutea Walt. YELLOW MEADOW BEAUTY.
Rhexia mariana L. PALE MEADOW BEAUTY.
Rhexia nashii Small. NASH'S MEADOW BEAUTY.
Rhexia parviflora Chapm. WHITE MEADOW BEAUTY.
Rhexia petiolata Walt. CILIATE MEADOW BEAUTY.
Rhexia virginica L. TALL MEADOW BEAUTY.
MYRICACEAE
Myrica heterophylla Raf. BAYBERRY.
Myrica inodora Bartr. ODORLESS WAX-MYRTLE.
NYMPHAEACEAE
Nuphar luteum (L.) Sibth. & Smith ssp. macrophyllum (Small) Beal. COW LILY.
Nymphaea odorata Ait. WHITE WATER LILY.
NYSSACEAE
Nyssa sylvatica Marsh var. **biflora** (Walt.) Sarg. BLACKGUM SWAMP TUPELO.
Nyssa ursina Small. DWARF BLACKGUM.
OLEACEAE
Fraxinus caroliniana Mill. POPASH WATER ASH.
Osmanthus americanus (L.) Gray. WILD OLIVE.
ONAGRACEAE
Ludwigia linearis Walt. NARROW-LEAF SEEDBOX.
Ludwigia linifolia Poir. in Lam. SOUTHEASTERN SEEDBOX
Ludwigia maritima F. Harper. SEASIDE SEEDBOX.
Ludwigia microcarpa Michx. SMALL-FRUIT SEEDBOX.
Ludwigia pilosa Walt. HAIRY SEEDBOX.
Oenothera biennis L. WEEDY EVENING-PRIMROSE.
PLANTAGINACEAE
Plantago virginica L. HOARY PLANTAIN.
POLYGALACEAE
Polygala brevifolia Nutt. LITTLE-LELAF MILKWORT.
Polygala cruciata L. DRUMHEADS.
Polygala cymosa Walt. TALL MILKWORT.
Polygala lutea L. ORANGE MILKWORT BOG BACHELOR'S BUTTON.
Polygala nana (Michx.) DC. WILD BACHELOR'S BUTTON.
POLYGONACEAE
Polygonum punctatum Ell. DOTTED SMARTWEED.
RHAMNACEAE
Berchemia scandens (Hill.) K.Koch. RATTAN VINE.

Table 9-5 (continued)

ROSACEAE
Photinia pyrifolia (Lam.) Robertson & Phipps. RED CHOKEBERRY.
Rubus betulifolius Small. HIGHBUSH BLACKBERRY.
Rubus cuneifolius Pursh. SAND BLACKBERRY.
RUBIACEAE
Diodia teres Walt. POOR JOE ROUGH BUTTONWEED.
Diodia virginiana L. BUTTONWEED.
Hedyotis uniflora (L.) Lam. ONE-FLOWERED SWEET EAR.
SALICACEAE
Salix caroliniana Michx. COASTAL PLAIN WILLOW.
SARRACENIACEAE
Sarracenia flava L. UMPETS YELLOW PITCHER-PLANT.
Sarracenia psittacina Michx. PARROT PITCHER-PLANT.
SAXIFRAGACEAE
Itea virginica L. VIRGINIA WILLOW.
SCROPHULARIACEAE
Agalinis linifolia (Nutt.) Britt. GERARDIA FALSE FOXGLOVE.
Mecardonia acuminata (Walt.) Small. PURPLE MECARDONIA.
Micranthemum umbrosum (Gmel.) Blake. SHADE MUDFLOWER.
STYRACACEAE
Styrax americana var. **pulverulenta** (Michx.) Perkins. SNOWBELL STORAX.
VERBENACEAE
Callicarpa americana L. BEAUTYBUSH FRENCH MULBERRY. TR
Phyla nodiflora (L.) Greene. CAPEWEED FROG-FRUIT.
VIOLACEAE
Viola lanceolata L. BOG-WHITE VIOLET LONGLEAF VIOLET
VITACEAE
Vitis rotundifolia Michx. SCUPPERNONG MUSCADINE.

Table 9-6. Vascular plant species found at reference site in Apalachicola National Forest.

FERNS AND FERN ALLIES
Lycopodium alopecuroides L. FOXTAIL CLUBMOSS
Lycopodium carolinianum L. SLENDER CLUBMOSS
Osmunda regalis L. ROYAL FERN
Pteridium aquilinum (L.) Kuhn. BRACKEN
Woodwardia virginica (L.) J. E. Smith. VIRGINIA CHAIN-FERN

GYMNOSPERMS
PINACEAE
Pinus elliottii Engelm. SLASH PINE
Pinus palustris Mill. LONGLEAF PINE
TAXODIACEAE
Taxodium ascendens Brongn. PONDCYPRESS

ANGIOSPERMS: MONOCOTYLEDONS
AMARYLLIDACEAE
Hymenocallis henryae Traub. GREEN SPIDER LILY
ARECACEAE (Palmae)
Serenoa repens (Bartr.) Small. SAW-PALMETTO
BROMELIACEAE
Tillandsia usneoides (L.) L. SPANISH MOSS
CYPERACEAE
Bulbostylis ciliatifolia (Ell.) Fern. var. **coarctata** (Ell.) Kral. HAIRSEDGE
Carex glaucescens Ell. WAXY SEDGE
Carex lonchocarpa Willd. ex Spreng. [C. folliculata L. misapplied] SOUTHERN LONG-SEDGE
Carex striata Michx. WALTER'S SEDGE
Carex turgescens Torr. SWAMP SEDGE
Fuirena breviseta (Cov.) Cov. in Harper. SALT MARSH UMBRELLA-GRASS
Rhynchospora careyana Fernald. HORNED BEAK-RUSH
Rhynchospora cephalantha Gray var. **pleiocephala** Fern. & Gale. BUNCHED BEAK-SEDGE
Rhynchospora chapmanii Curtis. CHAPMAN'S BEAK-RUSH
Rhynchospora curtissii Britt. ex Small. CURTISS' BEAK-RUSH
Rhynchospora fascicularis (Michx.) Vahl. FASCICLED BEAK-RUSH
Rhynchospora filifolia Gray. THREAD-LEAF BEAK-RUSH
Rhynchospora harperi Small. CYPRESS SWAMP BEAK-RUSH
Rhynchospora inexpansa (Mihcx.) Vahl. NODDING BEAK-RUSH
Rhynchospora latifolia (Baldw.) W.W.Thomas. [Dichromena] WHITE-TOP SEDGE
Rhynchospora oligantha Gray. FEW-FLOWER BEAK-RUSH
Rhynchospora plumosa Ell. PLUMED BEAK-RUSH
Rhynchospora pusilla Chapm. ex M. A. Curtis. FAIRY BEAK-SEDGE
Rhynchospora wrightiana Boeckl. WRIGHT'S BEAK-RUSH
Scirpus cyperinus (L.) Kunth. WOOLGRASS
Scleria baldwinii (Torr.) Steud. BALDWIN'S NUT-RUSH
Scleria ciliata Michx. var. **ciliata**. FRINGED NUT-RUSH
Scleria georgiana Core. GEORGIA NUT-RUSH
Scleria pauciflora Muhl. var. **caroliniana** (Willd.) Wood. FEW-FLOWER NUT-RUSH
Scleria triglomerata Michx. TALL NUT-RUSH

Table 9-6. (continued)

ERIOCAULACEAE
Eriocaulon compressum Lam. HAT-PINS
Eriocaulon decangulare L. COMMON PIPEWORT
Lachnocaulon anceps (Walt.) Morong. BOG-BUTTONS
Syngonanthus flavidulus (Michx.) Ruhl. SHOE-BUTTONS
HAEMODORACEAE
Lachnanthes caroliniana (Lam.) Dundy. REDROOT
Lophiola americana (Pursh) Wood. GOLDCREST
HYPOXIDACEAE
Hypoxis wrightii (Baker) Brackett. YELLOW STARGRASS
JUNCACEAE
Juncus elliottii Coville. ELIOTT'S RUSH
Juncus marginatus Rostk. SHORE RUSH
Juncus repens Michx. CREEPING RUSH
Juncus scirpoides Lam. GLOBE RUSH
LILIACEAE
Aletris lutea Small. YELLOW COLIC-ROOT
Aletris obovata Nash. WHITE COLIC-ROOT
Lilium catesbaei Walt. PINE LILY
Pleea tenuifolia Michx. RUSH-FEATHERING
Tolfieldia racemosa (Walt.) BSP. ASPHODEL
Zigadenus densus (Desr.) Fern. CROW POISON
Zigadenus glaberrimus Michx. SNAKEROOT
ORCHIDACEAE
Calopogon pallidus Chapm. PALE GRASS-PINK
Calopogon tuberosus (L.) BSP. GRASS PINK
Pogonia ophioglossoides (L.) Ker-Gawl. ROSE POGONIA
POACEAE (Gramineae)
Aristida beyrichiana Trin. & Rupr. WIREGRASS
Aristida palustris (Chapm.) Vasey. MARSH THREE-AWN
Aristida purpurescens Poir. var. **tenuispica** (Hitchc.) Allred. ARROW FEATHER THREE-AWN
Axonopus affinis Chase. COMMON CARPETGRASS
Ctenium aromaticum (Walt.) Wood. TOOTHACHE GRASS
Eragrostis atrovirens (Desf.) Trin. ex Steud. THALIA LOVEGRASS
Muhlenbergia capillaris (Lam.) Trin. var. **trichopodes** Vasey. CUT-OVER MUHLY
Panicum acuminatum Swartz var. **acuminatum**. POINTED-TOP PANICGRASS
Panicum erectifolium Nash. MARSH PANICGRASS
Panicum rigidulum Bosc ex Nees var. **pubescens** (Vasey) Lelong. REDTOP PANICUM
Panicum scabriusculum Ell. TALL SWAMP PANICGRASS
Panicum scoparium Lam. BROOM PANICGRASS
Panicum strigosum Muhl. SHORT-BRISTLE PANICGRASS
Panicum tenue Muhl. WHITE-EDGE PANICGRASS
Panicum verrucosum Muhl. WARTY PANICUM
Paspalum notatum Flügge. BAHIA GRASS
Paspalum praecox Walt. EARLY PASPALUM
Tridens ambiguus (Ell.) Schult. PINE BARREN IDENS
SMILACACEAE
Smilax auriculata Walt. GREENBRIER WILD BAMBOO
Smilax laurifolia L. BAMBOO-VINE CATBRIER

Table 9-6. (continued)

XYRIDACEAE
Xyris ambigua Beyr. YELLOW-EYED GRASS
Xyris baldwiniana Schult. ST. MARY'S GRASS
Xyris caroliniana Walt. CAROLINA YELLOW-EYED GRASS
Xyris stricta Chapm. PINELAND YELLOW-EYED GRASS

ANGIOSPERMS: DICOTYLEDONS
ACANTHACEAE
Justicia crassifolia (Chapm.) Small. THICK-LEAF WATER WILLOW
ACERACEAE
Acer rubrum L. ssp. **rubrum.** RED MAPLE
ANNONACEAE
Asimina longifolia Kral var. **spatulata** Kral. LONGLEAF PAWPAW
APIACEAE (Umbellifrae)
Centella erecta (L.f.) Fern. COINWORT SPADELEAF
Eryngium integrifolium Walt. BLUE-FLOWR COYOTE-THISTLE
Eryngium yuccifolium Michx. var. **synchaetum** (Gray) C.& R. RATTLESNAKE MASTER
Oxypolis filiformis (Walt.) Britt. COMMON WATER-DROPWORT
AQUIFOLIACEAE
Ilex cassine L. DAHOON
Ilex coriacea (Pursh) Chapm. LARGE GALLBERRY SWEET GALLBERRY
Ilex glabra (L.) Gray. GALLBERRY INKBERRY
Ilex myrtifolia Walt. MYRTLE-LEAF HOLLY
Ilex vomitoria Ait. YAUPON
ASCLEPIADACEAE
Asclepias cinera Walt. CAROLINA MILKWEED
Asclepias connivens Baldw. ex Ell. LARGE-FLOWER MILKWEED
Asclepias michauxii Decne. MICHAUX'S MILKWEED
Asclepias viridula Chapm. SOUTHERN MILKWEED
ASTERACEAE (Compositae)
Aster chapmanii T. & G. SAVANNAH ASTER
Aster eryngiifolius T. & G. COYOTE-THISTLE ASTER
Balduina uniflora Nutt. HONEYCOMB-HEAD
Bigelowia nudata (Michx.) DC. RAYLESS GOLDENROD
Carphephorus pseudoliatris Cass. BRISTLE-LEAF CHAFFHEAD
Chaptalia tomentosa Vent. SUN-BONNETS PINELAND DAISY
Cirsium lecontei T. & G. LECONTE'S THISTLE
Coreopsis linifolia Nutt. NARROW-LEAVED TICKSEED
Coreopsis nudata Nutt. SWAMP TICKSEED
Helenium pinnatifidum (Nutt.) Rydb. SOUTHEATERN SNEEZEWEED
Helenium vernale Walt. SPRING SNEEZEWEED
Helianthus heterophyllus Nutt. WETLAND SUNFLOWER
Helianthus radula (Pursh) T. & G. RAYLESS SUNFLOWER
Marshallia tenuifolia Raf. BARBARA'S BUTTONS
Pityopsis graminifolia (Michx.) Nutt. var. **graminifolia.** GOLDEN ASTER SILK-GRASS
Pityopsis oligantha (Chapm.) Small. FEW-FLOWERED SILK-GRASS
Rudbeckia graminifolia (T. & G.) Boyn. & Beadle. PURPLE CONEFLOWER
Verbesina chapmanii Coleman. CHAPMAN'S CROWNBEARD
Vernonia angustifolia var. **mohri.** IRONWEED

Table 9-6. (continued)

CAMPANULACEAE
Lobelia brevifolia Nutt. ex DC. SHORT-LEAF LOBELIA
Lobelia floridana Chapm. FLORIDA LOBELIA
Lobelia glandulosa Walt. GLADES LOBELIA
Lobelia paludosa Nutt. PALE LOBELIA
Wahlenbergia marginata (Thunb.) A.DC. ASIATIC BELLFLOWER
CHRYSOBALANACEAE
Licania michauxii Prance. GOPHER APPLE GROUND OAK
CLETHRACEAE
Clethra alnifolia L. SWEET PEPPERBUSH
CLUSIACEAE (Guttiferae)
Hypericum brachyphyllum (Spach) Steud. COASTAL PLAIN ST. JOHN'S WORT
Hypericum chapmanii Adams. CHAPMAN'S ST. JOHN'S WORT
Hypericum cistifolium Lam. ROUND-POD ST. JOHN'S-WORT
Hypericum exile Adams. SLENDER ST. JOHN'S WORT
Hypericum gentianoides (L.) BSP. PINEWEED
Hypericum microsepalum (T. & G.) Gray ex Wats. EARLY ST. JOHN'S-WORT
Hypericum reductum Adams. ATLANTIC ST. JOHN'S WORT
CONVOLVULACEAE
Cuscuta indecora Choisy. LARGE-SEED DODDER
CYRILLACEAE
Cliftonia monophylla (Lam.) Sarg. BLACK TITI BUCKWHEAT EE
Cyrilla racemiflora L. var. **racemiflora.** TITI LEATHERWOOD
Cyrilla racemiflora L. var. **parvifolia** (Raf.) Sarg. SMALL-LEAVED TIT
DROSERACEAE
Drosera capillaris Poir. PINK SUNDEW
Drosera tracyi MacFarlane. DEW-THREADS
ERICACEAE
Gaylussacia dumosa (Anderz.) T. & G. DWARF HUCKELBERRY
Gaylussacia mosieri Small. WOOLLY-BERRY
Leucothoe racemosa (L.) Gray. FETTERBUSH
Lyonia lucida (Lam.) K.Koch. FETTERBUSH
Pieris phillyreifolia (Hook.) DC. VINE-WICKY
Vaccinium darrowi Camp. GLAUCOUS BLUEBERRY
Vaccinium myrsinites Lam. SHINY BLUEBERRY
EUPHORBIACEAE
Cnidoscolus stimulosus (Michx.) Engelm. & Gray. EAD SOFTLY STINGING SPURGE
Euphorbia inundata Torr. ex Chapm. FLORIDA PINE SPURGE
FAGACEAE
Qurecus minima (Sarg.) Small. DWARF LIVE-OAK
Quercus pumila Walt. RUNNING OAK
GENTIANACEAE
Sabatia bartramii Wilbur. BARTRAM MARSH PINK
Sabatia campanulata (L.) Torr. SLENDER ROSE GENTIAN
Sabatia stellaris Pursh. MARSH PINK ROSE GENTIAN
Sabatia macrophylla Hook. LARGE-LEAF ROSE GENTIAN
LAMIACEAE (Labiatae)
Physostegia godfreyi Cantino. OBEDIENT PLANT GODFREY'S DRAGON-HEAD

Table 9-6. (continued)

LAURACEAE
Persea palustris (Raf.) Sarg. SWAMPBAY
LENTIBULARIACEAE
Pinguicula lutea Walt. YELLOW BUTTERWORT
Pinguicula planifolia Chapm. RED-LEAVED BUTTERWORT
Utricularia cornuta Michx. HORNED BLADDERWORT
Utricularia gibba L. SMALL BLADDERWORT
Utricularia purpurea Walt. PURPLE BLADDERWORT
Utricularia subulata L. ZIGZAG BLADDERWORT
LINACEAE
Linum floridanum (Planch) Trel. var. **floridanum.** FLORIDA YELLOW FLAX
Linum medium (Planch.) Britt. var. **texanum** (Planch.) Fern. YELLOW FLAX
LOGANIACEAE
Polypremum procumbens L. RUSTWEED COPPERWEED
MAGNOLIACEAE
Magnolia virginiana L. SWEETBAY
MALVACEAE
Hibiscus aculeatus Walt. PINELAND ROSE MALLOW
MELASTOMATACEAE
Rhexia alifanus Walt. ROSE MEADOW BEAUTY
Rhexia lutea Walt. YELLOW MEADOW BEAUTY
Rhexia mariana L. PALE MEADOW BEAUTY
Rhexia petiolata Walt. CILIATE MEADOW BEAUTY
MYRICACEAE
Myrica heterophylla Raf. BAYBERRY
NYSSACEAE
Nyssa sylvatica Marsh var. **biflora** (Walt.) Sarg. BLACKGUM SWAMP TUPELO
Nyssa ursina Small. DWARF BLACKGUM
OLEACEAE
Fraxinus caroliniana Mill. POPASH WATER ASH
Osmanthus americanus (L.) Gray. WILD OLIVE DEVILWOOD
ONAGRACEAE
Ludwigia linifolia Poir. in Lam. SOUTHEASTERN SEED BOX
Ludwigia microcarpa Michx. SMALL-FRUIT SEED BOX
Ludwigia pilosa Walt. HARIY SEED BOX
Ludwigia virgata Michx. SAVANNAH SEED BOX
POLYGALACEAE
Polygala crenata James. CRENATE MILKWORT
Polygala cruciata L. DRUMHEADS
Polygala cymosa Walt. TALL MILKWORT
Polygala hookeri Torr. & Gray
Polygala lutea L. ORANGE MILKWORT BOG BACHELOR'S BUTTON
Polygala nana (Michx.) DC. WILD BACHELOR'S BUTTON
Polygala ramosa Ell. LOW PINEBARREN MILKWORT
ROSACEAE
Rubus cuneifolius Pursh. SAND BLACKBERRY
RUBIACEAE
Diodia virginiana L. BUTTONWEED
SARRACENIACEAE
Sarracenia flava L. UMPETS YELLOW PITCHER-PLANT
Sarracenia psittacina Michx. PARROT PITCHER-PLANT

Table 9-6. (continued)

SAXIFRAGACEAE
Itea virginica L. VIRGINIA WILLOW
Parnassia caroliniana Michx. GRASS-OF-PARNASSUS
SCROPHULARIACEAE
Agalinis aphylla (Nutt.) Raf. LEAFLESS FALSE-FOXGLOVE
Gratiola pilosa Michx. BRANCHING HEDGE-HYSSOP
Scoparia dulcis L. SWEET BROOM
Seymeria cassioides (Gmel.) Blake BLACK SENNA
STYRACACEAE
Styrax americana var. **pulverulenta** (Michx.) Perkins. SNOWBELL STORAX
VERBENACEAE
Callicarpa americana L. BEAUTYBUSH FRENCH MULBERRY
Phyla nodiflora (L.) Greene CAPEWEED FROG-FRUIT
VIOLACEAE
Viola lanceolata L. BOG-WHITE VIOLET LONGLEAF VIOLET
Viola septemloba LeConte. SOUTHERN COAST VIOLET
VITACEAE
Vitis rotundifolia Michx. SCUPPERNONG MUSCADINE

CONCLUSIONS

The primary objectives of this study were to examine ecosystem response to a hydrologic restoration demonstration project at Big Slough Branch and to establish the baseline ecosystem status of Tates Hell Swamp. Pre- and post-restoration monitoring was conducted at the demonstration site, a non-restored control site, and a relatively unimpacted reference site. Due to administrative, legal, and technical delays, pre-restoration monitoring was limited to a three month period, most of which fell during a severe drought. Post-restoration monitoring continued for fifteen months following completion of restoration. The study has provided a great deal of insight into ecosystem function in Tates Hell, but many questions regarding response to restoration remain unanswered.

Hydrologic monitoring has established that restoration efforts were successful in raising and stabilizing water levels in the demonstration site. Wetland hydroperiods in the demonstration site have been significantly increased over those observed at the topographically similar control site. Median wetland hydroperiod at the demonstration site was 230 days during the first water year following restoration (WY 1998-1999), compared to only 34 days at the control site. Erratic rainfall patterns throughout the course of the study make it difficult to predict long-term hydrology of the demonstration site.

Water quality was excellent at all three sites, both prior to and following restoration. A primary reason for this was the lack of significant silvicultural activities in or adjacent to any of the study sites during the study period. It is believed that restoration will reduce impacts of future silvicultural activities by reducing runoff velocity and providing increased wetland filtration functions. By rehydrating previously drained wetlands, restoration also precludes silvicultural activities in many areas that are particularly vulnerable to the impacts of logging.

Only the most tentative biological responses to restoration were observed. Failure to detect major biological responses can be attributed to several factors. Drought conditions during pre-restoration monitoring and erratic rainfall throughout the study confounded possible comparisons between the pre- and post-restoration conditions, and rendered comparisons among sites somewhat equivocal as well. Erratic rainfall during the course of the study provided valuable insight into the effect of these drought conditions on the Tate's Hell Swamp biota. As noted previously in this report, literature about the effects of drought on swamp biota is sparse. The research performed here will help scientists and land managers better assess the anticipated effects of future droughts.

Benthic macroinvertebrate communities appeared largely unchanged within the timeframe of this study, except for the effects of the summer 1998 drought. Four taxa, the amphipod *Crangonyx*, isopod *Caecidotea*, ceratopogonid dipterans,and chironomid

dipterans made up almost 89% of the benthic macroinvertebrates collected. The densities of all four decreased during the summer of 1998, but rebounded once water refilled the wetlands. The drying of the surface layers of sediment likely played a large role in the decrease of these taxa during this time, as none of these groups is able to emigrate to find standing water.

Use of corers for benthic macroinvertebrate sampling was found to be more effective in this study than sweep net sampling. Core data showed far less variability—especially during dry conditions—than sweep net data. Mean taxon richness values from cores were only slightly lower than those from sweep nets, and densities were much higher. These observations suggest that corers may be generally more appropriate than sweep nets for sampling intermittently-inundated wetlands. It is clear that core sampling should be given careful attention in any future development of a wetland condition index for Florida.

Zooplankton density was largely governed by seasonality, with peak densities generally occurring in the late winter-early spring. Certain taxa (e.g. *Chaoborus* and *Ilyocryptus*) did not regain the abundance late in the study that they had exhibited during peaks early in the study.

Fish abundance was largely a function of water level, with highest abundance collected during periods when wetlands were dry. Ditches serve as refugia from drought conditions in these circumstances. However, predation by piscivores at this time was likely high, so wetlands can also serve as refugia, this time from predation. The dominance shift from pygmy killifish (*Leptolucania ommata*) to mosquitofish (*Gambusia holbrooki*) was one of the more notable observations made during the course of the study. However, whether this is a lasting effect, and the extent to which restoration was responsible for this, are unknown. Young-of-the-year and first-year warmouth (*Lepomis gulosus*)were the only other fish to display a clear response to restoration. The young warmouth were encountered more frequently in the wetlands and low water crossings subsequent to restoration.

Although a large number of plant species—including many endemic and threatened species—are present in the demonstration site, disturbance of native plant communities due to impaired hydrology, fire exclusion, and logging activities is apparent throughout the area. Intensive plant monitoring indicates ongoing encroachment of wetland habitat by upland and weedy species, even in the wettest habitats. Hydrologic restoration in combination with an appropriate fire regime and elimination of destructive logging practices are expected to reverse encroachment and encourage re-establishment of more natural plant communities. While no burning was conducted in the demonstration site or the control site during the study period, the Florida Division of Forestry has an aggressive burning plan for this portion of Tates Hell. Limited burning was conducted near the control site during the final months of the study, and it is anticipated that large portions of the demonstration site will be burned in the next two years.

In order to develop a more complete understanding of ecosystem response to hydrologic restoration in Tates Hell, longer-term study that can account for inter-annual variability will be required. Continued hydrologic monitoring will establish the degree to which restoration has increased long-term wetland hydroperiods, and will clarify the effects of restoration on the timing of fresh water delivery from Tates Hell to East Bay. Future water quality monitoring should be structured to examine the impacts of such management practices as prescribed burning and tree harvesting. Future biological work should focus on benthic macroinvertebrates and fish, as long term wetland utilization by both of these groups is expected to increase as longer hydroperiods become the norm.

Plant community response to restoration is anticipated to occur over a decades-long time frame. The permanent plant quadrats will allow plant response to be studied on a long-term, quantitative basis. Long-term changes in the plant community will undoubtedly have an impact on all levels of the Tates Hell ecosystem. Continued commitment to monitoring—on both an interim and truly long-term basis—can provide valuable guidance for the further restoration of Tates Hell as well as similar wetlands throughout the region.

REFERENCES

Abtew, W., and N. Khanal. 1994. Water budget analysis for the Everglades agricultural area drainage basin. *Water Resources Bulletin* 30 (3):429-439.

Adamus, Paul R. & Karla Brandt. 1990. Impacts on Quality of Inland Wetlands of the United States. EPA/600/3-90/073.

Agostinho, A. A., and M. Zalewski. 1995. The dependence of fish community structure and dynamics on floodplain and riparian ecotone zone in Parana River, Brazil. *Hydrobiologia* 303:141-148.

Allan, J.D. 1976. Life history patterns in zooplankton. *American Naturalist* 110:165-176.

Anderson, D. H., and A. C. Benke. 1994. Growth and reproduction of the cladoceran *Ceriodaphnia dubia* from a forested floodplain swamp. *Limnology and Oceanography* 39 (7):1517-1527.

Anderson, D. H., S. Darring, and A. C. Benke. 1998. Growth of crustacean meiofauna in a forested floodplain swamp: implications for biomass turnover. *Journal of the North American Benthological Society* 17 (1):21-36.

Anderson, K. B., E. F. Benfield, and A. L. Buikema, Jr. 1977. Zooplankton of a swamp water ecosystem. *Hydrobiologia* 55 (2):177-185.

Anderson, L.C. 1987. Noteworthy plants of north Florida. *Boltonia apalachicholensis* (Asteracaeae): a new species from Florida. Systematic Bot. 12:133-138.

Arruda, J.A., G. Marzolf, and R.T. Faulk. 1983. The role of suspended sediments in the nutrition of zooplankton in turbid reservoirs. *Ecology* 64:1225-1235.

Arthrington, A.H. 1987. Diet of *Gambusia affinis holbrooki* (Baird and Girard) in streams of the Brisbane region, southeastern Queensland, Australia. *Verhandlungen. Internationale Vereinigung fur theoretische und angewandte Limnologie* 23:1777.

Arthrington, A.H. 1991. Ecological and genetic impacts of introduced and translocated freshwater fishes in Australia. *Canadian Journal of Fisheries and Aquatic Sciences* 48 (suppl.):33-43.

Bachmann, R. W., B. L. Jones, D. D. Fox, M. Hoyer, L. A. Bull, and D. E. Canfield, Jr. 1996. Relations between trophic state indicators and fish in Florida (U.S.A.) lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 53 (4):842-855.

Baensch, H.A., and R. Riehl. 1997. *Baensch Aquarium Atlas* 3rd ed. 3 vols. Vol. 2. Melle, Germany: MERGUS-Verlag.

Barlocher, F., R.J. MacKay, and G.B. Wiggins. 1978. Detritus processing in a temporary vernal pool in southern Ontario. *Archiv fur Hydrobiologie* 81:269-295.

Barrier, R. F. G., and B. J. Hicks. 1994. Behavioral interactions between black mudfish (*Neochanna diversus* Stokell, 1949: Galaxiidae) and mosquitofish (*Gambusia affinis* Baird & Girard, 1854). *Ecology of Freshwater Fish* 3 (3):93-99.

Bataille, K. J., and G. A. Baldassarre. 1993. Distribution and abundance of aquatic macroinvertebrates following drought in three prairie pothole wetlands. *Wetlands* 13 (4):260-269.

Batzer, D. P., and V. H. Resh. 1991. Trophic interactions among a beetle predator, a chironomid grazer, and periphyton in a seasonal wetland. *Oikos* 60 (2):251-257.

Batzer, D.P., and S.A. Wissinger. 1996. Ecology of insect communities in nontidal wetlands. *Annual Review of Entomology* 41:75-100.

Bays, J.S. 1983. Zooplankton-trophic state relationships and seasonality in Florida lakes. Master's thesis, University of Florida, Gainesville, FL.

Bays, J. S., and T. L. Crisman. 1983. Zooplankton and trophic state relationships in Florida lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 40 (10):1813-1819.

Beaver, J. R., and T. L. Crisman. 1990. Seasonality of planktonic ciliated protozoa in 20 subtropical Florida lakes of varying trophic state. *Hydrobiologia* 190 (2):127-135.

Beaver, J.R., A.M. Miller-Lenke, and J.K. Acton. 1999. Midsummer zooplankton assemblages in four types of wetlands in the Upper Midwest, USA. *Hydrobiologia* 380:209-220.

Beck, L.T. 1977. Distribution and relative abundance of freshwater macroinvertebrates of the lower Atchafalaya River Basin, Louisiana. Master's thesis, Louisiana State University, Baton Rouge, LA.

Beck, R. E. 1994. The movement in the United States to restoration and creation of wetlands. *Natural Resources Journal* 34 (4):781-822.

Belk, M. C., and C. Lydeard. 1994. Effect of *Gambusia holbrooki* on a similar-sized, syntopic poeciliid, *Heterandria formosa*: Competitor or predator? *Copeia* 1994:296-302.

Bellerud, B. L., L. M. Pote, T. L. Lin, M. J. Johnson, and C. R. Boyle. 1995. Etiological and epizootological factors associated with outbreaks of proliferative gill disease in channel catfish. *Journal of Aquatic Animal Health* 7 (2):124-131.

Benfield, M. C., and T. J. Minello. 1996. Relative effects of turbidity and light intensity on reactive distance and feeding of an estuarine fish. *Environmental Biology of Fishes* 46 (2):211-216.

Berg, L., and T. G. Northcote. 1985. Changes in territorial, gill-flaring, and feeding behavior in juvenile coho salmon (*Oncorhynchus kisutch*) following short-term pulses of suspended sediment. *Canadian Journal of Fisheries and Aquatic Sciences* 42 (8):1410-1417.

Bienert, R.W. 1982. The plankton communities of selected colored lakes in north-central Florida. Master's thesis, University of Florida, Gainesville, FL.

Billets, B. D., and J. A. Osborne. 1985. Zooplankton abundance and diversity in Spring Lake, Florida. *Florida Scientist* 48 (3):129-138.

Blancher, E. C. 1984. Zooplankton-trophic state relationships in some north and central Florida lakes. *Hydrobiologia* 109 (3):251-263.

Bohanan, R. E., and D. M. Johnson. 1983. Response of littoral invertebrate populations to a spring fish exclusion experiment. *Freshwater Invertebrate Biology* 2 (1):28-40.

Bozelli, R. L. 1994. Zooplankton community density in relation to water level fluctuations and inorganic turbidity in an Amazonian lake, "Lago Batata," State of Para, Brazil. *Amazoniana* 13:17-32.

Brazner, J. C., and E. W. Beals. 1997. Patterns in fish assemblages from coastal wetland and beach habitats in Green Bay, Lake Michigan: A multivariate analysis of abiotic and biotic forcing factors. *Canadian Journal of Fisheries and Aquatic Sciences* 54 (8):1743-1761.

Breitburg, D. L. 1988. Effects of turbidity on prey consumption by striped bass larvae. *Transactions of the American Fisheries Society* 117 (1):72-77.

Brightman, R.S. 1984. Benthic macroinvertebrate response to secondarily treated wastewater in north-central Florida cypress domes. In *Cypress Swamps*, edited by K. C. Ewel and H. T. Odum. Gainesville, FL: University Presses of Florida.

Brooks, J.L., and S.I. Dodson. 1965. Predation, body size, and composition of plankton. *Science* 150:28-35.

Browder, J.A. 1976. Water, wetlands, and wood storks in South Florida. Ph.D. dissertation, University of Florida, Gainesville, FL.

Brown, S. C., K. Smith, and D. Batzer. 1997. Macroinvertebrate responses to wetland restoration in northern New York. *Environmental Entomology* 26 (5):1016-1024.

Burt, W.H., and R.P. Grossenheider. 1980. *A Field Guide to Mammals*., *Peterson Field Guides*, Edited by R. T. Peterson. Boston, MA: Houghton Mifflin.

Butorina, L.G. 1986. On the problem of aggregation of planktonic crustaceans [*Polyphemus pediculus* (L.), Cladocera]. *Archiv fur Hydrobiologie* 105:355-386.

Byron, E. R., C. L. Folt, and C. R. Goldman. 1984. Copepod and cladoceran success in an oligotrophic lake. *Journal of Plankton Research* 6 (1):45-65.

Campeau, S., H.R. Murkin, and R.D. Titman. 1994. Relative importance of algae and emergent plant litter to freshwater invertebrates. *Canadian Journal of Fisheries and Aquatic Sciences* 51:681-692.

Carlson, J. E., and M. J. Duever. 1977. Seasonal fish population fluctuations in south Florida swamp. *Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies* 31 (1977):603-611.

Chapman, L. J., C. A. Chapman, R. Ogutu-Ohwayo, M. Chandler, L. Kaufman, and A. E. Keiter. 1996. Refugia for endangered fishes from an introduced predator in Lake Nabugabo, Uganda. *Conservation Biology* 10 (2):554-561.

Chaston, I. 1969. Anaerobiosis in *Cyclops varicans*. *Limnology and Oceanography* 14:298-301.

Cline, J. M., T. L. East, and S. T. Threlkeld. 1994. Fish interactions with the sediment-water interface. *Hydrobiologia* 275 (276):301-311.

Clewell, A.F. 1985. Guide to the vascular plants of the Florida Panhandle. Fla. State University Press/University Presses of Florida, Tallahassee.

Coffman, W.P., and L.C. Jr. Ferrington. 1996. Chironomidae. In *An Introduction to the Aquatic Insects of North America*, edited by R. W. Merritt and C. K.W. Dubuque, IA: Kendall/Hunt Publishing Co.

Cohn, J. P. 1994. Restoring the Everglades. *Bioscience* 44 (9):579-583.

Coile, N.C. 1998. Notes on Florida's endangered and threatened plants. Botany Contrib. 38, 2nd ed. Florida Department of Agriculture and Consumer Services, Division of Plant Industry, Gainesville.

Conant, R., and J.T. Collins. 1991. *Reptiles and Amphibians: Eastern/Central North America., Peterson Field Guides*, Edited by R. T. Peterson. Houghton Mifflin: Boston, MA.

Cooley, J.M. 1978. Effect of temperature on development of freshwater copepod eggs. *Crustaceana* 35:27-34.

Copp, G.H. 1989. The habitat diversity and fish reproductive function of floodplain ecosystems. *Environmental Biology of Fishes* 26:1-27.

Corti, D., S. L. Kohler, and R. E. Sparks. 1997. Effects of hydroperiod and predation on a Mississippi River floodplain invertebrate community. *Oecologia* 109 (1):154-165.

Coveney, M. F., D. L. Stites, E. F. Lowe, and L. E. Battoe. 1994. Nutrient removal in the Lake Apopka marsh flow-way demonstration project. *Lake and Reservoir Management* 9 (2):66.

Covich, A.P., and J.H. Thorp. 1991. Crustacea: introduction and Pericardia. In *Ecology and Classification of North American Freshwater Invertebrates*, edited by J. H. Thorp and A. P. Covich. San Diego, CA: Academic Press.

Crome, F. H. J., and S. M. Carpenter. 1988. Plankton community cycling and recovery after drought--dynamics in a basin on a flood plain. *Hydrobiologia* 164 (3):193-211.

Cummins, K. W., and C. N. Dahm. 1995. Introduction to restoring the Kissimmee. *Restoration Ecology* 3 (3):147-148.

Cummins, K.W., and R.W. Merritt. 1996. Ecology and distribution of aquatic insects. In *An Introduction to the Aquatic Insects of North America*, edited by R. W. Merritt and K.W. Cummins. Dubuque, IA: Kendall/Hunt.

Dahl, T.E. 1990. Wetland Losses in the United States, 1790s to 1980s. Washington D.C.: U.S. Department of the Interior, Fish and Wildlife Service.

Dahm, C. N., K. W. Cummins, H. M. Valett, and R. L. Coleman. 1995. An ecosystem view of the restoration of the Kissimmee River. *Restoration Ecology* 3 (3):225-238.

Daigle, J.J. 1991. Florida Damselflies (Zygoptera): A Species Key to the Aquatic Larval Stages. Tallahassee, FL: Florida Department of Environmental Regulation.

Daigle, J.J. 1992. Florida Dragonflies (Anispotera): A Species Key to the Aquatic Larval Stages. Tallahassee, FL: Florida Department of Environmental Regulation.

David, P. G. 1996. Changes in plant communities relative to hydrologic conditions in the Florida Everglades. *Wetlands* 16 (1):15-23.

Davidson, N. L., Jr., W. E. Kelso, and D. A. Rutherford. 1998. Relationships between environmental variables and the abundance of cladocerans and copepods in the Atchafalaya River Basin. *Hydrobiologia* 379:175-181.

DeAngelis, D. L., W. F. Loftus, J. C. Trexler, and R. E. Ulanowicz. 1997. Modeling fish dynamics and effects of stress in a hydrologically pulsed ecosystem. *Journal of Aquatic Ecosystem Stress and Recovery* 6 (1):1-13.

Dierberg, F. E., and P. L. Brezonik. 1984. Water chemistry of a Florida cypress dome. In *Cypress Swamps*, edited by K. C. Ewel and H. T. Odum. Gainesville, FL: University Presses of Florida.

Dodson, S., and D. Frey. 1991. Cladocera and other Branchiopoda. In *Ecology and Classification of North American Freshwater Invertebrates*, edited by J. H. Thorp and A. P. Covich. San Diego, CA: Academic Press.

Doty, M.G. 1997. Relationship between chydorid cladoceran assemblages and macrophyte species composition in a central Florida constructed wetland. Non-thesis research project. University of Florida, Gainesville, FL.

Duffy, W. G., and D. J. LaBar. 1994. Aquatic invertebrate production in southeastern USA wetlands during winter and spring. *Wetlands* 14 (2):88-97.

Dvorak, J. 1987. Production ecological relationships between aquatic vascular plants and invertebrates in shallow waters and wetland--a review. *Ergebnisse der Limnologie* 27:181-184.

Dye, C.W., D.A. Jones, L.T. Ross, and J.L. Gernert. 1980. Del variations of selected physico-chemical parameters in Lake Kissimmee, Florida. *Hydrobiologia* 71:51-60.

Eggers, D.M. 1982. Planktivore preference by prey size. *Ecology* 63:381-390.

Elmore, J. L., B. C. Cowell, and D. S. Vodopich. 1984. Biological communities of three subtropical Florida lakes of different trophic character. *Archiv fur Hydrobiologie* 100:455-478.

Elser, J. J., and N. A. Mackay. 1989. Experimental evaluation of effects of zooplankton biomass and size distribution on algal biomass and productivity in three nutrient-limited lakes. *Archiv fur Hydrobiologie* 114 (4):481-496.

Epler, J.H. 1995. Identification Manual for the Larval Chironomidae (Diptera) of Florida. Tallahassee, FL: Florida Department of Environmental Protection.

Epler, J.H. 1996. Identification Manual for the Water Beetles of Florida. Tallahassee, FL: Florida Department of Environmental Protection.

Erwin, K. L., and G. R. Best. 1985. Marsh community development in a central Florida phosphate surface-mined reclaimed wetland. *Wetlands* 5:155-166.

Evans, D.L. 1996. Aquatic macroinvertebrate communities of constructed and natural marshes in central Florida. Ph.D. dissertation, University of Florida, Gainesville, FL.

Fabre, A. 1988. Experimental studies on some factors influencing phosphorus solubilization in connection with drawdown of a reservoir. *Hydrobiologia* 159:153-158.

Fedorenko, A.Y. 1975. Feeding characteristics and predation impact of *Chaoborus* larvae in a small lake. *Limnology and Oceanography* 20:250-258.

Fernandes, C. C. 1997. Lateral migration of fishes in Amazon floodplains. *Ecology of Freshwater Fish* 6 (1):36-44.

Fernando, C.H., and D Calbraith. 1972. Seasonality and dynamics of aquatic insects colonizing small habitats. *Verhandlungen. Internationale Vereinigung fur theoretische und angewandte Limnologie* 18:1564-1575.

Fletcher, A. R., A. K. Morison, and D. J. Hume. 1985. Effects of carp, *Cyprinus carpio* L., on communities of aquatic vegetation and turbidity of waterbodies in the lower Goulburn River basin. *Australian Journal of Marine and Freshwater Research* 36 (3):311-327.

Foran, J.A. 1983. An analysis of the effect of competition on *Daphnia* species diversity, or why are there so few species of *Daphnia* in the tropics and subtropics. Ph.D. dissertation, University of Florida, Gainesville, FL.

Frey, D.G. 1982. The reticulated species of *Chydorus* (Cladocera, Chydoridae): two new species with suggestions of convergence. *Hydrobiologia* 93:255-279.

Frydenborg, R. 1997. Do forestry best management practices really work? Ambient Newsletter 1: 3 (August 1997): 1-2.

Gaiser, E. E., and K. L. Lang. 1998. Distribution of cladoceran zooplankton among prairie pothole wetlands in northwest Iowa. *Lake and Reservoir Management* 14 (1):37-51.

Giesen, Wbjt, M. M. van Katwijk, and C. den Hartog. 1990. Eelgrass condition and turbidity in the Dutch Wadden Sea. *Aquatic Botany* 37 (1):71-85.

Gladden, J.E., and L.A. Smock. 1990. Macroinvertebrate distribution and production on the floodplains of two lowland headwater streams. *Freshwater Biology* 24:533-545.

Golladay, S. W., B. W. Taylor, and B. J. Palik. 1997. Invertebrate communities of forested limesink wetlands in southwest Georgia, USA: Habitat use and influence of extended inundation. *Wetlands* 17 (3):383-393.

Gons, H.J., R. van Keulen, and R. Veeningen. 1986. Effects of wind on a shallow lake ecosystem. Redistribution of particles in the Loosdrecht lakes. *Hydrobiological Bulletin* 23:109-120.

Gophen, M. 1984. Effect of fish predation on size class distribution of cladocerans in Lake Kinneret. *Verhandlungen. Internationale Vereinigung fur theoretische und angewandte Limnologie* 22:3104-3108.

Gosselink, J.G., and E. Maltby. 1990. Wetland losses and gains. In *Wetlands: A Threatened Landscape*, edited by M. Williams. Oxford, England: Blackwell.

Gottgens, J. F. 1994. Redistribution of organic sediments in a shallow lake following a short-term drawdown. *Archiv fur Hydrobiologie* 130 (2):179-194.

Goulden, C.E. 1971. Environmental control of the abundance and distribution of the chydorid Cladocera. *Limnology and Oceanography* 16:320-331.

Greenberg, A.E., L.S. Clesceri, and A.D. Eaton, eds. 1992. *Standard Methods*- 18th Ed. Washington, DC: American Public Health Association.

Greeson, P.E. 1985. Diel oxygen-curve method for estimating primary productivity and community metabolism in streams. Techniques of Water Resources Investigation of the USGS (B-8120-85). U.S. Geologic Survey, Washington, D.C.

Grimshaw, H. J., R. G. Wetzel, M. Brandenburg, K. Segerblom, L. J. Wenkert, G. A. Marsh, W. Charnetzky, J. E. Haky, and C. Carraher. 1997. Shading of periphyton communities by wetland emergent macrophytes: Decoupling of algal photosynthesis from microbial nutrient retention. *Archiv fur Hydrobiologie* 139 (1):17-27.

Guardo, M., L. Fink, T. D. Fontaine, S. Newman, M. Chimney, R. Bearzotti, and G. Goforth. 1995. Large-scale constructed wetlands for nutrient removal from stormwater runoff: An Everglades restoration project. *Environmental Management* 19 (6):879-889.

Guillory, V. 1978. Life history of warmouth in Lake Conway, Florida. *Proceedings of the Annual Conference of the Southeastern Fish and Wildlife Agencies* 32 (1978):492-501.

Gunderson, L.H., and W.T. Loftus. 1993. The Everglades. In *Biodiversity of the Southeastern United States: Lowland Terrestrial Communities*, edited by W. H. Martin, *et al.* New York: Wiley.

Haack, S.K. 1984. Aquatic macroinvertebrate community structure in a forested wetland: interrelationships with environmental parameters. Master's thesis, University of Florida, Gainesville, FL.

Hall, D.J., S.T Threlkeld, C.W. Burns, and P.H. Crowley. 1976. The size-efficiancy hypothesis and the size structure of zooplankton communities. *Annual Revue of Ecological Systems* 7:177-208.

Hamilton, S.K., and W.M. Jr. Lewis. 1990. Basin morphology in relation to chemical and ecological characteristics of lakes on the Orinoco River floodplain, Venezuala. *Archiv fur Hydrobiologie* 119:393-425.

Hanazato, T., and M. Yasuno. 1989. Zooplankton community structure driven by vertebrate and invertebrate predators. *Oecologia* 81:450-458.

Hann, B.J. 1985. Influence of temperature on life-history characteristics of two sibling species of Eurycercus (Cladocera, Chydoridae). *Canadian Journal of Zoology* 63:891-898.

Hann, B. J., and L. Zrum. 1997. Littoral microcrustaceans (Cladocera, Copepoda) in a prairie coastal wetland: seasonal abundance and community structure. *Hydrobiologia* 357:37-52.

Hanson, M. A., and M. R. Riggs. 1995. Potential effects of fish predation on wetland invertebrates: A comparison of wetlands with and without fathead minnows. *Wetlands* 15 (2):167-175.

Havens, K. E. 1991a. Fish-induced sediment resuspension: Effects on phytoplankton biomass and community structure in a shallow hypereutrophic lake. *Journal of Plankton Research* 13 (6):1163-1176.

Havens, K. E. 1991b. Summer zooplankton dynamics in the limnetic and littoral zones of a humic acid lake. *Hydrobiologia* 215 (1):21-29.

Herzig, A. 1984. Temperature and life cycle strategies of *Diaphanosoma brachyurum*: An experimental study on development, growth, and survival. *Archiv fur Hydrobiologie* 101:143-178.

Hessen, D.O. 1989. Factors determining the nutritive status and production of zooplankton in a humic lake. *Journal of Plankton Research* 11:649-664.

Hey, D. L., and N. S. Philippi. 1995. Flood reduction through wetland restoration: The upper Mississippi River basin as a case history. *Restoration Ecology* 3 (1):4-17.

Hinshaw, J.M. 1985. Effects of illumination and prey contrast on survival and growth of larval yellow perch (*Perca flavescens*). *Canadian Journal of Fisheries and Aquatic Sciences* 114:540-545.

Holdren, G.C. Jr., and D.E Armstrong. 1980. Factors affecting phosphorus release from intact sediment cores. *Environmental Science and Technology* 14:79-86.

Holland, L.E., and M.L. Huston. 1985. Distribution and food habits of Y-O-Y fishes in a backwater lake of the upper Mississippi river. *Journal of Freshwater Ecology* 3:81-91.

Howe, E., C. Howe, R. Lim, and M. Burchett. 1997. Impact of the introduced poeciliid *Gambusia holbrooki* (Girard, 1859) on the growth and reproduction of *Pseudomugil signifer* (Kner, 1865) in Australia. *Marine & Freshwater Research* 48 (5):425-434.

Hoyer, M.V., and D.E. Canfield Jr. 1994. *Handbook of Common Freshwater Fish in Florida lakes*. Gainesville, FL: IFAS, University of Florida.

Hurbert, W.A. 1983. Passive capture techniques. In *Fisheries Techniques*, edited by L. A. Neilsen and D. L. Johnson. Bethesda, MD: American Fisheries Society.

Hurlbert, S.H., and M.S. Mulla. 1981. Impacts of mosquitofish (*Gambusia affinis*) predation on plankton communities. *Hydrobiologia* 83:125-151.

Jamieson, C.D. 1980. Observations on the effect of diet and temperature on rate of development of *Mesocyclops leuckarti* (Claus) (Copepoda:Cyclopoida). *Crustaceana* 38:145-154.

Jeffries, M. 1994. Invertebrate communities and turnover in wetland ponds affected by drought. *Freshwater Biology* 32 (3):603-612.

Jordan, F., K. J. Babbitt, and C. C. McIvor. 1998. Seasonal variation in habitat use by marsh fishes. *Ecology of Freshwater Fish* 7 (4):159-166.

Junk, W.J., P.B. Bayley, and R.E. Sparks. 1989. The flood pulse concept in river floodplain systems. *Canadian Special Publications of Fisheries and Aquatic Sciences* 106:110-127.

Kadlec, R.H., and R.L. Knight. 1995. Treatment Wetlands. Lewis Publishers.

Kahl, M.P. Jr. 1964. Food ecology of the wood stork (*Myceteria americana*) in Florida. *Ecological Monographs* 34:97-117.

Kajak, Z., and J. Ryback. 1979. The feeding of *Chaoborus flavicans* Meijen (Diptera, Chaoboridae) and its predation on lake zooplankton. *Internationale Revue der gesamtem Hydrobiologie* 64:361-378.

Kalk, M. J., and C. M. Schulten-Senden. 1977. Zooplankton in a tropical endorheic lake (Lake Chilwa, Malawi) during drying and recovery phases. *Journal of the Limnological Society of South Africa* 3 (1):1-7.

Keddy, P.A. 1983. Freshwater wetland human-induced changes: indirect effects must also be considered. *Environmental Management* 7:299-302.

Keenan, C.M. 1995. A preliminary evaluation of zooplankton in constructed wetlands: sample size determination and spatial patterns. Master's thesis, University of Florida, Gainesville, FL.

Killgore, K. J., and J. A. Baker. 1996. Patterns of larval fish abundance in a bottomland hardwood wetland. *Wetlands* 16 (3):288-295.

Kindell,C. 1997. Historic Distribution of Wet Savannas in Tates Hell State Forest. Final Report for the U.S. Fish and Wildlife Service (Agreement #1448-0004-96-9102) and Northwest Florida Water Management District. Florida Natural Areas Inventory, Tallahassee, FL.

Kneib, R. T., and A. E. Stiven. 1982. Benthic invertebrate responses to size and density manipulations of the common mummichog, *Fundulus heteroclitus*, in an intertidal salt marsh. *Ecology* 63 (5):1518-1532.

Knight, J. G., and M. B. Bain. 1996. Sampling fish assemblages in forested floodplain wetlands. *Ecology of Freshwater Fish* 5 (2):76-85.

Koch, M.S., and K.R. Reddy. 1992. Distribution of soil and plant nutrients along a trophic gradient in the Florida Everglades. *Soil Science Society of America Journal* 56:1492-1499.

Kolasa, K. V. 1994. Zooplankton -- trophic state relationships in several central Florida lakes. *Lake and Reservoir Management* 9 (2):87.

Kramer, D.L., and J.P. Mehegan. 1981. Aquatic surface respiration, an adaptive response to hypoxia in the guppy, *Poecilia reticulata* (Pisces, Poecilidae). *Environmental Biology of Fishes* 6 (3-4):299-313.

Kushlan, J. A., and B. P. Hunt. 1979. Limnology of an alligator pond in south Florida. *Florida Scientist* 42 (2):65-84.

Kwak, T.J. 1988. Lateral movement and use of floodplain habitat by fishes of the Kankakee River, Illinois. *American Midland Naturalist* 102:241-249.

Lalancette, L. M. 1984. The effects of dredging on sediments, plankton and fish in the Vauvert area of Lake St. Jean., Quebec. *Archiv fur Hydrobiologie* 99 (4):463-477.

Lee, D.S., C.R. Gilbert, C.H. Hocutt, R.E. Jenkins, D.E. McAllister, and J. Stauffer. 1981. *Atlas of North American Freshwater Fish*: Publication # 1980-12 of the North Carolina Biological Survey, Raleigh, NC.

Leeper, D. A., and B. E. Taylor. 1998a. Abundance, biomass and production of aquatic invertebrates in Rainbow Bay, a temporary wetland in South Carolina, USA. *Archiv fur Hydrobiologie* 143 (3):335-362.

Leeper, D. A., and B. E. Taylor. 1998b. Insect emergence from a South Carolina (USA) temporary wetland pond, with emphasis on the Chironomidae (Diptera). *Journal of the North American Benthological Society* 17

Leslie, A. J., T. L. Crisman, J. P. Prenger, and K. C. Ewel. 1997. Benthic macroinvertebrates of small Florida pondcypress swamps and the influence of dry periods. *Wetlands* 17 (4):447-455.

Leslie, A.J., J.P. Prenger, and T.C. Crisman. 1999. Cypress domes in North Florida: Invertebrate ecology and response to human disturbance. In *Invertebrates in Freshwater Wetlands of North America: Ecology and Management*, edited by D. P. Batzer, *et al.* New York: Wiley.

Lick, W. 1982. Entrainment, deposition, and transport of fine-grained sediments in lakes. *Hydrobiologia* 91:31-40.

Livingston, R.J. 1983. Resource Atlas of the Apalachicola Estuary. Florida Sea Grant Rep. No. 55.

Lowe, E. F., L. E. Battoe, D. L. Stites, and M. F. Coveney. 1992. Particulate phosphorus removal via wetland filtration: An examination of potential for hypertrophic lake restoration. *Environmental Management* 16 (1):67-74.

Lythgoe, J.N. 1979. *The Ecology of Vision*. Oxford: Oxford University Press.

Marburger, J. E. 1994. Restoration of the Emeralda Marsh Conservation Area. *Lake and Reservoir Management* 9 (2):95.

McCabe, G.D., and W.J. O'Brien. 1983. The effects of suspended silt on feeding and reproduction of *Daphnia pulex*. *American Midland Naturalist* 11 (324-337).

McClachlan, A.J., and A.M. Cantrell. 1980. Survival strategies in tropical rainpools. *Oecologia* 47:344-351.

McCormick, P. V., R. B. E. Shuford, III, J. G. Backus, and W. C. Kennedy. 1998. Spatial and seasonal patterns of periphyton biomass and productivity in the northern Everglades, Florida, U.S.A. *Hydrobiologia* 362:185-208.

McCoy, E. D., H. R. Mushinsky, D. Johnson, and W. E. Meshaka, Jr. 1996. Mangrove damage caused by Hurricane Andrew on the southwestern coast of Florida. *Bulletin of Marine Science* 59 (1):1-8.

Meffe, G. K., D. A. Hendrickson, W. L. Minckley, and J. N. Rinne. 1983. Factors resulting in decline of the endangered Sonoran topminnow *Poeciliopsis occidentalis* (Atheriniformes: Poeciliidae) in the United States. *Biological Conservation* 25 (2):135-159.

Meijer, M. L., M. W. De Haan, A. W. Breukelaar, and H. Buiteveld. 1990. Is reduction of the benthivorous fish an important cause of high transparency following biomanipulation in shallow lakes? *Hydrobiologia* 200 (201):303-315.

Meijer, M.L., A.J.P. Raat, and R.W. Doef. 1989. Restoration by biomanipulation of the Dutch shallow eutrophic lake Bleiswijkse Zoom; first results. *Hidrobiological Bulletin* 23:49-59.

Merritt, R.W., and K.W. Cummins, eds. 1996. *An Introduction to the Aquatic Insects of North America* 3rd ed. Dubuque, IA: Kendall/Hunt.

Meyer, R.L. and J. J. Ewel. 1990. Ecosystems of Florida. University of Central Florida Press/University Presses of Florida, Orlando.

Miner, J. G., and R. A. Stein. 1993. Interactive influence of turbidity and light on larval bluegill (*Lepomis macrochirus*) foraging. *Canadian Journal of Fisheries and Aquatic Sciences* 50 (4):781-788.

Mitsch, W.J., and J.G. Gosselink. 1993. *Wetlands* 2nd ed. New York: Van Nostrand Reinhold.

Mitsch, W. J., A. Van der Valk, and E. Jaworski. 1994. Wetland restoration at a former Nike missile base in the Great Lakes Basin. *Restoration Ecology* 2 (1):31-42.

Moore, M.V., and J.J. Gilbert. 1987. Age-specific *Chaoborus* predation on rotifer prey. *Freshwater Biology* 17:223-236.

Moorhead, D. L., D. L. Hall, and M. R. Willig. 1998. Succession of macroinvertebrates in playas of the Southern High Plains, USA. *Journal of the North American Benthological Society* 17 (4):430-442.

Muck, P., and W. Lampert. 1984. An experimental study ofn the importance of food conditions for the relative abundance of calanoid copepods and cladocerans. *Archiv fur Hydrobiologie (Suppl.)* 66:157-179.

Muller, J.W., E. D.Hardin, D. R. Jackson, S. E. Gatewood, and N. Caire. 1989. Summary report of the vascular plants, animals and plant communities endemic to Florida. Florida Game and Fresh Water Fish Comm. Nongame Wildl. Program Tech. Rep. no. 7.

Muntz, W.R.A. 1982. Visual adaptations to different light environments in Amazonian fishes. *Revue Canadienne de Biologie Experimentale* 41:35-36.

Murkin, E. J., H. R. Murkin, and R. D. Titman. 1992. Nektonic invertebrate abundance and distribution at the emergent vegetation-open water interface in the Delta Marsh, Manitoba, Canada. *Wetlands* 12 (1):45-52.

Murkin, H. R., and B. D. J. Batt. 1987. The interactions of vertebrates and invertebrates in peatlands and marshes. *Memoirs of the Entomological Society of Canada* 140:15-30.

Murkin, H. R., and J. A. Kadlec. 1986. Responses by benthic macroinvertebrates to prolonged flooding of marsh habitat. *Canadian Journal of Zoology* 64 (1):67-72.

Neill, W. E. 1981. Impact of *Chaoborus* predation upon the structure and dynamics of a crustacean zooplankton community. *Oecologia* 48 (2):164-177.

Nordlie, F, G. 1976. Plankton communities of three central florida lakes. *Hydrobiologia* 48:65-78.

Northwest Florida Water Management District. 1999. Comparative Ecosystem Analysis of Hydrologic Restoration of Tates Hell Swamp. DEP Contract WM 691.

Odum, Howard T. 1956. Primary Production in Flowing Waters. Limnol. Oceanogr. 1:102-117.

Odum, H. T., G. R. Best, M. A. Miller, B. T. Rushton, and R. Wolfe. 1990. Accelerating natural processes for wetland restoration after phosphate mining: Publications of the Florida Institute of Phosphate Research.

Ogden, J.C., J.A. Kushlan, and J.T. Tilmant. 1976. Prey selectivity by the wood stork. *Condor* 78:324-330.

Olila, O. G., K. R. Reddy, and D. L. Stites. 1997. Influence of draining on soil phosphorus forms and distribution in a constructed wetland. *Ecological Engineering* 9:157-169.

Orcutt, J. D., Jr. 1985. Food level effects on the competitive interactions of two co-occurring cladoceran zooplankton: *Diaphanosoma brachyurum* and *Daphnia ambigua*. *Ergebnisse der Limnologie* 21:465-473.

Orth, D.J., and D.J. White. 1993. Stream habitat management. In *Inland Fisheries Manegement in North America*, edited by C. C. Kohler and W. A. Hubert. Bethesda, MD: American Fisheries Society.

Osborne, J. A., and C. Jansen. 1993. The zooplankton community in an acidic central Florida lake. *Journal of Freshwater Ecology* 8 (1):47-56.

Page, L.M., and B.M. Burr. 1991. *A Field Guide to Freshwater Fishes. Peterson Field Guides,* Edited by R. T. Peterson. Boston, MA: Houghton Mifflin.

Paller, M. H. 1987. Distribution of larval fish between macrophyte beds and open channels in a southeastern floodplain swamp. *Journal of Freshwater Ecology* 4 (2):191-200.

Parker, A. and T. Rasmussen. 1997. Hydraulic Restoration of Tates Hell Swamp: A comparative watershed analysis of ecosystem response and recovery, results from prerestoration analysis, Final Report. Daniel B. Warnell School of Forest Resources, University of Georgia, Athens, GA.

Patric, R., M.H. Hohn, and J.H. Wallace. 1954. A new method of determining the pattern of the diatom flora. *Bulletin of the Philadelphia Academy of Natural Sciences* 259:1.

Patrick, W.H. Jr., and R.A. Khalid. 1976. Phosphate release and sorption by soils and sediments: effect of aerobic and anaerobic conditions. *Science* 186:53-55.

Pen, L. J., and I. C. Potter. 1991. Reproduction, growth and diet of *Gambusia holbrooki* (Girard) in a temperate Australian river. *Aquatic Conservation: Marine and Freshwater Ecosystems* 1 (2):159-172.

Pennak, R.W. 1989. *Fresh-water Invertebrates of the United States : Protozoa to Mollusca* 3rd ed. New York: Wiley.

Pennak (1989), Daigle (1991), Thorp (1991), Daigle (1992), Epler (1995), Pescador (1995), Epler (1996, and Merritt (1996).

Pescador, M.L., A.K. Rasmussen, and S.C. Harris. 1995. Identification Manual for the Caddisfly (Trichoptera) Larvae of Florida. Tallahassee, FL: Florida Department of Environmental Protection.

Pezeshki, C. 1987. Response of benthic macroinvertebrates of a shrub swamp to discharge of treated wastewater. Master's thesis, University of Florida, Gainesville, FL.

Pezold, F. 1998. Fish diversity in an isolated artificial wetland. *Journal of Freshwater Ecology* 13 (2):171-179.

Phillips, G., R. Jackson, C. Bennett, and A. Chilvers. 1994. The importance of sediment phosphorus release in the restoration of very shallow lakes (The Norfolk Boards, England) and the implications for biomanipulation. *Hydrobiologia* 275-276:445-456.

Pollard, J. E., S. M. Melancon, and L. S. Blakey. 1983. Importance of bottomland hardwoods to crawfish and fish in the Henderson Lake area, Atchafalaya Basin, Louisiana. *Wetlands* 3:1-25.

Porter, K.G., A. Bergstedt, and M.C. Freeman. 1999. The Okefenokee Swamp: Invertebrate communities and food webs. In *Invertebrates in Freshwater Wetlands of North America: Ecology and Management*, edited by D. P. Batzer *et al.* New York: Wiley.

Prenger, J.P., A.J. Leslie, T.L. Crisman, and K.C. Ewel. In prep. Impacts of clearcutting on benthic macroinvertebrate communities of small Florida pondcypress swamps. *(Manuscript in preparation)*.

Pyke, G.H., H.R. Pulliam, and E.L. Charnov. 1977. Optimal foraging: a selective review of theory and tests. *Quarterly Revue of Biology* 52:137-154.

Qiu, Song, and A. J. McComb. 1994. Effects of oxygen concentration on phosphorus release from reflooded air-dried wetland sediments. *Australian Journal of Marine and Freshwater Research* 45 (7):1319-1328.

Quade, H.W. 1969. Cladoceran faunas associated with aquatic macrophytes in some lakes in northern Minnesota. *Ecology* 50 (170-179).

Rader, R. B. 1994. Macroinvertebrates of the northern Everglades: Species composition and trophic structure. *Florida Scientist* 57:22-33.

Rahel, F. J., and J. W. Nutzman. 1994. Foraging in a lethal environment: Fish predation in hypoxic waters of a stratified lake. *Ecology* 75 (5):1246-1253.

Reimer, G. 1991. The ecological importance of floodplains for fish at the River March (Austria). *Archiv fur Hydrobiologie* 121 (3):355-363.

Richardson, M. J., and F. G. Whoriskey. 1992. Factors influencing the production of turbidity by goldfish (*Carassius auratus*). *Canadian Journal of Zoology* 70 (8):1585-1589.

Riehl, R., and H.A. Baensch. 1997. *Baensch Aquarium Atlas* 6th ed. 3 vols. Vol. 1. Melle, Germany: MERGUS-Verlag.

Rivier, I.K. 1998. *The Predatory Cladocera (Onychopoda: Podonidae, Polyphemidae, Cercopagidae) and Leptadorida of the World.* Leiden, The Netherlands: Backhuys

Robinson, G. G. C., S. E. Gurney, and L. G. Goldsborough. 1997. The primary productivity of benthic and planktonic algae in a prairie wetland under controlled water-level regimes. *Wetlands* 17 (2):182-194.

Rodriguez, M. A., and W. M. Lewis, Jr. 1997. Structure of fish assemblages along environmental gradients in floodplain lakes of the Orinoco River. *Ecological Monographs* 67 (1):109-128.

Ryan, P. A. 1991. Environmental effects of sediment on New Zealand streams: A review. *New Zealand Journal of Marine and Freshwater Research* 25 (2):207-221.

Rzoka, J. 1961. Observations on tropical rainpools and general remarks on temporary waters. *Hydrobiologia* 17:265-286.

SAS Institute. 1995. JMP Statistics and Graphics Guide, Version 3.1. SAS Institute, Cary, N.C.

Sasser, L.D., K.L. Moore and N. Schuster. 1994. Soil Survey of Franklin County, U.S. Department of Agriculture, Soil Conservation Service, Washington, D.C.

Schaefer, J. F., S. T. Heulett, and T. M. Farrell. 1994. Interactions between two poeciliid fishes (*Gambusia holbrooki* and *Heterandria formosa*) and their prey in a Florida marsh. *Copeia* 1994:516-520.

Schoenberg, S. A. 1988. Microcrustacean community structure and biomass in marsh and lake habitats of the Okefenokee Swamp: Seasonal dynamics and responses to resource manipulations. *Holarctic Ecology* 11 (1):8-18.

Schoenberg, S. A., and J. D. Oliver. 1988. Temporal dynamics and spatial variation of algae in relation to hydrology and sediment characteristics in the Okefenokee Swamp, Georgia. *Hydrobiologia* 162 (2):123-133.

Schoenherr, A.A. 1981. The role of competition in the displacement of native fishes by introduced species. In *Fishes in North American Deserts*, edited by R. J. Naiman and D. L. Stolz. New York: Wiley Interscience.

Shormann, D. E., and J. B. Cotner. 1997. The effects of benthivorous smallmouth buffalo (*Ictiobus bubalus*) on water quality and nutrient cycling in a shallow floodplain lake. *Lake and Reservoir Management* 13 (3):270-278.

Sklar, F. H. 1985. Seasonality and community structure of the backswamp invertebrates in a Louisiana cypress-tupelo wetland. *Wetlands* 5:69-86.

Sklar, F. H., and J. W. Day, Jr. 1985. Backswamp invertebrates: A rare glimpse of the aquatic fauna in a Louisiana cypress-tupelo wetland. *Estuaries* 8 (2B):97A.

Smirnov, N. N. 1974. *Fauna of the U.S.S.R.: Crustacea : Chydordae*. Jerusalem: Keter Press.

Smith, I.M., and D.R. Cook. 1991. Water Mites. In *Ecology and Classification of North American Freshwater Invertebrates*, edited by J. H. Thorp and A. P. Covich. San Diego, CA: Academic Press.

Smith, W.F. 1992. Response of mosquitofish (*Gambusia affinis*) and least killifish (*Heterandria formosa*) to water quality and vegetation associated with wastewater addition to a forested wetland in Central Florida. Master's thesis, University of Florida, Gainesville, FL.

Smock, L.A., and D.L. Stoneburner. 1980. The response of macroinvertebrates to aquatic macrophyte decomposition. *Oikos* 33:397-403.

Soto, D., and S. H. Hurlbert. 1991. Long-term experiments on calanoid-cyclopoid interactions. *Ecological Monographs* 61 (3):245-265.

Starrett, W.C. 1951. Some factors affecting the abundance of minnows in the Des Moines River, Iowa. *Ecology* 32:13-27.

Stenson, J. A. E. 1978. Differential predation by fish on two species of *Chaoborus* (Diptera, Chaoboridae). *Oikos* 31 (1):98-101.

Stenson, J.A.E., and H.G. Oscarson. 1985. Crustacean zooplankton in the acidified Lake Gardsjon system. *Ecology Bulletin* 37:224-131.

Stoneburner, D.L., and L.A. Smock. 1980. Plankton communities of an acidic, polymictic brownwater lake. *Hydrobiologia* 69:131-138.

Streever, W. J., and T. L. Crisman. 1993. A comparison of fish populations from natural and constructed freshwater marshes in central Florida. *Journal of Freshwater Ecology* 8 (2):149-153.

Swiadek, J. W. 1997. The impacts of Hurricane Andrew on mangrove coasts in southern Florida: A review. *Journal of Coastal Research* 13 (1):242-245.

Taylor, B.E., D.A. Leeper, M.A. McClure, and A.E. DeBiase. 1999. Carolina bays: ecology of aquatic invertebrates and perspectives on conservation. In *Invertebrates in Freshwater Wetlands of North America: Ecology and Management*, edited by D. P. Batzer *et al.* New York: Wiley.

Taylor, B. E., and D. L. Mahoney. 1990. Zooplankton in Rainbow Bay, a Carolina bay pond: Population dynamics in a temporary habitat. *Freshwater Biology* 24 (3):597-612.

Tejerina-Garro, F. L., R. Fortin, and M. A. Rodriguez. 1998. Fish community structure in relation to environmental variation in floodplain lakes of the Araguaia River, Amazon Basin. *Environmental Biology of Fishes* 51 (4):399-410.

Tessier, A.J. 1983. Coherence and horizontal movements of a patch of *Holopedium gibberum* (Cladocera). *Oecologia* 60:71-75.

Thibodeau, F. R., and N. H. Nickerson. 1985. Changes in a wetland plant association induced by impoundment and draining. *Biological Conservation* 33 (2):269-280.

Thorp, J.H. , and A.P. Covich, eds. 1991. *Ecology and Classification of North American Freshwater Invertebrates*. San Diego, CA: Academic Press.

Thorp, J. H., E. M. McEwan, M. F. Flynn, and F. R. Hauer. 1985. Invertebrate colonization of submerged wood in a cypress-tupelo swamp and blackwater stream. *American Midland Naturalist* 113 (1):56-68.

Threlkeld, S.T. 1986. Life table responses and population dynamics of four cladoceran zooplankton during a reservoir flood. *Journal of Plankton Research* 8:639-647.

Tibbs, J. E., and D. L. Galat. 1998. The influence of river stage on endangered least terns and their fish prey in the Mississippi River (USA). *Regulated Rivers: Research & Management* 14 (3):257-266.

Tilmant, J. T., R. W. Curry, R. Jones, A. Szmant, J. C. Zieman, M. Flora, M. B. Robblee, D. Smith, R. W. Snow, and H. Wanless. 1994. Hurricane Andrew's effects on marine resources. The small underwater impact contrasts sharply with the destruction in mangrove and upland-forest communities. *Bioscience* 44 (4):230-237.

Timms, R. M., and B. Moss. 1984. Prevention of growth of potentially dense phytoplankton populations by zooplankton grazing, in the presence of zooplanktivorous fish, in a shallow wetland ecosystem. *Limnology and Oceanography* 29 (3):472-486.

Toth, L. A. 1993. The ecological basis of the Kissimmee River restoration plan. *Florida Scientist* 56 (1):25-51.

Turner, T. F., J. C. Trexler, G. L. Miller, and K. E. Toyer. 1994. Temporal and spatial dynamics of larval and juvenile fish abundance in a temperate floodplain river. *Copeia* 1994:174-183.

Ulanowicz, R. E. 1995. *Utricularia*'s secret: The advantage of positive feedback in oligotrophic environments. *Ecological Modelling* 79:49-57.

Vanni, M. J. 1986. Fish predation and zooplankton demography: Indirect effects. *Ecology* 67 (2):337-354.

Vanni, M. J. 1987. Effects of nutrients and zooplankton size on the structure of a phytoplankton community. *Ecology* 68 (3):624-635.

Vijverberg, J. 1980. Effect of temperature in laboratory studies on development and growth on Cladocera and Copepoda from Tjeukemeer, the Netherlands. *Freshwater Biology* 10:317-340.

Vinyard, G.L., and R.A. Menger. 1980. *Chaoborus americanus* predation on various zooplankters: functional response and behavioral observations. *Oecologia* 45:90-93.

Vinyard, G.L., and W.J. O'Brien. 1976. Effects of light and turbidity on the reactive distance of bluegill (*Lepomis macrochirus*). *Journal of the Fisheries Research Board of Canada* 33:2845-2849.

von Ende, C. N. 1979. Fish predation, interspecific predation, and the distribution of two *Chaoborus* species. *Ecology* 60 (1):119-128.

Vymazal, J., and C. J. Richardson. 1995. Species composition, biomass, and nutrient content of periphyton in the Florida Everglades. *Journal of Phycology* 31 (3):343-354.

Walls, M., M. Rajasilta, J. Sarvala, and J. Salo. 1990. Diel changes in horizontal microdistribution of littoral Cladocera. *Limnologica* 20 (2):253-258.

Wegener, W., D. Holcomb, and V. Williams. 1973. Sampling shallow water fish populations using the Wegener ring. *Proceedings of the Annual Conference of Southeastern Game and Freshwater Fish Commissions* 27:663-673.

Welcomme, R.L. 1979. *Fisheries Ecology of Floodplain Rivers*. London: Longman.

Weller, J. D. 1995. Restoration of a south Florida forested wetland. *Ecological Engineering* 4 (2):143-151.

Westfall, M.J. Jr. 1996. Odonata. In *An Introduction to the Aquatic Insects of North America*, edited by R. W. Merritt and K.W.Cummins. Dubuque, IA: Kendall/Hunt.

Wetzel, R.G. 1983. *Limnology*. 2nd ed. Orlando, FL: Saunders.

Whitaker, G., and C. R. Terrell. 1992. Federal programs for wetland restoration and use of wetlands for nonpoint source pollution control. *Ecological Engineering* 1:157-170.

White, D. C. 1985. Lowland hardwood wetland invertebrate community and production in Missouri. *Archiv fur Hydrobiologie* 103 (4):509-533.

White, D.S., and W.U. Brigham. 1996. Aquatic Coleoptera. In *An Introduction to the Aquatic Insects of North America*, edited by R. W. Merritt and K.W. Cummins. Dubuque, IA: Kendall/Hunt.

Whiteside, M.C. 1988. 0+ Fish as major factors affecting abundance patterns of littoral zooplankton. *Verhandlungen. Internationale Vereinigung fur theoretische und angewandte Limnologie* 23:1710-1714.

Whiteside, M.C., W.L. Doolittle, and C.M. Swindoll. 1985. Factors affecting the early life-history of yellow perch, *Perca flavescens*. *Environmental Biology of Fishes* 12:47-56.

Whiteside, M.C., J.B. Williams, and C.P. White. 1978. Seasonal abundance and pattern of chydorid Cladocera in mud and vegetative habitats. *Ecology* 59:1177-1188.

Wiggins, G.B., R.J. Mackay, and I.M. Smith. 1980. Evolutionary strategies of animals in temporary pools. *Archiv fur Hydrobiologie Supplement* (58).

Williams, D.D. 1987. *The Ecology of Temporary Waters*. London: Croom Helm.

Williams, D. D. 1997. Temporary ponds and their invertebrate communities. *Aquatic Conservation: Marine and Freshwater Ecosystems* 7 (2):105-117.

Williams, W. D. 1985. Biotic adaptations in temporary lentic waters, with special reference to those in semi-arid and arid regions. *Hydrobiologia* 125:85-110.

Williamson, C. 1991. Copepoda. In *Ecology and Classification of North American Freshwater Invertebrates*, edited by J. H. Thorp and A. P. Covich. San Diego, CA: Academic Press.

Wolf, S.H., J. A. Reidenauer, and D.B. Means. 1988. An ecological characterization of the Florida Panhandle. Biological Report 88(12). New Orleans, LA: U.S. Department of the Interior, Fish and Wildlife Service.

Wyngaard, G.A., J.L. Elmore, and B.C. Cowell. 1982. Dynamics of a subtropical zooplankton community, with emphasis on the copepod *Mesocyclops edax*. *Hydrobiologia* 89:39-48.

Wyngaard, G.A., B.E. Taylor, and D.L. Mahoney. 1991. Emergence and dynamics of cyclopoid copepods in an unpredictable environment. *Freshwater Biology* 25:219-232.

Zentner, J. 1985. Wetland restoration in coastal California: A decade of management lessons. *Estuaries* 8 (2B):30A.

Zettler, C.D., and J.C.H. Carter. 1986. Zooplankton community and species responses to a natural turbidity gradient in Lake Temiskamig, Ontario-Quebec. *Canadian Journal of Fisheries and Aquatic Sciences* 43:665-673.