Technical Publication SJ 84-15

U.S. EPA Clean Lakes Program, Phase I Diagnostic-Feasibility Study of the Upper St. Johns River Chain of Lakes
Volume 1 - Diagnostic Study

by

E. F. Lowe, J. E. Brooks, C. J. Fall,
L. R. Gerry, and G. B. Hall
Environmental Specialists

Department of Water Resources
St. Johns River Water Management District
P. O. Box 1429
Palatka, Florida

November 1984

ACKNOWLEDGMENTS

We thank Don Mock, for unflagging assistance in the field; David Girardin, for work on the chronology of basin development; Marvin Williams, for assistance in the analysis of aerial photographs; Rick Keller and Joel Steward, for laboratory analysis of water samples; Bruce Ford, for assistance with graphics; and Sandra Cason and Dolores Messer, for typing the first draft. Special thanks are owed Mike Binford and Edward Deevey who kindly consented to analyze the sediments of Lake Hell 'n Blazes. In addition, we are grateful for the editorial efforts of Vernon Meyers, Ron Raschke, and Jim Hulbert.

The sedimentation and agricultural pumpage portions of this study benefitted from data collected by Tom Belanger, Scott van Vonderen, and Tom Carberry under a contract between the St. Johns River Water Management District and Florida Institute of Technology.

TABLE OF CONTENTS

LIST OF FIGURES

List of Tables

ABSTRACT

The Upper St. Johns River and its chain of lakes, located in east-central Florida, lie within a large floodplain marsh which has been extensively modified for agricultural development of its rich histosols. Approximately 65 percent of the floodplain marsh has been drained and altered for the production of cattle, citrus, and row crops. The literature indicates that concomitant with agricultural development there were significant declines in river and lake populations of game fish, wading birds and waterfowl (coots and ducks). In addition, the literature contains poorly substantiated claims that river and lake water quality has declined due to the poor quality water pumped off floodplain farms and ranches and due to channelization of the marsh, that the sedimentation rate in Lake Hell 'n Blazes has increased due to chemical control of water hyacinth, and that the vegetation of the floodplain marsh has changed due to a decline in water levels.

We found that the water quality of pumped agricultural discharges and of agricultural canals is poorer than that of more natural tributaries. During the wet season, pumped discharges had higher mean concentrations of turbidity (7.9 NTU), conductivity (820 umhos/cm), color (307 cpu), alkalinity (135 mg/l), orthophosphate (0.16 mg P/l), total phosphorus (0.26 mg P/l), nitrate-nitrite nitrogen (0.31 mg N/l) and TKN (2.5 mg/l) than did tributaries and an undisturbed marsh (Jane Green Marsh) while canals had the highest mean concentrations of conductivity (1193 umhos/cm), dissolved solids (700 mg/l), chloride (238 mg/l), and sulfate (68 mg/l). These data and trend analysis of chloride levels in Lake Washington from 1959 - 1982 indicate that agricultural

discharges and canals, coupled with a decline in water levels, have caused a progressive increase in the levels of total dissolved solids. During droughts this increased load of salts can result in exceedance of state standards for potable water and thus jeopardize the use of Lake Washington as a potable water supply by Brevard County. This problem argues strongly for containment of agricultural discharges, elimination of canal flow, and restoration of higher water levels.

The most severe water quality problem in an ecological sense is large daily and seasonal fluctuations in the concentration of dissolved oxygen. The causes of daily fluctuations in dissolved oxygen are poorly understood but they probably represent natural fluctuations caused by day to day variation in the balance between primary production; respiration; and physical factors affecting oxygen levels, such as winds and rainfall. Seasonal declines in oxygen concentrations occur during inundation of previously dry marsh and can result in low dissolved oxygen concentrations nearly basin-wide, particularly after prolonged periods of low water level. Because development of the floodplain caused a chronic surface water deficit (increased frequency and intensity of marsh dewatering) for much of the river and for lakes Hell 'n Blazes, Sawgrass, and Washington, it exacerbated this problem. To alleviate the surface water deficit canal flow and diversion of water to the coast should be eliminated and floodplain storage augmented to the extent practicable.

Two quite different temporal patterns of sedimentation in the lakes are suggested by current information: 1) temporary accrual of sediments in upstream lakes and periodic transport of sediments downstream during intense storms; and 2) slow, but permanent, accumulation of sediments in the small upstream lakes, with the deposition rate increasing markedly (from <0.02 - 0.27 g/cm^2/year) over the last century, and temporary accrual in the large, downstream lakes. We believe the preponderance of evidence supports the latter pattern and indicates that the primary cause of increased sedimentation was channelization of the marsh rather than chemical control of water hyacinth. As with the problem of water column mineralization, this argues for the elimination of canal flow.

Analysis of early (1943) and late (1980 or 1981) aerial photographs of selected areas (approximately two square miles each) of the marsh demonstrates a large increase (up to 89 percent) in the cover of woody vegetation. This alteration of the species structure of the marsh happened only where a long-term decline in surface water levels had occurred (i.e. between the Fellsmere Grade and the Lake Washington weir). Normalization of the hydrologic regime through elimination of both canal flow and interbasin diversion and through augmentation of floodplain storage would probably reverse this trend.

Neither the decline in water quality nor the increase in sedimentation rates in Lake Hell 'n Blazes seems sufficiently severe or pervasive to have caused the declines in fish and wildlife populations. These impacts were, apparently, largely the result of alteration of the floodplain marsh by an anthropogenic surface water deficit and reduction in the size of the marsh via drainage and agricultural development of large areas of the floodplain. Restoration of the river and its lakes, therefore, must entail restoration of the floodplain marsh.

INTRODUCTION

The Upper St. Johns River consists of a chain of lakes lying within a vast, floodplain marsh. With a combined area of about 7,126 ha, the lakes are a relatively small component of a riverine, lacustrine, and palustrine ecosystem which once covered more than 173,000 ha. Consequently, one would expect that their ecological integrity is inextricably linked to the integrity of the river and the floodplain marsh. Since early in this century, however, the floodplain has been extensively altered to allow agricultural development of its histosols and easy traverse of its broad marsh. Roadbeds now segregate large portions of the marsh and borrow canals channelize the flow of water. Greater than half of the original marsh has been diked and drained. Remarkably, the river and its lakes have sustained these alterations without severe ecological damage. There is disturbing evidence, however, that deleterious ecological trends have been initiated by floodplain development. Most of these trends are difficult to substantiate because of the dearth of biological data antecedent to development, but the high regional value of the river and it lakes makes it imperative that the impacts of development be well delineated and clearly understood. This is particularly important considering that a major goal of the multi-million dollar water management project to be constructed by the U.S. Army Corps of Engineers and locally sponsored by the St. Johns River Water Management District (District) is an ecological restoration of the river and its lakes.

In order to enhance our understanding of the ecological effects of floodplain development, the District submitted a proposal for research under the Clean Lakes Program of EPA. On October 14, 1981, a work agreement was signed between the Florida Department of Environmental Regulation and the District to conduct ecological studies of the Upper St. Johns River chain of lakes (i.e., Lakes Blue Cypress, Hell'n Blazes, Sawgrass, Washington, Winder and Poinsett). This agreement provided partial funding to the District for a phase I study under the Clean Lakes program of EPA. The plan of study for the diagnostic portion of this work outlined investigations in four major areas: 1) general water quality, 2) agricultural pumpage water quality, 3) the rate and nature of lake sedimentation, and 4) the relationship between floodplain vegetation and hydrology. This report discusses the results and conclusions of the diagnostic study. In addition an extensive literature review is provided in order to familiarize the reader with background information on the natural history and regional value of the Upper St. Johns River and on the ecological impacts of its development.

Regional Importance of the River

The Upper St. Johns River Basin is located in east central Florida in Brevard, Indian River, Okeechobee, Orange and Osceola Counties (Figure 1). It is bounded by the Atlantic Coastal Ridge to the east and the Kissimmee River Basin to the west. For this report, the northern boundary is SR 520 and the southern boundary is the Florida Turnpike giving a total area of 372,240 ha (919,805 acres).

The total population of the basin is about 95,080, with about 92,500 people in Brevard, 2,000 in Indian River, and 320 and 260 people in Okeechobee and Osceola Counties, respectively (Marella and Ford, 1983). About 7 km^2 (2.7 $mi.^2$) of the basin lie within Orange County.

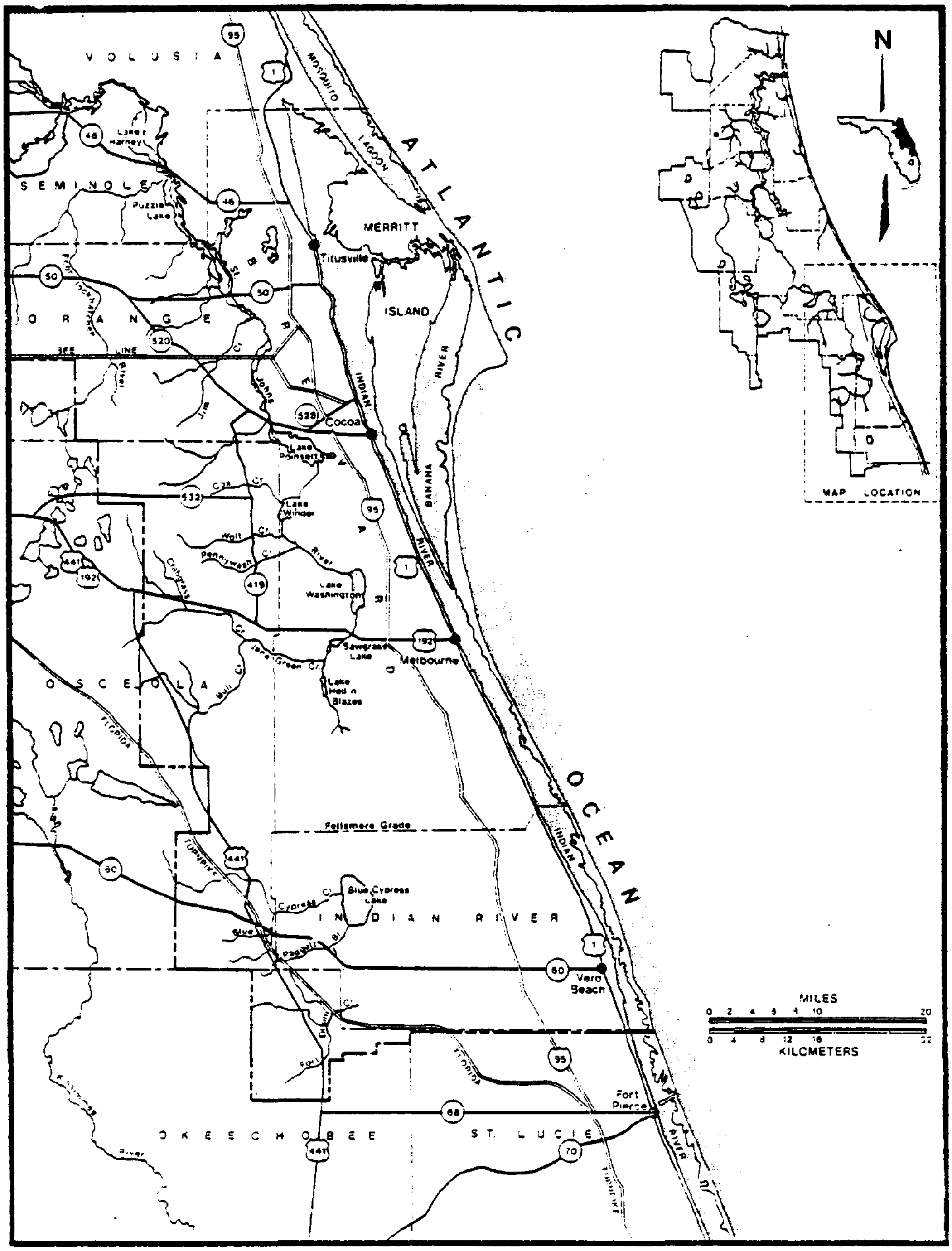


Figure 1. Map of the southern portion of the St. Johns River Water Management District including the Upper Basin.

The population of this area is unknown but is estimated to be less than 50 people. There is, of course, additional population outside the hydrologic basin but in close proximity to it (Table 1). The major cities in the study area are Melbourne(pop. ca. 50,000) and Cocoa (pop. ca. 17,000) located in Brevard County, and Vero Beach (pop. ca. 16,000) in Indian River County.

Table 1. Current and projected population estimates for counties represented in the upper St. Johns River basin (Smith, 1982).

	Estimated population			Estimated 20 yr. increase	
County	1981	1990	2000	Number	Percent
Brevard	281,496	331,100	337,300	95,804	34
Indian River	63,100	86,800	108,700	45,600	72
Okeechobee	21,139	25,700	30,700	9,561	45
Orange	481,731	573,900	668,700	186,969	39
Osceola	55,332	81,700	107,100	51,768	94

The surface waters of the Upper St. Johns River are a valuable regional resource in terms of potable water supplies, agriculture, recreation, and wildlife. Although underground aquifers are the most important source of potable water in the Upper Basin, the city of Melbourne withdraws about 12.4 MGD from Lake Washington to provide drinking water for about 107,000 people in the southern part of Brevard County (Brevard County Comprehensive Plan, 1981). In recognition of its value as a potable water source, the Florida Department of Environmental regulation classifies the river as "I" (public water supply) from Lake Washington south.

Drained areas of the river floodplain support extensive agriculture. Pasture, both native and improved, is the major land category in the Upper Basin (Table 2). Over 70 percent of the basin, about 263,000 ha (649,450 acres) is used for beef and dairy cattle production. Crop agriculture, the second most important land use, occupies only about 18,660 ha (46,109 acres) or 5 percent of the basin area, and consists predominantly of citrus and some corn.

Table 2. Land-use in the upper St. Johns River basin (SJRWMD, 1980).

Land Use Class	Hectares	Acres	Percentage of Total Basin Area
Open Water	8,981	21,303	2.3
Wetlands	68,981	170,451	18.5
Silvaculture	519	1,283	0.1
Pasture	262,829	649,450	70.6
Agriculture	18,660	46,109	5.0
Urban	12,544	30,997	3.4
Total	372,154	919,593	99.9

The river is an important source of irrigation water for agricultural industries, particularly citrus and row crop production. Although crop agriculture occupies only 5 percent of the basin area, irrigation accounts for about 92 percent of the total ground and surface water consumption in the basin. By comparison, public and domestic uses account for about 6.9 percent (Marella, 1982).

The river is also an important recreational resource for the roughly 200,000 people in Brevard and Indian River counties, because it is their closest freshwater fishing and recreational area. In the past, recreational activity in the Upper St. Johns chain of lakes was restricted primarily to fishing. However, with rapid population growth, the Upper Basin is now used for a variety of other pursuits such as bird watching, frogging, air boating and hunting. The Upper Basin lakes are readily accessible to the general public, in fact, the entire basin population is located within a one hour drive of one of the lakes. Several boat ramps provide access to the lakes or the river mainstream and recreation areas enhance utilization (Table 3).

Table 3. Boat ramps and recreation areas in the upper basin.

County	Location	Name	Comments
Indian River	Blue Cypress Lake	Blue Cypress Lake Park	Camping, picnicing, boat ramp
Indian River	Blue Cypress Lake	Middleton Fish Camp	Fuel, groceries, boat rentals, cabin rentals
Brevard	Lake Washington	Lake Washington Resort, Inc.	Fishing supplies, groceries, fuel, boat ramp
Brevard	St. Johns River & US 192	Camp Holly	Fishing supplies, camping, boat rentals
Brevard	St. Johns River & US 192	Lake Hell 'n Blazes Fish Camp	Fishing supplies, boat ramp, marine repair
Brevard	St. Johns River & US 192	Florida Airboat Assn. Ramp	Public
Brevard	Lake Poinsett off SR 520A	Poinsett Lodge	Restaurant, fishing supplies, boat slips, Fuel, boat ramp
Brevard	St. Johns River & SR 520	P. Lover Point Boat Launching	
Brevard	Lake Washington	Fish camp west of town	Fishing supplies, snacks, boat ramp, dock, boat slips
Brevard	St. Johns River & SR 520	Lone Cabbage Fish Camp & Marina	Restaurant, bar, gameroom, boat slips, boat ramp
Orange	St. Johns River & SR 520	Orange County Boat Ramp	County ramp

Many anglers fish in the basin for bass, crappie, bream and catfish each year. About 90 percent of the fishermen live within 80 km (50 mi.) of their fishing destination, and less than 5 percent are from outside the state (Cox et al, 1982). A creel survey conducted in 1978 (Cox et al, 1982) showed that about 60,000 hours were spent fishing in the area between the Fellsmere Grade and Lake Poinsett. Assuming an average expenditure of $11 per 4 hour fishing day (U.S. Dept. of Commerce, 1982), the survey indicates that about $165,000 per year is spent by fishermen using this area.

Although the basin is not used extensively for duck and coot hunting, certain areas of the basin provide good waterfowl hunting habitat (Sincock, 1958). The marshes south and east of Lakes Poinsett and Winder provide the best habitat, while the marsh west of Lake Washington provides fair waterfowl habitat. Hunting is poor from US 192 to SR 60 except for the marshes near the junction or SR 512 and SR 60 which offer good habitat and fair recreational hunting.

From a less anthropogenic perspective, the river and its associated wetlands provide critical habitats to many non-game species which depend on good water quality and the seasonal inundation of vegetated floodplains for their existence. For example, the basin has been used by several "rare" or "endangered" wetland species such as the wood stork (*Mycteria americana*) (endangered), snail kite (*Rostrhamus sociabilis plumbeus* (endangered), dusky seaside sparrow (*Ammospiza maritima nigrescens*) (endangered), osprey (*Pandioa haliaetus carolinensis*) (threatened), bald eagle (*Haliaeetus lencocephalus lencocephalus*) (threatened), and the Florida sandhill crane (*Grus canadensis pratensis*) (threatened). Moreover, several species of herons, egrets and ibis, depend on the natural wetlands habitat found in the basin for feeding and nesting. Nesbitt et al. (1982) reported nine nesting sites for herons and their allies in the basin (Table 4). Undoubtedly, inumerable species of invertebrates, fish, amphibians, reptiles, birds, and mammals which are unrecognized due to their lack of direct value as game or due to their unspectacular nature depend upon and enhance the productive wetland ecosystems of the basin.

Climate and Surface Water Hydrology

The unique and valuable ecological system of the basin is a result of its climate, topography, hydrology,and geology. The climate is considered semi-tropical. Yearly rainfall averages 136 cm (53.4 in.),

Table 4. Wading bird nesting sites and number of nesting pairs in the Upper St. Johns River Basin (Nesbitt et al., 1982). '+' indicates nesting pairs were present but not counted.

Location	Site	Species	Nesting pairs		
			1976	1977	1978
1. St. Johns Drainage District	Cypress Trees	Anhinga		50	0
		Cattle Egret		1000	500
		Great Egret		150	100
		Snowy Egret		0	50
		White Ibis		0	100
2. Blue Cypress Creek Impoundment	Cypress Trees	Great Blue Heron		6	0
3. N. Little Sawgrass	Willows on small island	Cattle Egret		500	500
		Louisiana Heron		150	10
4. N.E. Sawgrass Sweetwater Camp	Willows	Anhinga		50	0
		Little B. Heron		100	+
		Cattle Egret		2500	8000
		Great Egret		10	0
		Snowy Egret			+
		Louisiana Heron			+
		Yellow-Crowned Night Heron		25	+
5. N.W. Sawgrass	Willows & shrubs along canal	Great Blue Heron	65	0	0
		Little Blue Heron	65		
		Cattle Egret	2300		
		Snowy Egret	125		
		Black-Crowned Night Heron	15		
6. N. E. Lake Washington	Willows	Little Blue Heron	25	5	0
		Cattle Egret	500	200	
		White Ibis	100	0	
7. River Channel S. of Lake Winder	Willows	Anhinga		25	
		Cattle Egret		2000	
8. Jane Green Swamp	Cypress	Wood Stork		20	0
9. Crabgrass Creek	Cypress	Great Egret		10	0
		Wood Stork		30	

with more than 60 percent occurring during the five month period, June through October (Table 5). During this period there is nearly a 50 percent chance of rain on any given day, usually in the form of a local thunderstorm (SJRWMD, 1979). Stream flow and water levels generally peak during the month of October in response to the seasonally heavy rainfall (Figures 2 and 3).

Table 5. Average monthly rainfall at two locations in the upper St. Johns River basin, 1941-1970.

Month	Titusville	Fellsmere
January	5.8 cm (2.29 in.)	5.6 cm (2.08 in.)
February	7.3 cm (2.88 in.)	5.8 cm (2.29 in.)
March	9.3 cm (3.66 in.)	9.0 cm (3.54 in.)
April	6.8 cm (2.67 in.)	6.7 cm (2.63 in.)
May	7.8 cm (3.06 in.)	9.8 cm (3.86 in.)
June	20.5 cm (8.09 in.)	18.8 cm (7.41 in.)
July	21.8 cm (8.60 in.)	17.2 cm (6.79 in.)
August	20.3 cm (8.00 in.)	20.3 cm (7.99 in.)
September	23.9 cm (9.44 in.)	22.0 cm (8.66 in.)
October	15.8 cm (6.22 in.)	17.4 cm (6.85 in.)
November	5.5 cm (2.18 in.)	5.3 cm (2.07 in.)
December	5.3 cm (2.11 in.)	4.3 cm (1.17 in.)
Total	150 cm (59.20 in.)	142.2 cm (55.88 in.)

The basin is in the coastal lowlands region of Florida, an area of low relief. Most of the river valley lies between 6.1 and 9.1 m (20 - 30 ft.) NGVD (National Geodetic Vertical Datum) (Figure 4). The river channel south of Lake Hell 'n Blazes is approximately 6.1 m (20 ft.) NGVD, with a vertical gradient of about 3.8 cm/km (0.2 ft./mi.), considerably steeper than in the lower St. Johns River (SJRWMD, 1979). East of the river valley lies the Atlantic Coastal Ridge, a natural drainage divide between the St. Johns River and the Indian River basin which ranges in altitude from 0 to 13.4 m (0 - 44 ft.) NGVD. In the

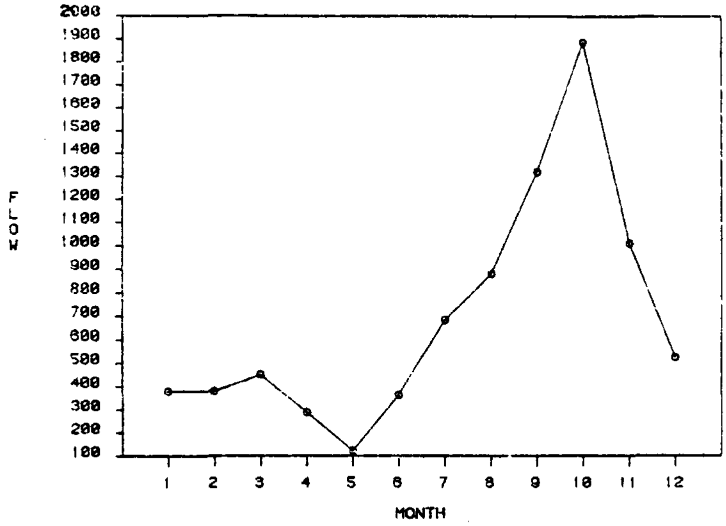


Figure 2. Mean monthly flow (cfs) of the St. Johns River at Melbourne, 1940 - 1978.

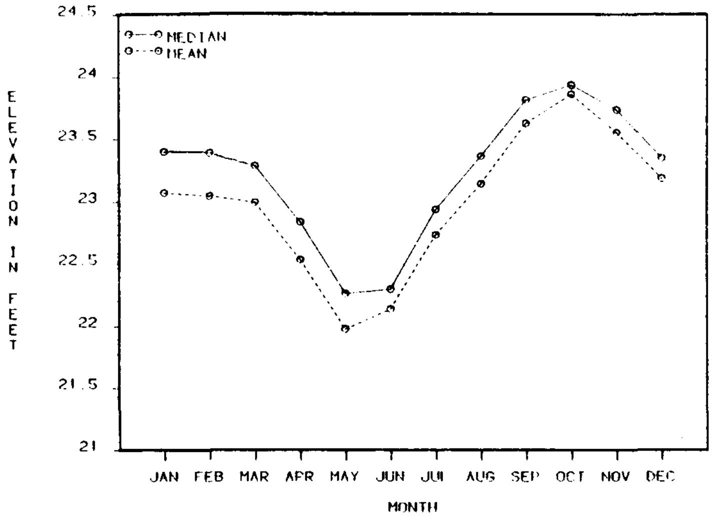


Figure 3. Monthly variation in median and mean lake elevation. Blue Cypress Lake, October 1956 - November 1982.

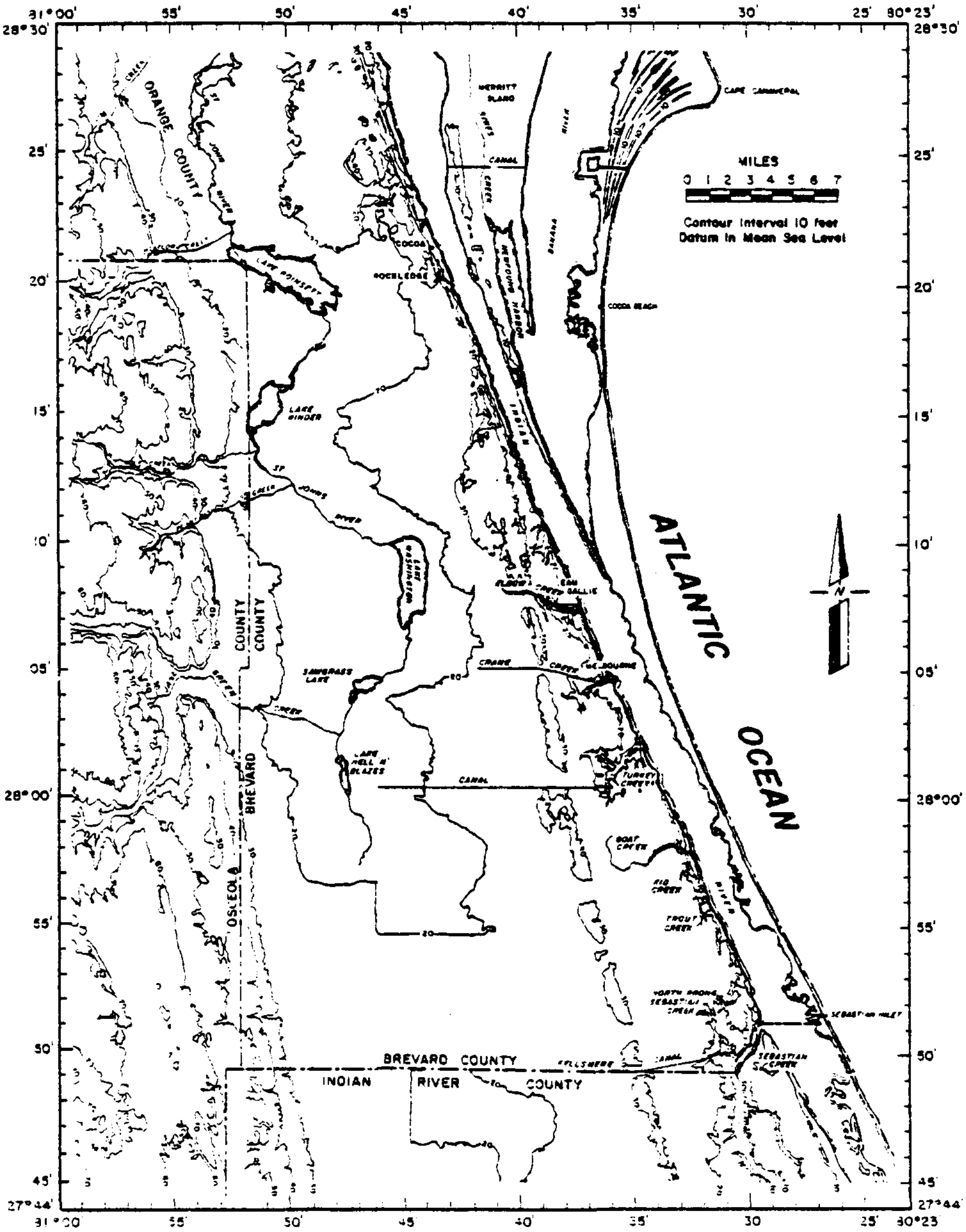


Figure 4. Topographic map of the Upper St. Johns River Basin (Brown et al., 1962).

southern part of Indian River County the Ridge is less prominent, making it difficult to distinguish the boundary between the St. Johns and Indian River basins. The river valley is bordered on the west by the Osceola Plain (18 m [60 ft.] NGVD) and the Talbot Terrace (12 m [40 ft.] NGVD) (White, 1970).

Permanent tributaries occur only on the west side of the Upper St. Johns River Basin, originating on the slopes of the Osceola Plain and the Talbot Terrace. The major tributaries are Fort Drum, Blue Cypress, Jane Green, Pennywash and Wolf Creeks (Table 6; Figure 1). USGS stream flow data are available for only Wolf and Jane Green Creeks. The average flow for Wolf Creek near Deer Park is 0.97 cms (34.4 cfs) while the average flow for Jane Green Creek near Deer Park is 7.5 cms (264 cfs) (SJRWMD, 1979). Average flow of the St. Johns River at Melbourne is 19.5 cms (691 cfs), while the maximum recorded flow is 509 cms (18,000 cfs). Interestingly, although the average flow of the river at Melbourne is greater than the average flow from Jane Green Creek, the maximum recorded discharge for the creek (521 cms [18,400 cfs]) is

Table 6. Characteristics of major tributaries of the upper St. Johns River.

Tributary	Length (km)	Source Elevation (m)	Mouth Elevation (m)	Average Fall (m/km)	Maximum Flow* (cms)	Receiving Body
Fort Drum	11.9	19.5	8.2	0.9	39.9	St. Johns Marsh
Blue Cypress	21.0	18.2	7.3	0.5	-	Blue Cypress Lake
Jane Green	21.0	7.6	6.1	0.1	521	St. Johns River
Pennywash	17.0	18.2	4.6	0.8	-	St. Johns River
Wolf	10.0	18.2	6.1	1.2	218	St. Johns Marsh

* Average flows only available for Jane Green and Wolf Creeks; 7.0 cms and 0.9 cms, respectively.

greater because of its steeper gradient and lower floodplain storage capacity.

Intermittant suface water flow to the river is contributed by drainage from the Atlantic Coastal Ridge. The Ridge is drained by a series of small interconnecting depressions that direct water westward into the river (Brown et al., 1962).

There are six principle lakes in the Upper St. Johns: Blue Cypress, Hell 'n Blazes, Sawgrass, Washington, Winder and Poinsett (Table 7). Blue Cypress Lake, located in Indian River County, is the headwater lake of the basin but is not connected to the other lakes by a natural channel. Prior to development, water flowed north from Blue Cypress Lake through a vast marsh, to the south end of Lake Hell 'n Blazes, where a permanent natural channel existed. Now, however, the lake is directly connected to the downstream lakes via canals.

Table 7. Characteristics of principle Lakes in the upper St. Johns River basin.

	Surface Area		Center of Lake		Elevation (m)	
Lake	(ha)	(acres)	latitude	longitude	max.	min.
Blue Cypress	2653	6555	27°43'36"	80°45'12"	8.14	5.59[1]
Hell 'n Blazes	154	381	28°01'00"	80°47'44"		
Sawgrass	195	481	28°04'11"	80°46'48"		
Washington	1766	4362	28°08'19'	80°44'39"	6.21	3.01[2]
Winder	606	1496	28°15'05"	80°50'59"		
Poinsett	1755	4334	28°20'15"	80°49'38"	5.35	2.43[3]

---

[1] 1956 - present
[2] 1942 - present
[3] 1941 - present

Geology and Groundwater Aquifers

The topography and groundwater resources of the basin to a large extent reflect its geologic history. The river incises the Pamlico terrace, a flat plain formed when sea level was approximately 9.1 m (30 ft.) higher than today (Figure 5), and, apparently, the river was once a large coastal lagoon. The alignment of Upper Basin lakes along the river mainstream, and the presence of deep mineral and organic sediments between lakes indicates that they are remnants of this continous lagoon and became isolated due to sedimentation (White, 1970).

The shallow aquifer, the primary source of municipal and domestic water, consists of up to 45 m (150 ft.) of unconsolidated sand, shell and some silt and clay (Crain et al, 1975; Figure 6) of Pleistocene and Recent age (Brown et al., 1962). These sediments contain nonartesian water of generally good quality. Below these sediments lies the Hawthorn Formation. This formation, which is from 3 to 90 m (10 to 300 ft.) thick, is the confining bed which retards the upward movement of artesian water from the underlying Floridan aquifer (Brown et al., 1962; Crain et al., 1975). The top of the Floridan aquifer is generally between 30 and 120 m (100 and 400 ft.) below the ground surface. Water within the Floridan is generally not potable because of high chloride content, but since it is a high volume source of fair quality artesian water, it is used extensively as a source of irrigation water.

A fault apparently exists in the Floridan aquifer along the general alignment of the river between Lakes Washington and Poinsett (Brown et al., 1962). Surface water quality and geochemical analysis suggest that this fault allows upward infiltration of saline Floridan water into the surface waters, which elevates chloride levels in this area, especially

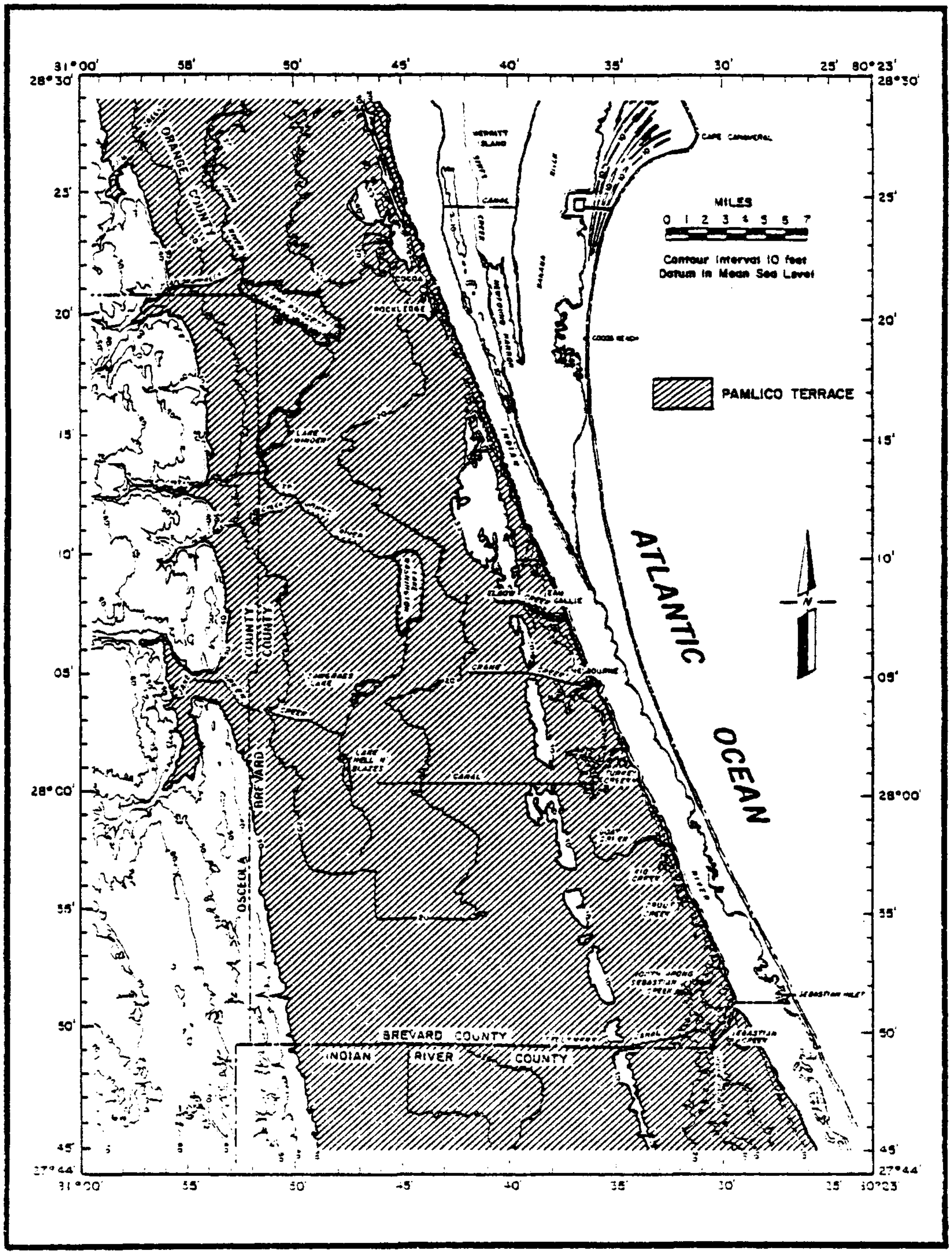
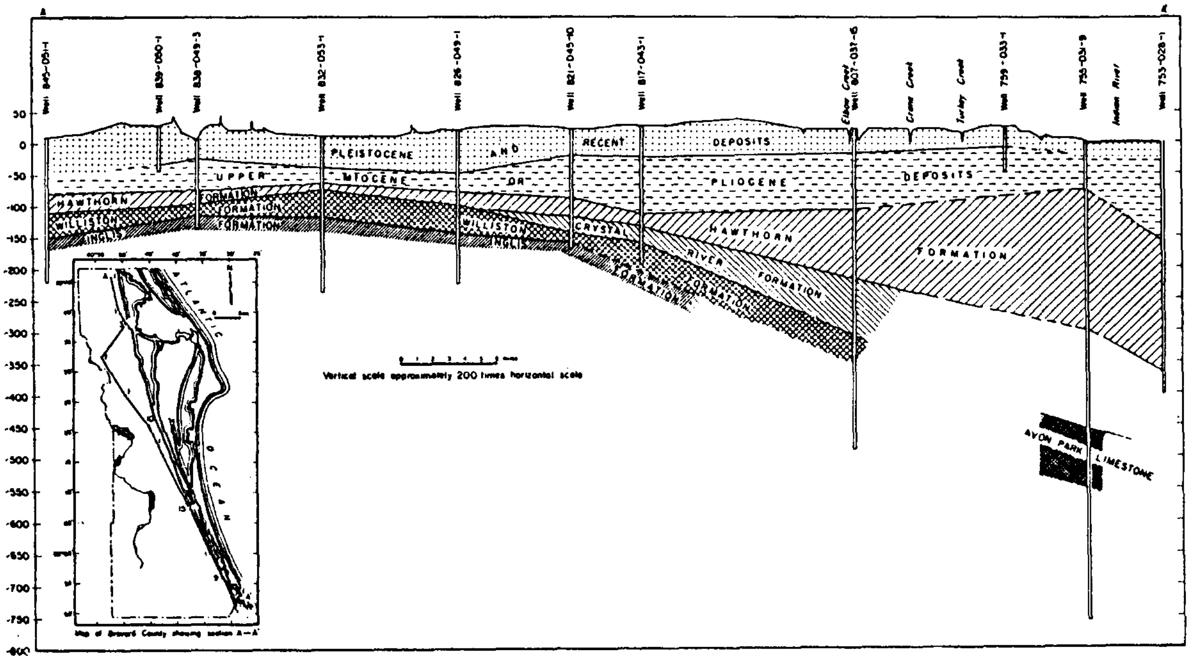


Figure 5. The Pamlico Terrace in the Upper St. Johns River Basin (Brown et al., 1962).



Geologic age	Stratigraphic unit		Approximate thickness (feet)	General lithologic character	Water-bearing properties
Recent	Pleistocene and Recent deposits		0-110	Fine to medium sand, coquina and sandy shell marl.	Permeability low due to small grain size, yields small quantities of water to shallow wells, principal source of water for domestic uses not supplied by municipal water systems.
Pleistocene					
Pliocene	Upper Miocene or Pliocene deposits		20-90	Gray to greenish gray sandy shell marl, green clay, fine sand, and silty shell.	Permeability very low, acts as confining bed to artesian aquifer, produces small amount of water to wells tapping shell beds.
Miocene	Hawthorn Formation		10-300	Light green to greenish gray sandy marl, streaks of greenish clay, phosphatic radiolarian clay, black and brown phosphorite, thin beds of phosphatic sandy limestone.	Permeability generally low, may yield small quantities of fresh water in recharge areas, generally permeated with water from the artesian zone. Contains relatively impermeable beds, that prevent or retard upward movement of water from the underlying artesian aquifer. Basal permeable beds are considered part of the Floridan aquifer.
Eocene	Ocala Group	Crystal River Formation	0-100	White to cream, friable, porous coquina in a soft, chalky, marine limestone.	Floridan aquifer: Permeability generally very high, yields large quantities of artesian water. Chemical quality of the water varies from one area to another and is the dominant factor controlling utilization. A large percentage of the ground water used in Brevard County is from the artesian aquifer. The Crystal River Formation will produce large quantities of artesian water. The Inglis Formation is expected to yield more than the Williston Formation. Local dense, indurated zones in the lower part of the Avon Park Limestone restrict permeability but in general the formation will yield large quantities of water.
		Williston Formation	10-50	Light cream, soft, granular marine limestone, generally finer grained than the Inglis Formation, highly fossiliferous.	
		Inglis Formation	70+	Cream to creamy white, coarse granular limestone, contains abundant echinoid fragments.	
	Avon Park Limestone		285+	White to cream, purple tinted, soft, dense chalky limestone. Localized zones altered to light brown or ashen gray, hard, porous, crystalline dolomite.	

Figure 6. Stratigraphic units and aquifers of the Upper St. Johns River Basin (Brown et al., 1962).

during low flow periods (personal communication, James Frazee, Hydrologist-St. Johns River Water Management District).

Soils

The soils of the Upper Basin can be divided into two broad categories: floodplain soils and flatwoods soils. The floodplain soils roughly delineate the 100 year floodplain, while the flatwoods soils occur in the areas between the river floodplain and the Atlantic Coastal Ridge to the east and the Osceola Plain to the west. Floodplain soils consist of very poorly drained peats and mucks (Histosols) adjacent to the river, and poorly drained more sandy soils (Alfisols and Mollisols) at slightly higher elevations. The U. S. Department of Agriculture, Soil Conservation Service (SCS) described the floodplain peats and mucks as a Montverde-Micco-Tomoka association, and the more sandy floodplain soils as a Felda-Floridana-Winder association (SCS, 1974; Figure 7).

The Montverde-Micco-Tomoka association is comprised of nearly level, very poorly drained, organic soils, with sandy-loamy material at a depth of 41 cm (16 in.) to greater than 132 cm (52 in.). Montverde and Micco peats contain about 80 percent fibrous organic material and occur mainly south of Lake Washington. Tomoka muck, commonly associated with Terra Ceia muck, is found mainly north of Lake Washington. It consists predominantly of well decomposed organics, with about 10-15 percent fibrous material. The peat and muck soils are frequently flooded for long periods (SCS, 1974).

The Felda-Floridana-Winder association, the other floodplain soil group, is sandy in contrast to the peats and mucks. It is comprised of nearly level, poorly drained and very poorly drained soils. They are

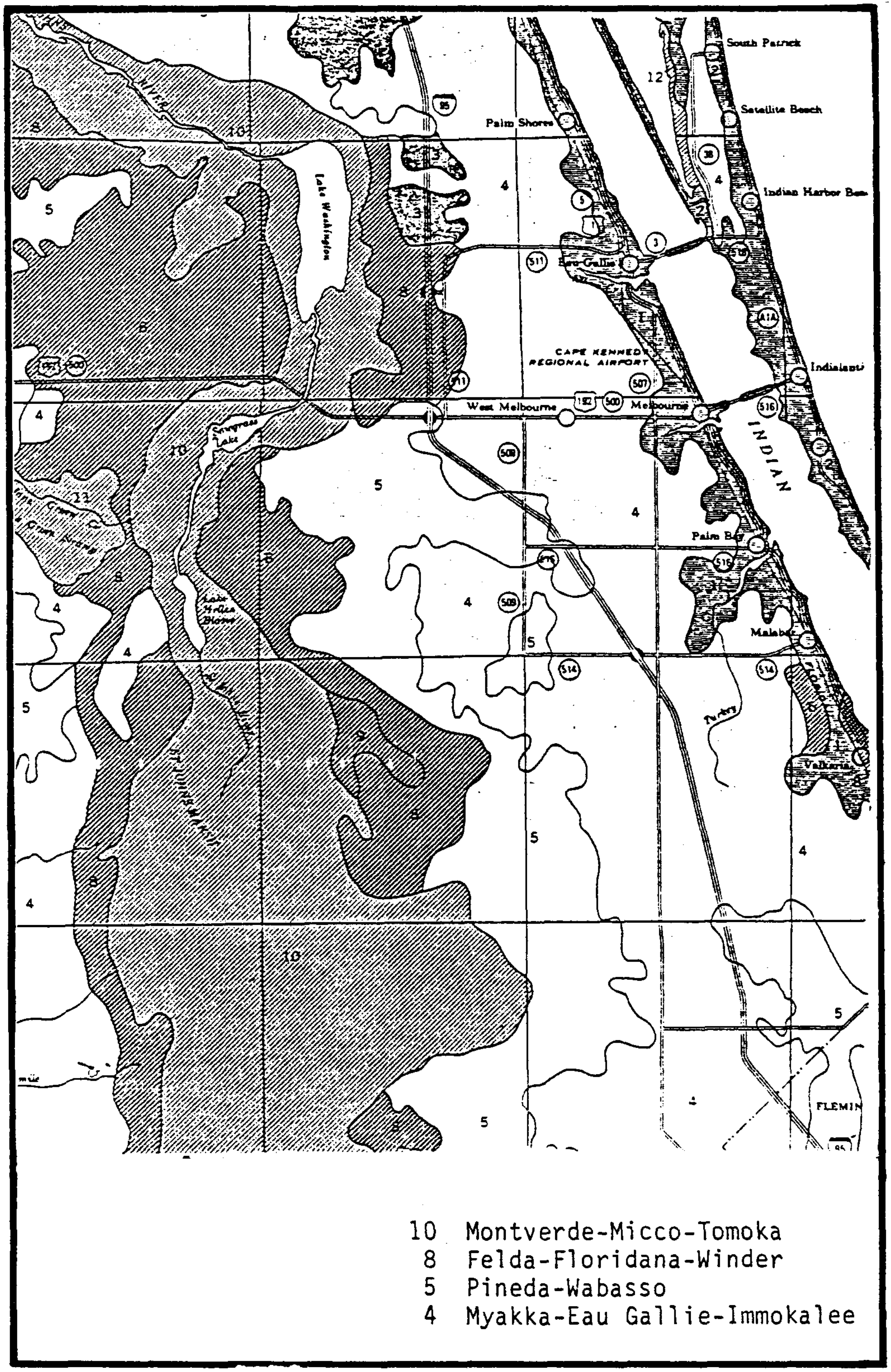


Figure 7. Soils of the Upper St. Johns River Basin (Soil Conservation Service, 1971).

sandy to a depth of less than 10 cm (40 in.) and are underlain by loamy material. The water table is generally close to the soil surface and flooding is common. Much of this floodplain soil association is used as improved or native pasture (SCS, 1974).

The flatwoods soils (Spodosols and Alfisols) are poorly drained, sandy soils which occur outside the river floodplain. Below the sandy surface layers are darkly colored, weakly cemented sandy layers that are underlain by sandy or loamy material. Two associations form what the SCS calls flatwoods soils: Myakka-Eau Gallie-Immokalee and the Pineda-Wabasso. These two associations make up a large portion of the basin land area, and generally support a combination of pine and saw-palmetto vegetation. The water table is high in most flatwood soils but the lands are readily converted to range, pasture or citrus (SCS, 1974).

Ecological alteration of the river

The high agricultural value of the floodplain soils of the basin spurred drainage of large areas of floodplain, beginning early in this century. Certain aspects of the ecology of the basin have been markedly affected by this development. The most conspicuous effect has been the extensive loss of wetland habitat. About 80 percent of the basin has been converted to agriculture, silviculture, pasture or housing (Table 2). Approximately 62 percent (65 percent excluding lakes) of the river's total floodplain and 38 percent (42 percent excluding lakes) of its annual floodplain is unavailable for floodwater storage or wildlife habitat (Table 8; Figure 8). The loss of annual floodplain, as opposed to floodplain at higher elevations, is especially detrimental because of the severe hydrological and ecological consequences.

Table 8. Areas and percentages of the entire upper St. Johns River basin and of the total and annual floodplains of the upper St. Johns River which have been developed (St. Johns River Water Management District, 1980).

Portion of Basin	Original Area		Developed Area		Percent
	Hectares	Acres	Hectares	Acres	
Total	372,240	919,805	294,552	727,839	79.1 (80.7)*
Total Floodplain	144,963	358,203	90,039	222,486	62.1 (65.3)
Annual Floodplain	71,515	176,713	27,006	66,733	37.8 (41.9)

*Values in parentheses are those percentages which result when the combined surface area of the lakes (17,609 acres) is not considered to be part of the annual or total floodplain.

Floodplain development has also had a direct impact on basin hydrology. The river floodplain once provided storage for large volumes of water during high flow periods. Prior to 1900, the width of the floodplain exceeded 20.9 km (13 mi.) at several locations along the river. Diking of developed wetlands reduced the maximum width to about 11.3 km (7 mi.) and eliminated over 60 percent of the total floodplain. The St. Johns River hydrologic model indicates that the concomitant reduction in floodplain storage capacity increased flood stages, caused peak flows to occur earlier during an equivalent rainfall event, and reduced dry season flow (Tai and Rao, 1982).

The extent of floodplain development was not uniform throughout the basin (Table 9) and this resulted in three fairly distinct hydrologic regimes in the three segments of the basin, i.e. above the Fellsmere Grade (Blue Cypress Lake area), between the grade and the Lake Washington Weir, (Lake Washington area), and below the Weir (Lake Poinsett area).

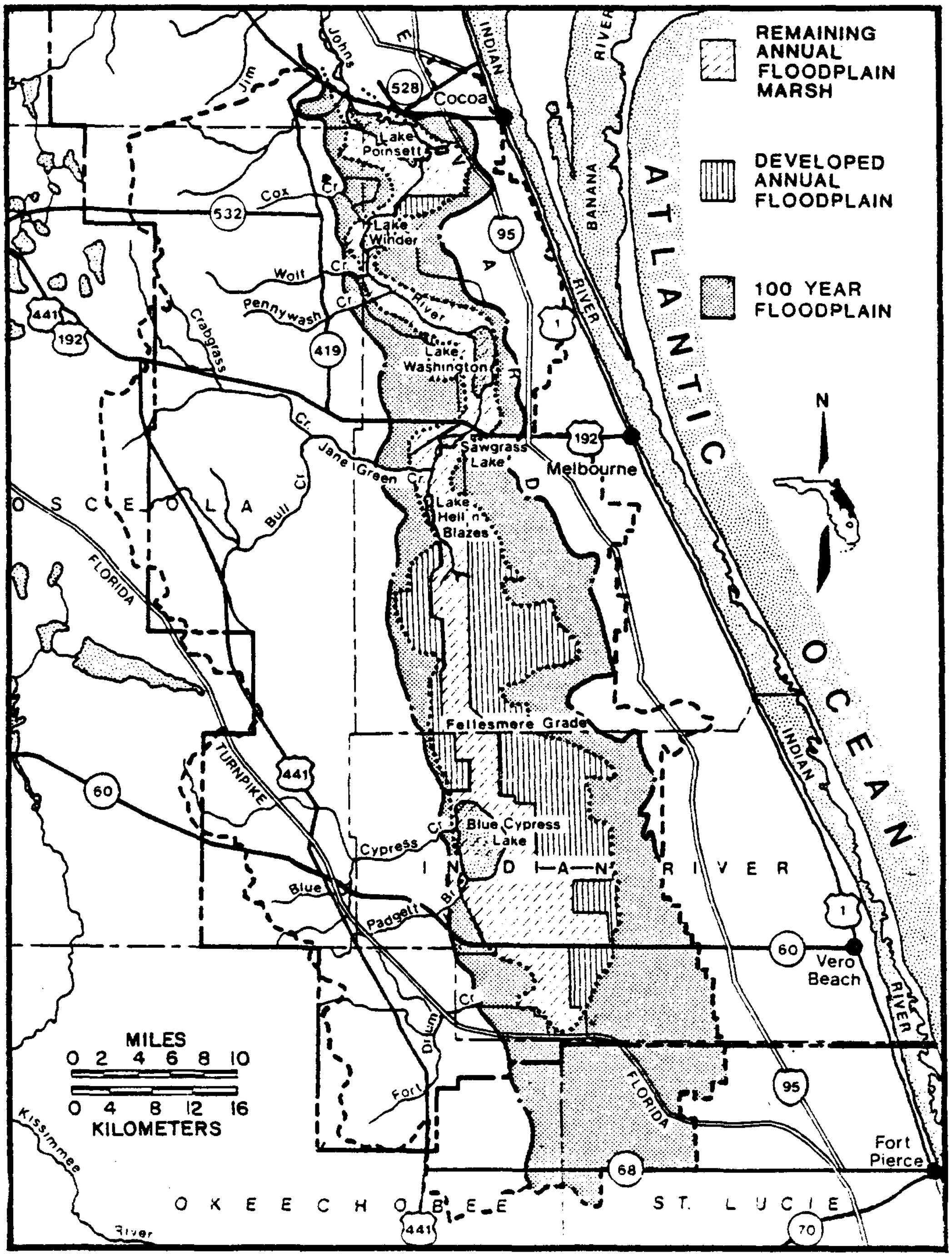


Figure 8. Development within the floodplain of the Upper St. Johns River.

Table 9. Areas and percentages of the total and annual floodplains of the upper St. Johns River which have been developed in three segments of the upper basin. Segments are as delineated in text. (St. Johns River Water Management District, 1980).

	Total Floodplain		Annual Floodplain	
Segments	Acreage	% Developed	Acreage	% Developed
Blue Cypress Lake Area	144,686	71.1	66,995	53.7
Lake Washington Area	126,369	75.1	58,759	45.3
Lake Poinsett Area	69,570	35.3	33,350	12.3

The Blue Cypress Lake area is hydrologically isolated during low flow periods by the Fellsmere Grade, a deserted road bed that crosses the river basin north of Blue Cypress Lake at the Brevard County line. The Grade prevents downstream flow, except for restricted flow through a small gap on the western edge of the marsh, when water levels in Blue Cypress Lake drop below 7.5 m (24.5 ft.) NGVD. Flow through the Grade is controlled by a privately owned, adjustable gate, water control structure (S-1). Water is normally impounded upstream of the structure in the dry season to be used for irrigation, and water is released during the rainy season. During the recent drought (1980-1981), however, the structure remained closed for two years, keeping water levels high south of the Grade and decreasing water levels north of the Grade. Flood stages for a mean annual or a 100 year storm event are about 0.61 m (2 ft.) higher than would have occurred under pre-development conditions (Tai and Rao, 1982) (Figure 9), but flood problems have been minimal due to the perimeter levees of agricultural lands and the diversion of water to the coast through canals.

Diversion of water from the St. Johns River to the Indian River Lagoon exacerbated low-flow problems caused by channelization and

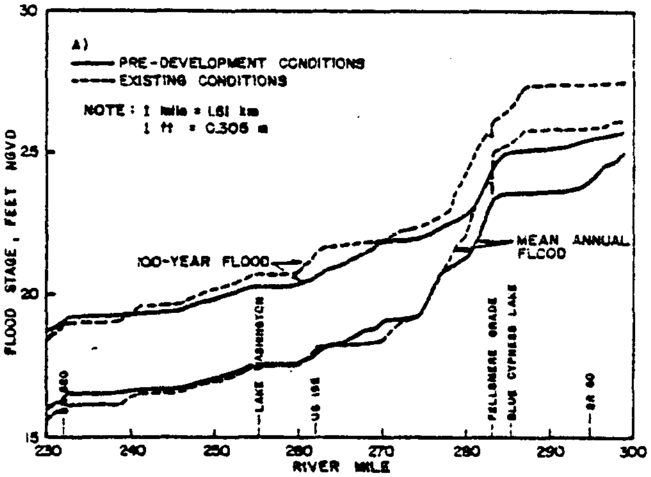


Figure 9. Changes in flood stages due to development of the floodplain of the Upper St. Johns River (Tai and Rao, 1982).

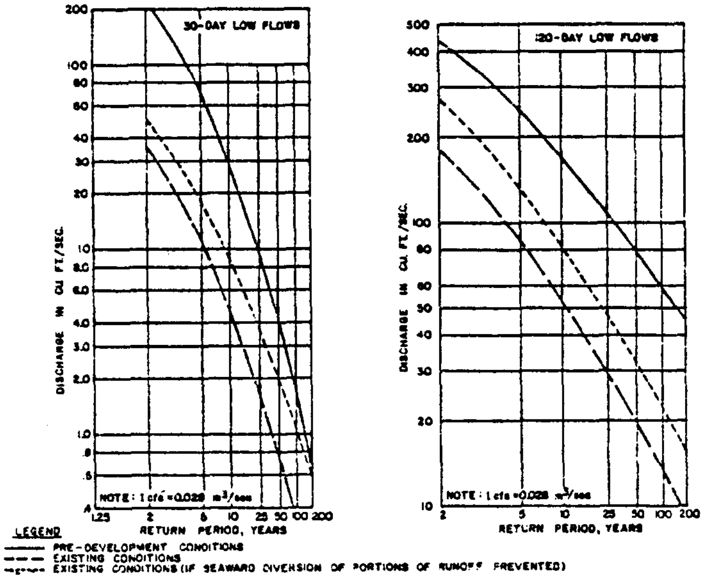


Figure 10. Low-flow frequency curves based on a 31 year water-shed simulation. St. Johns River at U.S. 192 (Tai and Rao, 1982).

diking. The average flow rate at US-192, between the Fellsmere Grade and Lake Washington, is only about 55 percent of the pre-development average flow (Tai and Rao, 1982). Without seaward diversion, the post-development average flow would be about 75 percent of the pre-development rate (Tai and Rao, 1982). The post-development average flow without diversion does not equal the pre-development rate because large volumes of rainfall are captured and retained behind levees. The current average 30 day and 120 day low-flow rates are about 13 percent and 38 percent, respectively, of the pre-development low-flows (Figure 10). These substantially decreased low-flow rates and the concomitant decreases in water quality, i.e. increased chloride concentrations, are of special concern to the City of Melbourne, which relies on Lake Washington for drinking water (Tai and Rao, 1982).

The Lake Washington area is isolated during dry periods from both upstream and downstream areas. Some isolation from downstream areas probably occurred prior to development, because flow was restricted below Lake Washington by the "Jams", floating islands of peat and vegetation lodged in the river channel north of Lake Washington. The Jams were removed in the early 1950's to improve navigation but were later replaced by man-made weirs. The present weir was built below Lake Washington in 1975 and prevents downstream flow when lake levels drop below 4.1 m (13.5 ft.) NGVD. Because of the weir and the Fellsmere Grade structure, flow into and out of this area stopped for extended periods during the recent drought (Fall, 1982). In contrast to the Blue Cypress Lake area, development decreased low-flows in this area and caused little increase in flood stages, except during severe storm events (Figure 9) (Tai and Rao, 1982). The net effect was apparently a reduction in the mean water level (Figure 11).

The Lake Poinsett area was apparently least affected by development (Table 9; Tai and Rao, 1982; SJRWMD, 1980). The stage-frequency relationship at SR 520 changed little over the past 20 years compared to the Lake Washington area (Figure 12), probably because the annual high flow discharges from Lake Washington did not increase significantly as a result of basin development (Tai and Rao, 1982) and because low flows were historically curtailed by the Jams.

Identification of hydrologic effects of development in the Lake Poinsett area is difficult for several reasons. As noted earlier, river flow was altered by removal of the Jams in the early 1950's. A sandbag weir was constructed below Lake Washington in 1961, but gradually collapsed. A permanent fixed weir was constructed at 4.1 m (13.5 ft.) NGVD in 1975. Finally, Taylor Creek, a tributary which historically flowed into the St. Johns downstream of SR 520, was diverted in 1969 by the Corps of Engineers so that it entered the river upstream of SR 520. It discharged into the river above SR 520 from 1969 to 1979, but in May 1979 was diverted back to its original channel. It appears that all these changes did not significantly alter either high or low stages.

Alteration of the basin in terms of floodplain habitat and hydrology would be expected to inititate biological changes and changes in water quality. Because the hydrologic alterations caused by development have been most severe in the area between the Fellsmere Grade and the Lake Washington Weir, the ecological effects of development would be expected to be more apparent there than in the Blue Cypress Lake or Lake Poinsett areas. Evidence does, indeed, indicate

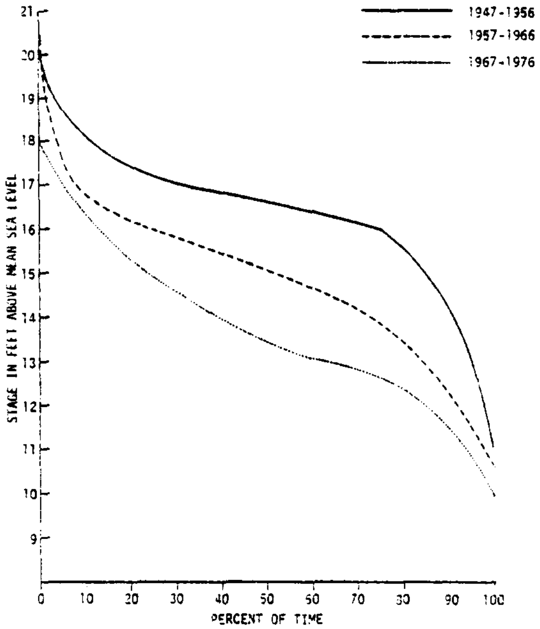


Figure 11. Stage-duration curves for the St. Johns River at Lake Washington (St. Johns River Water Management District, 1980).

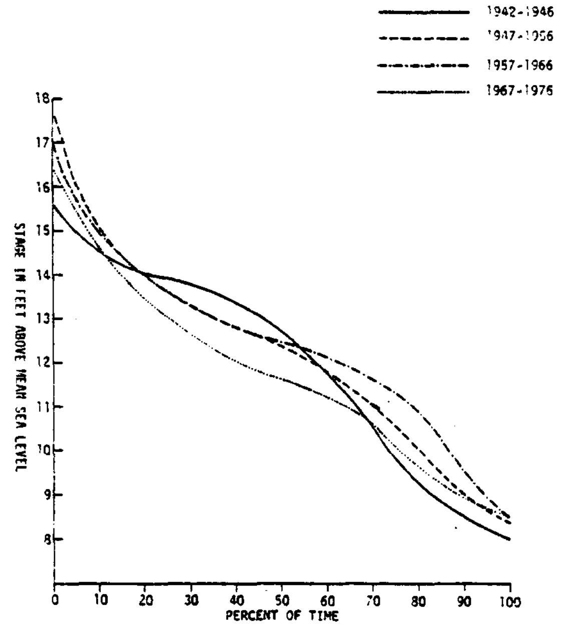


Figure 12. Stage-duration curves for the St. Johns River near Lake Poinsett at S.R. 520 (St. Johns River Water Management District, 1980).

that the Lake Washington area has been ecologically damaged.

Cox et al. (1976) presented evidence which suggested that a major problem in the basin was the rapid accumulation of unconsolidated organic sediments in the lakes, sediments primarily derived from water hyacinth detritus resulting from herbicide applications. They postulated that these sediments, primarily muck, had caused a gross degradation of spawning habitat for game fish. These ideas could not be substantiated, however, because they could find no reliable means for the determination of past or present sedimentation rates.

Despite the lack of conclusive data, the decrease in surface area of Lake Hell 'n Blazes indicated by their analysis of aerial photographs from 1920-1973 (459-290 acres; Table 10, Figure 13) suggests that increased sedimentation rates may be an important ecological effect of basin development.

Table 10. Changes in the open water area of Lake Hell 'n Blazes between 1920 and 1973 (from Cox et al, 1976).

Date	Lake Stage (msl)	Area of Open Water	Percent Change* in Area
1920	4.97 m (16.3 ft.)	185.8 ha (459 ac.)	-
1951	4.97 m (16.3 ft.)	151.8 ha (375 ac.)	19
1953	5.00 m (16.4 ft.)	110.9 ha (274 ac.)	40
1958	5.00 m (16.4 ft.)	110.9 ha (274 ac.)	40
1969	4.97 m (16.3 ft.)	110.1 ha (272 ac.)	41
1973	4.27 m (14.0 ft.)	109.3 ha (270 ac.)	41

* Percent change as compaared to area in 1920.

Cox et al.(1976) also reported that terrestrial plant species and woody species, such as willow (Salix caroliniana), had colonized large areas that were once dominated by wetland vegetation. They believed

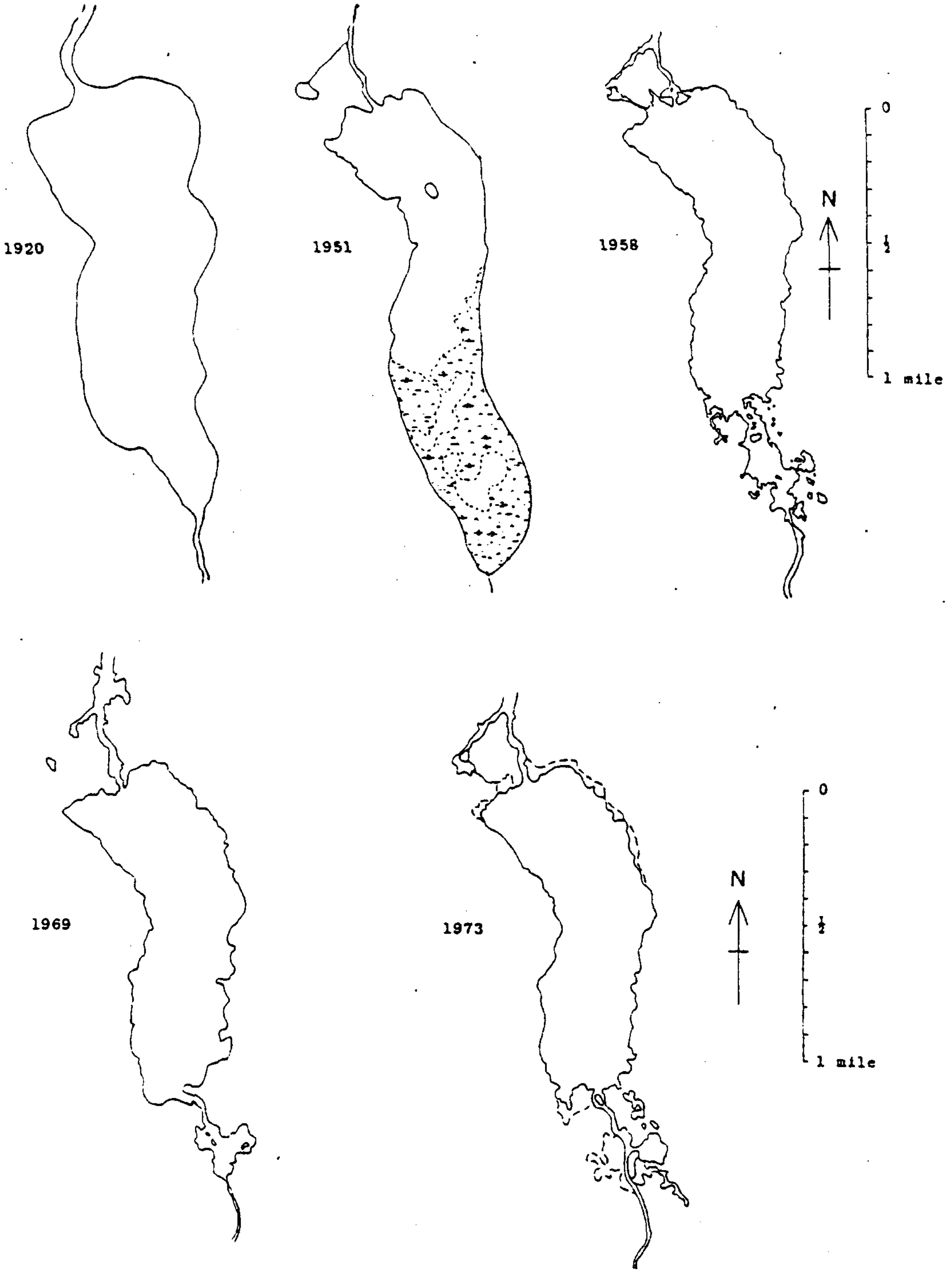


Figure 13. Shoreline changes in Lake Hell 'n Blazes, 1920 - 1973 (Cox et al., 1976).

this change was caused by a reduction in floodplain inundation but did not examine the basis for the reduction in inundation. Moreover, no quantitative data were presented to support the contention that plant communities had changed.

The Florida Game and Fresh Water Fish Commission (FGFWFC) (1981) reported that the 1980 migrant bird populations in the Upper Basin had declined substantially from pre-1958 levels. The Commission compared waterfowl populations from 1948 through 1958 to the 1972 to 1980 period, using low altitude aerial photographs. This work showed that the winter populations of coots and ducks had decreased significantly from about 12,000 birds per overflight to about 3,000 birds per overflight (t-test; 14 df; $p<0.001$; Table 11). They suggested that the decline was probably due to the loss of wetland feeding areas. They cited Sincock (1958) who believed that migrant waterfowl are attracted to three types of vegetation: deep marsh emergents such as Sagittaria spp. and

Table 11. Winter counts of populations of coots and ducks from 1949 to 1958 and from 1972 to 1980 (Florida Game and Freshwater Fish Commission, 1981).

Year	Count	Year	Count
1949	9,475	1972	5,100
1950	8,592	1974	1,500
1951	21,267	1975	1,200
1952	9,974	1976	6,300
1953	13,979	1977	3,700
1954	10,838	1978	1,300
1955	18,580	1979	300
1956	7,493	1980	4,900
1957	15,594		
1958	7,338		
Mean (S.D.)	12,313 (4,836)		3,038 (2,238)
95 % C.I.	8,854 - 15,772		1,167 - 4,909

Scirpus americanus; moist soil plants such as Rhynchospora spp. and Polygonum punctatum; and submerged aquatics such as Vallisneria americana and Ceratophyllum demersum. They reasoned that, if these plants provide habitat and food, as their populations decline so do the populations of migrant waterfowl.

Although quantitative data are unavailable, basin wading bird populations have likely declined as well since the 1930's. Kushlan and White (1977) reported that drastic declines have occurred in wading bird populations in south Florida despite the protection of nesting sites, indicating that the loss of feeding sites has been the key factor in the decline. For most wading birds feeding success depends on the seasonal water level fluctuations which concentrate fish in remnant pools during the winter and spring . These fish aggregations provide optimal foraging. The woodstork, a tactile or groping feeder, is probably the species most dependent on the dense fish concentrations caused by fluctuating water levels, and is on Florida's endangered species list. The number of woodstork breeding colonies has apparently declined severely in the Upper Basin. In 1927 there were about 125 active woodstork nests on Blue Cypress Lake (Howell, 1932). Now the total number of nests in the entire basin is about 50 and none are located around Blue Cypress Lake. Another endangered species, the Snail Kite, was once common in the basin (Howell, 1932) but is now rare.

Spatial variation in standing crops of fish and temporal variation in fishing success indicate that fish populations have also declined in the Lake Washington area. In 1925, the Lake Washington area was described in local newspapers as a fisherman's paradise, specifically renowned for its largemouth bass fishery (Cox et al, 1982). Recently, however, fishing success and effort have diminished (Cox et al., 1982). In fact, a creel survey conducted in 1955 and 1956 (Herke and Horel, 1958) indicated that about three or four times as many fishermen used the area in the late 50's as did in the late 70's. This suggests that fish populations have declined drastically, particularly in the Lake Washington area. One acre littoral zone block net samples taken by the Commission in the period 1972 to 1978 indicated that, even before the drought of 1980-1981, fish standing stocks in Lake Washington (15.5 kg/ha(13.8 lbs/ac)) were less than in Lake Poinsett (57.2 kg/ha(51.1 lbs/ac)) or Blue Cypress Lake (54.2 kg/ha(48.4 lbs/ac)) (Table 12). A two-sample t-test for populations with unequal variances (Ryan et al., 1982) shows that the standing stocks in Lake Washington were significantly smaller than in Lake Poinsett or Blue Cypress Lake (p = 0.024 and 0.045 respectively).

Table 12. Biomass (kg/ha.) of fish collected by one acre blocks nets in the littoral zones of lakes Blue Cypress, Washington and Poinsett. Unless otherwize indicated, number of samples (n) was one. Data from Cox et al. (1976) and Florida Game and Freshwater Fish Commission (1980).

	Lake		
Year	Blue Cypress	Washington	Poinsett
1972	-	24.0	35.3
1973	19.9	16.9	65.4
1974	119.8 (n= 5)	18.9	44.4
1975	21.8 (n= 5)	2.0	83.8
1976	86.9 (n= 6)		
1977	23.9 (n= 5)		
1978-1979	28.6 (n=10)		
1979-1980	78.7 (n= 4)		
Mean (S.D.)	54.2 (40.4)	15.5 (9.5)	57.2 (21.7)
95 % C.I.	16.8 - 91.6	0.4 - 30.6	22.7 - 91.7

The Florida Game and Fresh Water Fish Commission (Cox et al, 1982) reported that the fishery decline was due primarily to a decrease in the extent of marsh inundation. This contention is supported by the work of other researchers who have found positive relationships between fish production and marsh inundation (Wegener and Williams, 1971; Dineen et al., 1974; Aggus and Elliott, 1975; Martin et al., 1981; Ploskey, 1982). Forage fish and young sport fish thrive in inundated marsh vegetation such as maidencane (Panicum hemitomon) and pickerelweed (Pontederia cordata). Cox et al. (1982) reported densities greater than 200,000 small fish per hectare, equivalent to over 200 pounds of biomass, in these habitats. Such dense aggregations of fish reflect the value of these habitats in terms of overall productivity.

In addition to the apparent loss of production caused by development, fish kills seem to have occurred at an increasing rate during the past few years (Cox et al., 1982). Although the reduction in floodplain inundation may have contributed to this problem, other factors, more easily linked to development of the basin, may have precipitated the kills. For example, during the 1980-1981 drought, the Fellsmere Grade water control structure (S-1) was closed for over two years. Downstream flow was eliminated, while simultaneously, water was diverted out of the basin upstream of the grade. The cessation of flow through the grade apparently allowed sediments and organic detritus to collect in the canals above and below the structure. When normal rainfall returned and basin water levels rose in June 1982, S-1 was opened and the accumulated sediments were flushed downstream. This was the suspected cause of a fishkill several days later in Lake Sawgrass (Steve Moore, FGFWFC, personal communication, 1982). Apparently the biochemical oxygen demand of the resuspended sediments removed the dissolved oxygen from downstream waters. Several fish kills occurred over the last few years under similar conditions, i.e. a rapid release of water into canals that had been relatively stagnant for extended periods.

Development of the floodplain may also have initiated a decline in water quality. A natural characteristic of the basin is that when water levels are low mineralized ground water (surficial and artesian) flows more readily into surface water bodies (Brown et al, 1962; James Frazee, personal communication). This results in increased concentrations of chlorides, total dissolved solids, hardness, sulfate, magnesium, potassium, and sodium. Because development intensified low-flows, this problem was indirectly augmented by development. Agricultural activities may also have directly affected water quality but reports have been conflicting. Sullivan (1979) concluded, based primarily on an extensive literature review, that agricultural runoff and pumped discharges have adversely affected water quality. In particular, his report states that water quality is degraded through increased loadings of suspended solids, BOD and nutrients. The report was primarily concerned with the activities of Deseret Ranches of Florida, Inc., although the conclusion can apply to other ranches as well. On the other hand, a report by CH_2M Hill, a firm hired by Deseret Ranches, indicated that agricultural runoff and pumpage had insignificant effects on receiving waters (CH_2M Hill, 1979). These conclusions, however, were based on a preliminary water sampling program which covered only two sampling periods.

Objectives of the Diagnostic Study

As indicated by the foregoing review, several suspected impacts of basin development were poorly supported by data. These impacts; degradation of water quality due to agricultural discharges, augmentation of sedimentation rates in Lake Hell 'n Blazes through chemical control of water hyacinth, and alteration of floodplain vegetation due to a reduction in floodplain inundation; were the subjects of the diagnostic study.

The first of these impacts was examined through a general assessment of water quality and an investigation of the quality and quantity of pumped agricultural discharges. The second impact was examined through both field research and literature review. The field research produced data on bathymetry, sediment depths, and sediment composition and stratigraphy along longitudinal and transverse transects of each lake and along radial transects positioned at the river's entrance into each lake. Through comparison of these data with data from earlier studies an attempt was made to determine if sediment depths have increased and if the nature of sediments has changed. A literature study of control techniques and detrital dynamics of _Eichhornia crassipes_ (water hyacinth) was undertaken to complement the field studies (Gerry, 1982).

The last impact was investigated through an analysis of the role of hydrology in the determination of vegetation pattern in the marsh and through an analysis of temporal changes in the marsh vegetation and hydrology. The relationship between hydrology and floodplain vegetation was assessed through the collection of data on the present condition of the floodplain vegetation adjacent to the southernmost lake of the Upper

Basin, Blue Cypress Lake; (Figure 1) and examination of the relationship between the spatial distribution of plant species and hydrology (Lowe, 1983). Temporal changes in the distribution of plant communities were examined through interpretation of aerial photographs of selected areas over the period 1943-1981. For all but one of these areas long-term hydrologic data were also available so that temporal changes in hydrologic conditions could be assessed.

METHODS

General Water Quality

As part of the District's water quality monitoring program samples were collected quarterly, with November, February, May, and August as the target months, from November 1979 through October 1982. Sample collections were planned for 30 stations for the first year (Figure 14; Table 13). Beginning in November 1980, 16 stations were added in the Lake Washington area and in agricultural canals south of Three Forks Run. The station at Blue Cypress Marsh (BCM) was dropped because its location was subject to mixing with water from the lake. The actual number of stations sampled each quarter varied with water levels. Marsh and tributary sites which contained no water or were inaccessible under low flows were not sampled. The center of Lake Poinsett or Blue cypress Lake could not be sampled under extreme weather conditions.

During the first year in situ measurements were made at most stations three times daily (morning, noon, afternoon) to indicate daily fluctuations. This practice was discontinued in November 1980.

Water quality parameters measured varied throughout the study as equipment malfunctioned or new capabilities were added to the laboratory facility. Parameters sampled, analytical techniques, and equipment used are summarized in Table 14. Metals samples were collected only semiannually and were analyzed by the District's laboratory or by the South Florida Water Management District. Instruments were calibrated before and after each daily sampling period. pH meters were calibrated using two pH buffers. D.O. meters were initially calibrated using the Modified Winkler titration (Standard Methods, 14th ed.). Beginning in

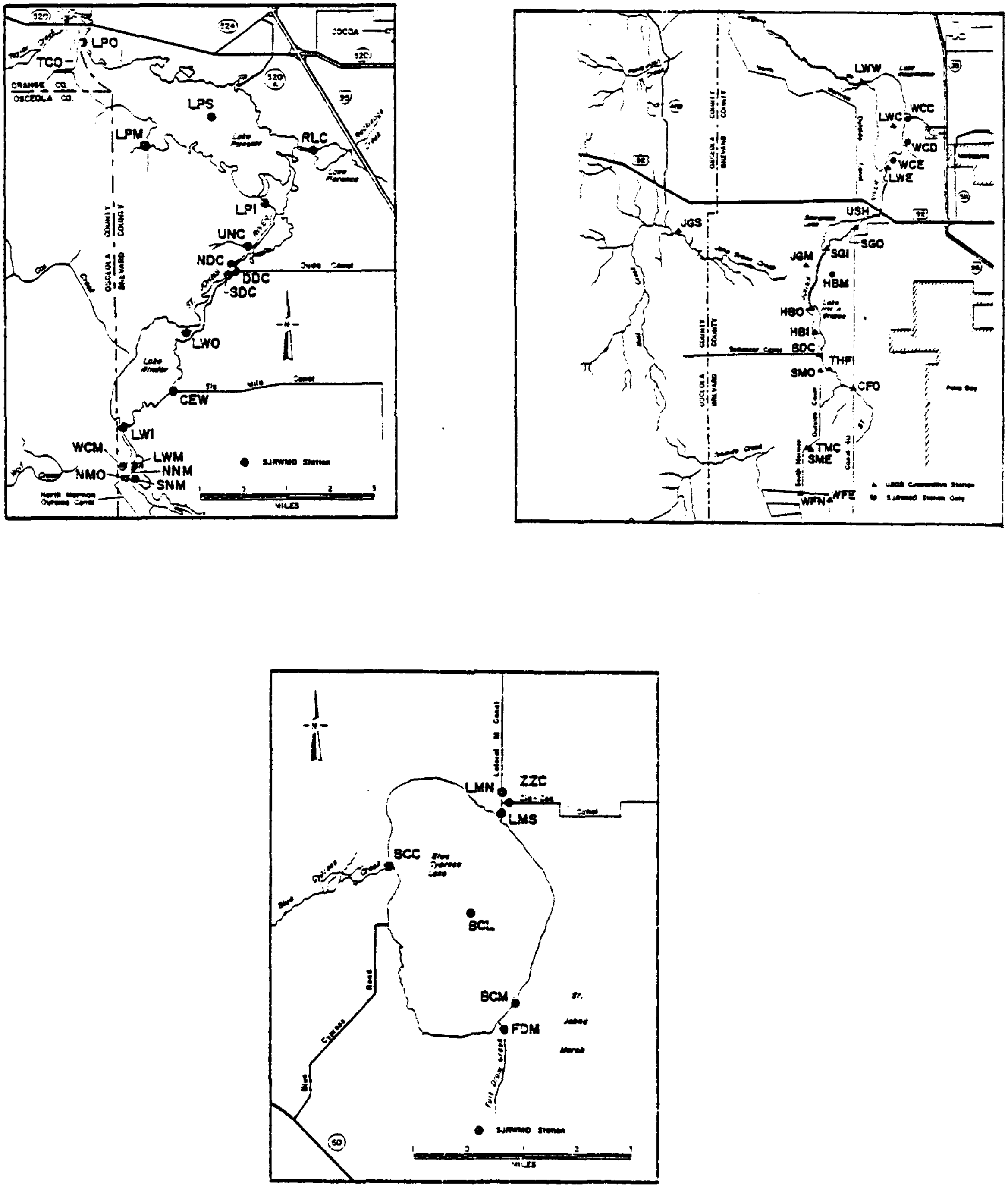


Figure 14. Locations of water quality stations in the Upper St. Johns River Basin monitored by the St. Johns River Water Management District.

Table 13. Names and locations of water quality monitoring stations of the St. Johns River Water Management District in the Upper St. Johns River Basin.

Station Name	Lat.-Long.	Description
FDM	274147-804432	Fort Drum Marsh
BCM**	274224-804412	Blue Cypress Marsh
BCC	274421-804639	Blue Cypress Creek
LMS	274520-804438	Lateral M South
ZZC	274527-804427	Zig Zag Canal
LMN	274538-804438	Lateral M North
BCL	274336-804512	Blue Cypress Lake
WFE*	275425-804713	Willowbrook Farms East
WFN*	275442-804823	Willowbrook Farms North
SME*	275621-804809	South Mormon Extension
SMO	275917-804732	South Mormon Outside Canal
CFO*	275834-804614	Canal 40
THF	275918-804710	Three Forks
BDC	275949-804735	Bulldozer Canal
HBI	280039-804744	Hell'n Blazes Inlet
HBO	280132-804756	Hell'n Blazes Outlet
HBM	280249-804708	Hell'n Blazes Marsh
JGM	280308-804811	Jane Green Marsh
SGI	280346-804723	Sawgrass Inlet
SGO	280437-804613	Sawgrass Outlet
USH*	280503-804511	U.S. Highway 192
LWE*	280648-804452	Lake Washington Entrance
LWC*	280819-804439	Lake Washington Center
LWW*	280956-804555	Lake Washington Weir
TMC*	275624-804811	Ten Mile Canal
JGS*	280427-805318	Jane Green Swamp
SNM	281255-805118	South of North Mormon Canal
NMO	281258-805139	North Mormon Outside Canal
NNM	281308-805130	North of North Mormon Canal
LWM	281330-805115	Lake Winder Marsh
LWI	281405-805132	Lake Winder Inlet
WCM*	281308-805132	Wolf Creek Marsh
CEW	281449-805026	Canal, East Bank of Winder
LWO	281605-804951	Lake Winder Outlet
SDC	281709-804914	South of Duda Canal
DDC	271715-804905	Duda Canal
NDC	281721-804909	North of Duda Canal
LPI	281835-804826	Lake Poinsett Inlet
LPS	282015-804938	Lake Poinsett
RLC	281938-804713	Rockledge Creek
LPM	281937-805105	Lake Poinsett Marsh
LPO	282131-805223	Lake Poinsett Outlet
TCO*	282211-805248	Taylor Creek Outlet

Table 13 (continued). Names and locations of water quality monitoring stations.

Station Name	Lat.-Long.	Description
UNC*	281741-804848	Unnamed Canal N. of Duda Canal
WCE*	280700-804438	S. Sarno Rd. Canal;E. Shore of L. Washington
WCD*	280744-804405	Tom's Canal; E. Shore of L. Washington
WCC*	280834-804402	Sands Canal; E. Shore of L. Washington

* Stations added after August 1980.
** Stations dropped after August 1980.

Table 14. Techniques for chemical and physical analysis of water samples.

Parameter	Storet Number	Sampled By	Method or Equipment
Turbidity	82079	2, 3	Nephelometric Method, Std. Methods, 14th ed., p. 132[1]
True Color	80	2, 3	Visual Comparison, Std. Methods, 14th ed., p. 64[2]. Spectrophotometric Method, Std. Methods, 14th ed.,p. 66[3]
B.O.D.	310	2, 3	Incubation, 5 day, Std. Methods, 14th ed., p. 543[1]
Total Alkalinity	410	2, 3	Bromcresol green-methyl red titration, Std. Methods, 14th ed., p. 280[1]
Suspended Solids	530	2, 3	Total Nonfilterable Residue @ 105° C, Std. Methods, 14th ed., p. 94[1]
Volatile and Fixed Residue	535	3	Total Nonfilterable Residue @ 600° C, Std. Methods, 14th ed., p. 94
Ammonia Nitrogen	610	2, 3	Automated Phenate Method, EPA Methods..., p. 350.1-132[1]
Nitrite Nitrogen	615	3	Diazotization Method, EPA Methods..., p. 354.1-1[3]
Nitrate Nitrogen	620	3	Colorimetric, Brucine, EPA Methods..., p. 352.1-1[3]
Total Kjeldahl Nitrogen	625	2, 3	Colorimetric, Semi-Automated Block Digestor, EPA Methods, p. 351.2-1[1]
TKN (Dissolved)	623	3	Colorimetric, Semi-Automated Block Digestor, EPA Methods, p. 351.2-1[1]
Nitrate-Nitrite Nitrogen	630	2, 3	Automated Cadmium Reduction Method, EPA Methods..., p. 353.2-6[1]
Total Phosphorus	665	2, 3	Single Reagent Method, EPA Methods..., p. 365.2-1[1]
Total Phosphorus (Dissolved)	666	3	Single Reagent Method, EPA Methods..., p. 365.2-1[1]
Hardness	900	2, 3	EDTA Titration, Std. Methods, 14th ed., p. 496

Table 14 (continued). Techniques for sample analysis.

Parameter	Storet Number	Sampled By	Method or Equipment
Calcium (Dissolved)	915	2, 3	Atomic Absorption Spectrophotometric Method, Std. Methods, 14th ed.
Magnesium (Dissolved)	925	2, 3	Atomic Absorption Spectrophotometric Method, Std. Methods, 14th ed.
Sodium (Dissolved)	930	2, 3	Atomic Absorption Spectrophotometric Method, Std. Methods, 14th ed.
Potassium (Dissolved)	935	2, 3	Atomic Absorption Spectrophotometric Method, Std. Methods, 14th ed.[2] Flame Photometric Method, Std. Methods, 14th ed.[3]
Chloride	940	2, 3	Automated Ferricyanide Method, EPA Methods... p. 325.1-1[1], Argentometric Method, Std. Methods, 14th ed., p. 303 (prior to 1981)[2]
Sulfate	945	2, 3	Turbidimetric Method, Std. Methods, 14th ed., p. 496[1]
Iron (Dissolved)	1046	2, 3	Atomic Absorption Spectrophotometric Method, Std. Methods, 14th ed.[1]
Chlorophyll-a, b, c	32210, 32212, 32214	2, 3	Trichromatic Spectrophotometric Method, Std. Methods, 14th ed., p. 1030[1]
Chlorophyll-a, corr.	32218, 32219	2	Pheophytin correction, Std. Methods, 14th ed., p. 1032[2]
Total Dissolved Solids	70300	2, 3	Total Filterable Residue @ 180° C, Std. Methods, 14th ed., p. 92[1]
Orthophosphate	70507	2, 3	Single Reagent Method, EPA Methods..., p. 365.2-1[1]

Table 14 (continued). Techniques for sample analysis.

Parameter	Storet Number	Sampled By	Method or Equipment
Orthophosphate (Dissolved)	671	3	Single Reagent Method, EPA Method..., p. 365.2-1[1]
Field Parameters			
Water Temperature	10	2, 3	*Hydrolab 6D12[2], YSI Model 33 SCT Meter[3]
Dissolved Oxygen	299	2, 3	Hydrolab 6D12[2], Leeds & Northrup 7932[3]
pH	400	2, 3	Hydrolab 6D12[2], YSI Model 33 SCT Meter[3]
Transparency	77	2, 3	Secchi Disk[1]
Depth		2, 3	Hydrolab 6D12[2], Secchi Disk[3]
Flow		2, 3	Marsh McBirney[1]

[1] FIT and SJRWMD

[2] SJRWMD

[3] FIT

*Back-up meters in the event of a malfunction consisted of a YSI Model 57 D.O. Meter, YSI Model 33 SCT Meter, and Cole-Parmer Digisense pH meter.

May 1980, air calibration with an air calibration chamber was used. Conductivity meters were calibrated with a 1413 umho/cm standard. Temperature readings were calibrated using a laboratory grade thermometer. Internal calibrations of the Hydrolab were checked. Calibration results were recorded on the Field Calibration Check Sheet.

Samples were primarily collected by hand at 0.5 m depth with a polyethylene bottle. Deep samples were collected using a Van Dorn sampler. Samples were collected the first year in re-usable Nalgene bottles appropriately washed and rinsed. Beginning November 1980 disposable polyethylene bottles were used, again appropriately rinsed. Samples were stored in iced coolers and transported to the laboratory within 24 hours. Metal and nutrient bottles were preserved with acid in the field or shortly before shipping. Metal samples were acidified to a pH less than 2 with nitric acid. Nutrient samples were acidified to a pH less than 2 with sulfuric acid. Acid used to preserve samples was submitted to the laboratory for preparation of an acid blank.

A field replicate sample was collected at one station each sampling day. *In situ* measurements and laboratory analyses were recorded on the Quality Assurance Field Replicates form.

A special project to measure diel fluctuations in dissolved oxygen concentration was conducted in August 1981. A USGS Minimonitor was installed at various locations for 2 to 4 days. Dissolved oxygen, temperature, and specific conductance were recorded hourly. Measurements were made in Fort Drum Marsh, Zig Zag Canal, South Mormon Outside Canal, and in the river at the exit from Lake Sawgrass and the entrance to Lake Washington.

In February 1983, a YSI model 56 dissolved oxygen monitor was installed on a stand in the marsh east of Bulldozer Canal. This monitor provided extended records of diel variation in dissolved oxygen and temperature.

For a more detailed description of sample collection techniques refer to the St. Johns River Water Management District Water Quality Monitoring Field Manual, July 1981. The laboratory quality control procedures are outlined in the St. Johns River Water Management District Laboratory Quality Assurance Program, October 1980.

Agricultural Pumpage

Information on the quality of agricultural pump discharges was obtained by Florida Institute of Technology (FIT) under a subcontract and by the District. The agricultural pump sites sampled by FIT were selected by the District based on location, land-uses served, and pump capacities (Figure 15, Table 15).

In the work performed by FIT, one gallon grab samples were taken at the pump, 250 meters upstream from the pump, 250 meters downstream from the pump, and at the confluence of the drainage canal associated with the pump and the river channel. The pump samples were taken as close to the discharge pipe as possible to minimize dilution effects from surrounding water. Grab samples at the other sites were collected two feet below the surface in the center of the channel. Upstream and downstream flow measurements were made, when possible, for estimates of pump discharge according to the method described earlier in this section. When this was not possible, due to pump flow directly into the marsh, subjective judgements on whether the pump was operating at full capacity, half capacity, etc. were made.

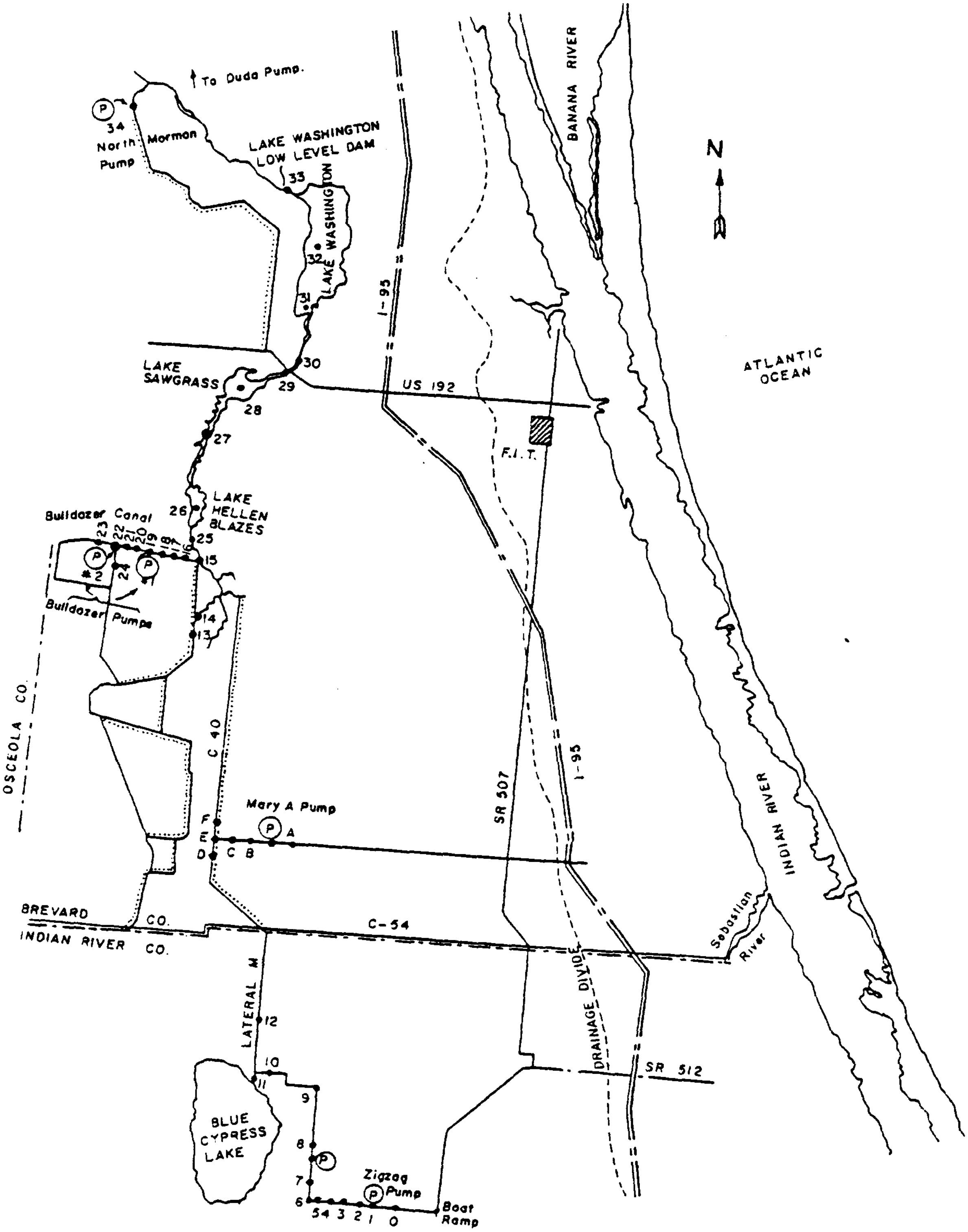


Figure 15. Locations of stations sampled by Florida Institute of Technology to determine the quality of agricultural pump dischaarges (Belanger et al., 1983).

Table 15. Characteristics of agricultural pumps selected as potential sample sites and the number of pump events sampled at each site by Florida Institute of Technology and the St. Johns River Water Mangement District.

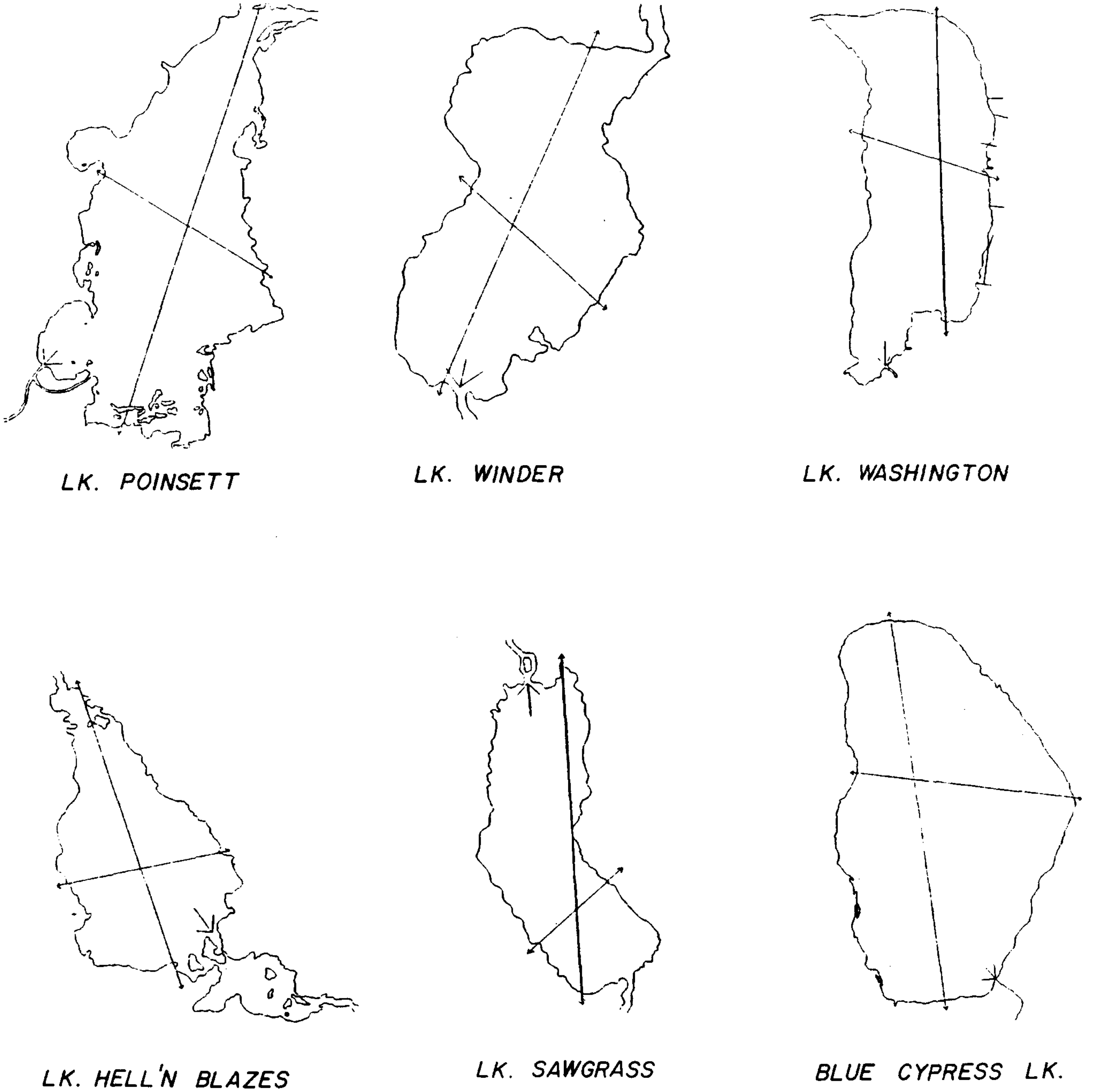
Pump	Capacity (GPM)	Owner	Land Use	Number of Pump Events FIT	SJRWMD
Zig-zag	150,000	Fellsmere Joint Ventures	Citrus	3	1
Lateral-M	100,000	Fellsmere Joint Ventures	Row Crops/Pasture	0	2
Mary-A	120,000	Fellsmere Joint	Row Crops	3	0
Bulldozer #1	16,000	Deseret Ranch	Pasture	1	0
Bulldozer #2	-	Deseret Ranch	Citrus	6	0
N. Morman	12,000	Deseret Ranch	Pasture	0	1
Duda	57,000	Duda Ranch	Pasture	0	0

Chemical analysis of all samples analyzed by FIT were done in accordance with EPA recommended procedures as described in EPA Document EPA/600-4-79-020 (Table 14). One sample from each site was run in duplicate and an EPA reference sample was routinely run as well.

Pump samples obtained by District personnel were collected as part of the water quality monitoring program and were analyzed in accordance with the techniques previously indicated. Data obtained by FIT were previously reported by Belanger, van Vonderen and Carberry (1983).

Sedimentation

Sediment thickness and stratigraphy was sampled with a hand coring device similar to that described by Baker, Pugh and Kimball (1977). The corer consisted of a clear plastic tube 48 inches in length and two inches in diameter. Cores were obtained along radial, transverse and longitudinal transects within each lake (Figure 16). Three radial



LK. POINSETT

LK. WINDER

LK. WASHINGTON

LK. HELL'N BLAZES

LK. SAWGRASS

BLUE CYPRESS LK.

Figure 16. Locations of transects sampled by Florida Institute of Technology for sediments and depth (Belanger et al., 1983).

transects, each extending 150 m into the lake from the river's entrance, were sampled at 30 m intervals within each lake (total of 16 cores per lake). Cores were obtained along longitudinal transects at intervals equal to 10 percent of the total length (11 cores) and along transverse transects at intervals equal to 20 percent of the length (6 cores). Sample locations were determined along radial transects and along the longitudinal and transverse transects of the smaller lakes by running the boat at a constant throttle speed for a predetermined length of time while siting across the lake to landmarks to stay on line. On lakes Blue Cypress and Poinsett, a calibrated rate meter automatically selected sample locations on longitudinal and transverse transects while compass headings kept the boat on the transect line. Each core was examined in the field to determine total organic sediment thickness and the thickness and nature of any other visible strata. Bathymetric profiles along each longitudinal and transverse transect were also obtained using a Ratheon DE-719B fathometer set on its lowest depth range (0 - 55 feet) (Appendix A). This work was subcontracted to Florida Institute of Technology and was performed and previously reported on by Belanger, van Vonderen and Carberry (1983).

In order to provide data on historical changes in sedimentation rates, personnel working in the laboratory of E. S. Deevey at the University of Florida were contacted to discuss the possibility of dating sediment strata using Pb-210 activity levels to determine the age of strata. In April 1982 two cores were obtained from each of lakes Blue Cypress, Hell 'n Blazes, and Washington. The collection and analysis of these cores was directed by Dr. Mike Binford. Due to time and man-power constraints, only the cores from Lake Hell 'n Blazes were analyzed.

The two cores analyzed were obtained about 1 m apart near the center of Lake Hell'n Blazes. They were sectioned at 5 cm intervals for determination of gross chemical composition (core 1) and levels of Pb-210 activity (core 2). Dry mass, determined by desiccation to constant weight at 110 C; levels of carbonate, indicated by weight loss upon combustion at 990 C; and levels of total organic material, indicated by weight loss during combustion at 550 C; were determined for core 1. Activity of Pb-210 was determined on sections of core 2 according to the technique of Eakins and Morrison (1978).

Floodplain Vegetation and Hydrology

In order to evaluate temporal changes in the vegetation of the upper St. Johns basin, study sites were chosen within four reaches of the upper basin. The study sites were located within the uppermost 80 miles of the St. Johns basin. They were not chosen at random, but rather for their type of vegetation, and for their representation of a particular reach of the river. The site locations were: Site 1 - sections 13 and 14, T.25S., R.35E., marsh southeast of Lake Poinsett; Site 2 - sections 23 and 24, T.28S., R.35E., marsh northeast of Lake Hell'n Blazes; Site 3 - sections 25 and 36, T.30S., R.36E., marsh adjacent to and directly north of the Fellsmere Grade, and bounded on the east by canal C-40-extension; and Site 4 - sections 22 and 23, T.32S., R.36E., marsh southeast of Blue Cypress Lake (Figure 17). All of the sites were approximately 2 mi^2 in area.

Black and white aerial photographs were obtained for each study

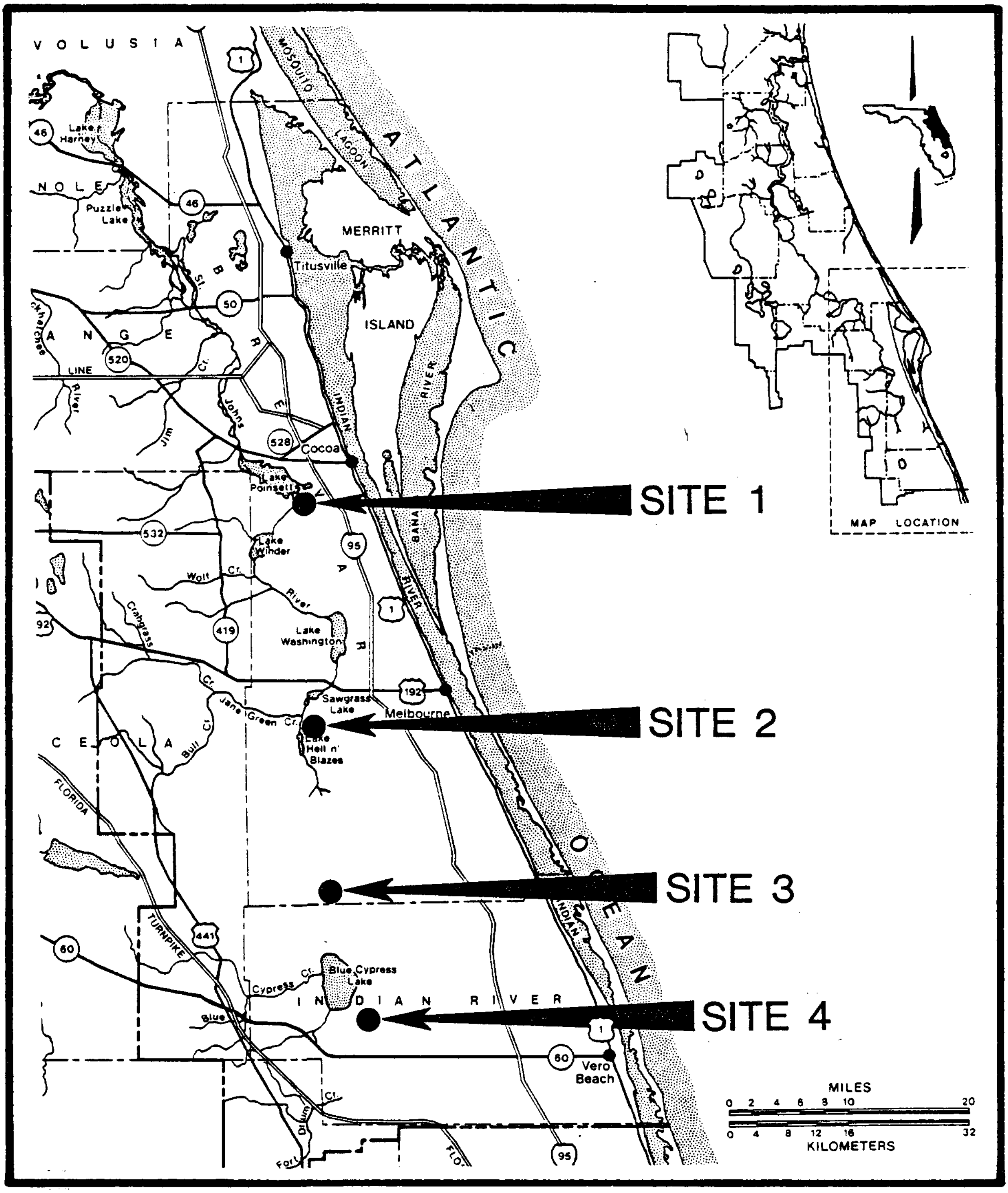


Figure 17. Locations of plots used to examine temporal changes in the distribution and composition of floodplain plant communities of the Upper St. Johns River.

site, for two time periods. Sites 1 through 3 were mapped from flights during 1943 (1':660') and 1980 (1':400'). The earliest flight for site 4 was 1951 (1':400'), and the covers mapped were compared to a Florida D.O.T. map (1':2000') prepared during 1981 from color IR photography. The 1981 map was reclassified to fit the cover classes used in this study. Plant communities and land features were delineated from these aerials, and assigned to one of seven cover classes (Table 16). The 1980 and 1981 aerials were ground-truthed and changes which occurred since the photography were incorporated in the mapping procedures.

The Environmental Systems Research Institute (ESRI), Geographical Information System (GIS) was used to digitize and assign numeric codings to the various map features. This information was then processed by GIS to calculate the areas for each cover type, calculate the area of each cover per contour interval, plot the cover maps, and present the data in tabular formats.

To provide background data for interpretation of vegetative changes, certain temporal trends in the basin were examined: 1) The pattern of basin development was investigated to establish a chronology of significant events in alteration of the floodplain. 2) Variation of surface water elevations in the marsh near Blue Cypress Lake, US 192, Lake Washington, and Lake Poinsett were analyzed to reveal temporal trends in hydrologic conditions. In this work, the District's records of daily surface water elevations for each area were used to determine ten-year moving averages for frequency of inundation (percent), hydroperiod (days), annual mean depth, annual maximum depth, annual minimum depth and annual range in depth for several marsh elevations.

3) Temporal trends in rainfall were examined using District rainfall records for Titusville, Fellsmere, Melbourne, and Vero Beach. The period of record for Titusville and Fellsmere was 1931-1982 (52 years) and for Melbourne and Vero Beach 1948-1982 (35 years). Thus, the datum used for the analyses, mean annual rainfall, was based on the average of two stations for 1931-1947 and of 4 stations from 1948-1982. Five-year, moving averages were used to accentuate long-term trends.

Table 16. Description of cover classes used for analysis of aerial photographs of selected sites in the floodplain of the Upper St. Johns River.

Cover class	Description
Low marsh	Dominated by maidencane (*Panicum hemitomon*), sawgrass (*Cladium jamaicense*), cattail (*Typha* spp.), and/or broadleaved herbs.
Low marsh/shrub	Low marsh with a conspicuous, but not dominant, shrub (Willow - *Salix caroliniana*, Red Maple - *Acer rubrum*, and/or Wax Myrtle - *Myrica cerifera*) component.
Shrub/low marsh	Low marsh with a dominant shrub component.
High marsh	Infrequently inundated areas of the floodplain dominated by Sand Cordgrass (*Spartina bakeri*), and unidentified grasses and largely lacking the more hydrophytic species of the low marsh.
High marsh/shrub	High marsh with a conspicuous, but not dominant, shrub component.
Trees	Areas highly dominated by cabbage palm (*Sabal palmetto*), Wax Myrtle (*Myrica cerifera*), and/or Red Maple (*Acer rubrum*).
Open water	Areas of lakes, rivers, ponds, and canals lacking conspicuous emergent macrophytes.

RESULTS

General Water Quality

The hydrologic condition of the basin varied widely over the sampling period. In November 1979, stages were high due to heavy rainfall in September (approximately 7.5 inches were associated with Hurricane David) (Figure 18). Stages declined for the next 13 - 24 months during which there was approximately a 30 inch rainfall deficit. Rainfall subsequently occurred at or near normal rates and stages rose rapidly. During the drought, lake elevations declined to recorded minima and the river became segmented into three sections separated by the Fellsmere Grade and the Lake Washington weir. When the elevation of Lake Washington was 14.5 ft. NGVD, it took approximately 32 days for water to travel from Bulldozer canal to the Lake Washington weir. At an elevation of 13.5 ft. NGVD, travel time over the same stretch of river was approximately 83 days. There was no flow over the weir from March through August 1981 and Blue Cypress lake was impounded from November 1980 until May 1982 when structure S1 was opened.

Broad spatial trends were evident for the major minerals and for total phosphorus. Concentrations of major minerals, and of chemical and physical indicators of the level of mineralization, increased from south to north. For example, mean chloride levels were 63 mg/l in the Blue Cypress Lake area, 180 mg/l between the Fellsmere Grade and Lake Washington weir, and 336 mg/l north of the weir (Table 17). A south to north gradient of increasing concentration was also observed for total dissolved solids, hardness, sulfate, magnesium, potassium, sodium, and specific conductance.

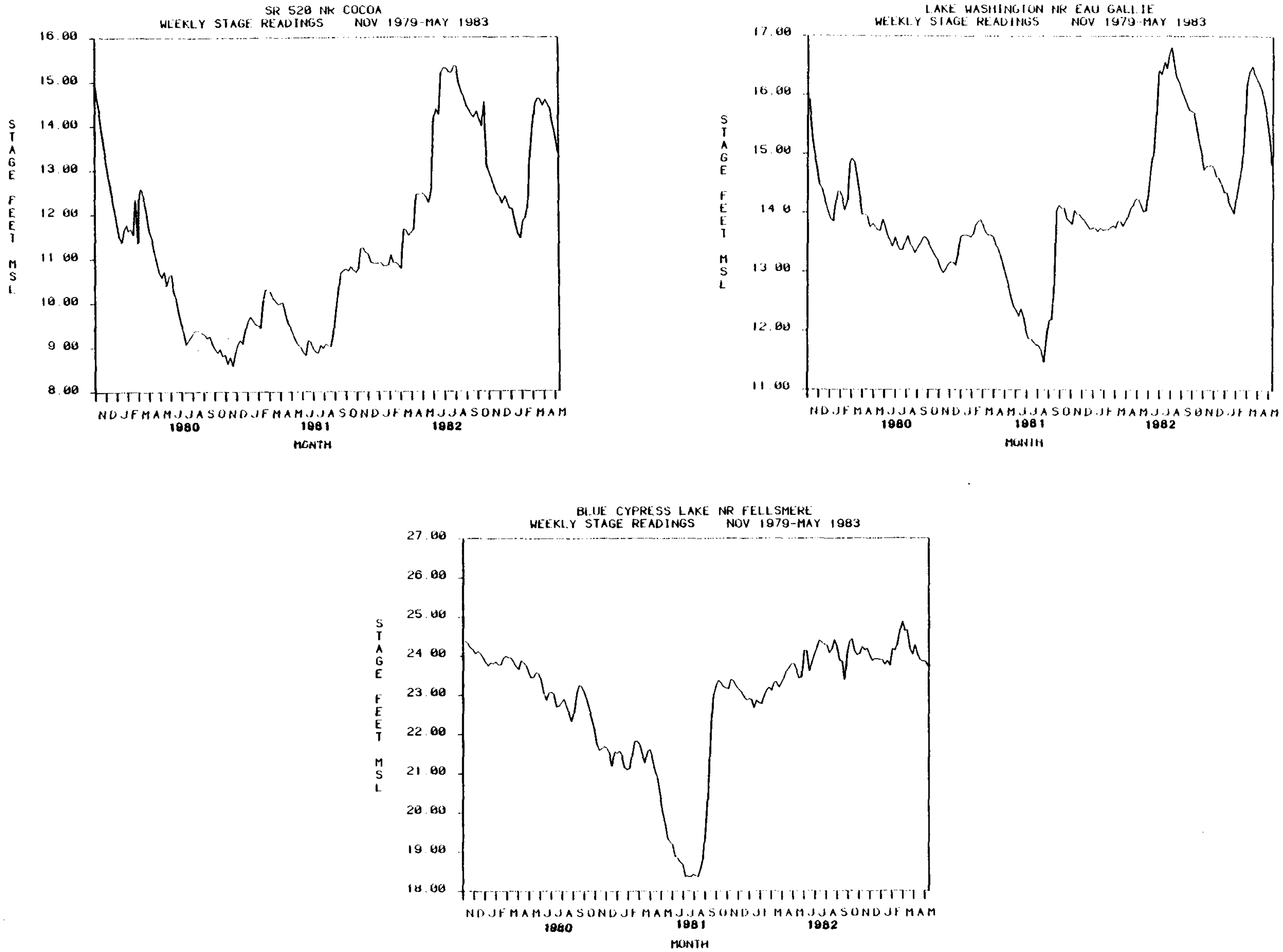


Figure 18. Variation in river stage over the sampling period of the study of general water quality at three locations.

Table 17. Mean concentrations of water chemistry constituents in three segments of the Upper St. Johns River.

Parameter	Area Blue Cypress Lake Area	 Fellsmere Grade to Lake Washington Weir	 Lake Washington Weir to SR 520
Water temp. (.C)	22.5	22.0	23.5
Turbidity (NTU)	2.4	1.4	4.4
Secchi Depth (in)	31	34	30
Conductivity (umhos/cm)	304	860	1380
Diss. Oxygen (mg/l)	4.7	5.8	6.3
pH	6.0	6.8	6.4
Color (cpu)	131	102	114
B.O.D. (mg/l)	2.6	1.8	2.7
Susp. Solids (mg/l)	6.5	5.4	16
Diss. Solids (mg/l)	267	610	880
Chloride (mg/l)	63	180	336
Sulfate (mg/l)	17	65	88
Alkalinity (mg/l)	46	120	76
Hardness (mg/l)	85	260	287
Magnesium (mg/l)	7.4	22	24
Calcium (mg/l)	23	65	59
Potassium (mg/l)	2.6	3.6	4.7
Sodium (mg/l)	32	85	172
Iron (ug/l)	267	147	300
Orthophosphate (mg P/l)	0.07	0.02	0.06
Total Phosphorus (mg P/l)	0.11	0.05	0.12
Ammonia - N (mg N/l)	0.06	0.10	0.28
Nitrate-Nitrite N (mg N/l)	0.08	0.05	0.03
TKN (mg N/l)	1.2	1.1	1.9
Chlorophyll-a,uncor.(ug/l)	14.8	9.1	24

Levels of total phosphorus were generally lowest in the segment of the river lying between the Fellsmere Grade and Lake Washington weir (Table 17; Figure 19). The mean concentration of total phosphorus was 0.11 mg/l south of the Fellsmere Grade and 0.12 mg/l north of the weir but only 0.04 mg/l in the intervening section of the river. This trend was also evident in lake levels of total phosphorus. The mean

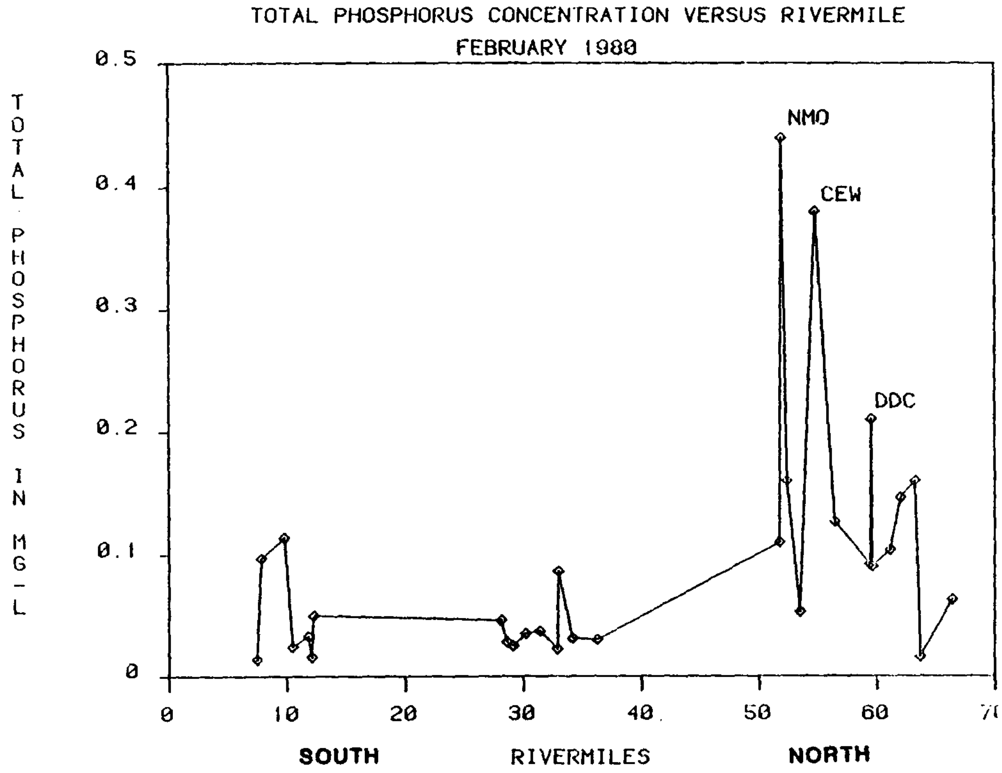


Figure 19. Variation in the concentration of total phosphorus along the length of the Upper St. Johns River (from south to north) during February 1980.

Table 18. Mean concentrations of water chemistry constituents in lakes of the Upper St. Johns River.

	Lake					
Parameter	Blue Cypress Lake	Lake Hell 'n Blazes	Lake Saw-Grass	Lake Wash-ington	Lake Winder	Lake Poinsett
Water Temp. (C)	24.5	22.0	22.0	22.0	23.0	24.5
Turbidity (NTU)	2.7	1.2	0.8	1.7	6.9	3.7
Conductivity (umhos/cm)	265	880	650	770	900	1600
Dissolved Oxygen (mg/l)	7.0	6.5	7.3	7.5	7.4	7.6
pH	6.4	7.2	7.1	7.5	7.4	7.1
Color (cpu)	135	110	125	106	118	109
B.O.D. (mg/l)	1.8	1.7	1.6	1.3	2.4	2.0
Suspended Solids (mg/l)	6.5	1.9	2.3	5.6	18	16
Dissolved Solids (mg/l)	200	585	535	550	530	810
Chloride (mg/l)	54	175	152	175	220	330
Sulfate (mg/l)	10.8	64	50	54	52	85
Alkalinity (mg/l)	28	120	99	85	65	67
Magnesium (mg/l)	4.8	22	20	18	17	21
Calcium (mg/l)	14.8	61	54	54	44	52
Potassium (mg/l)	2.2	3.8	3.5	3.6	3.6	4.5
Sodium (mg/l)	22.7	81	72	80	95	170
Iron (ug/l)	310	120	120	135	235	330
Orthophosphate (mg P/l)	0.07	0.02	0.02	0.02	0.05	0.04
Total phosphorus (mg P/l)	0.11	0.05	0.04	0.05	0.10	0.09
Ammonia-N (mg/l)	0.04	0.04	0.03	0.04	0.06	0.06
Nitrate-Nitrate N (mg N/l)	0.13	0.02	0.02	0.05	0.03	0.02
TKN (mg N/l)	0.8	1.0	1.2	1.3	1.3	1.6
Chlorophyll-a, uncorrected (ug/l)	6.5	5.3	7.5	3.7	11.0	18.0

concentration was 0.11 mg/l for Blue Cypress, 0.10 for Winder, and 0.09 for Poinsett but only 0.05 or less for Hell 'n Blazes, Sawgrass, and Washington (Table 18).

Smaller scale spatial variation, associated with the lakes, was found for dissolved oxygen and suspended solids. During periods of generally low dissolved oxygen, aeration of the water column in lakes caused outlet oxygen concentrations to exceed the concentrations at inlets. Because of this, the outlet of a lake typically had oxygen levels 32 percent higher than its inlet. As would be expected, the lakes also affected levels of suspended solids but in the shallow lakes north of the Grade these effects were different for large (Washington, Winder, Poinsett) and small (Hell 'n Blazes, Sawgrass) lakes. In Lake Hell 'n Blazes, suspended solids levels consistently decreased from the inlet (mean = 9.0 mg/l) to the outlet (mean = 2.3 mg/l) (Figure 20). Suspended solids also decreased, although to a lesser extent, across Lake Sawgrass, but in the larger lakes concentrations of suspended solids at outlets often exceeded concentrations at inlets. The smaller lakes also had lower mean concentrations of suspended solids at their outlets (means less than 2.3 mg/l) than the larger lakes (means of 9.4 - 18.2 mg/l). Loading rates cannot be calculated for the smaller lakes due to a lack of flow measurements, but, assuming the flow rate remains relatively constant across these lakes, it appears that they serve as settling basins for sediments. For one of the larger lakes, Lake Washington, a very small net input of suspended solids was calculated by Belanger et al. (1983). They also found, however, that outlet concentrations often exceeded inlet concentrations. It appears that due to turbulent mixing relatively little sediment is permanently deposited in the large lakes.

Certain areas had water quality notably different from that typical of the basin. Blue Cypress Lake, canal 40, and three forks run, typically had the highest levels of nitrate-nitrite nitrogen, with mean concentrations of 0.132, 0.17, and 0.14 mg N/l, respectively

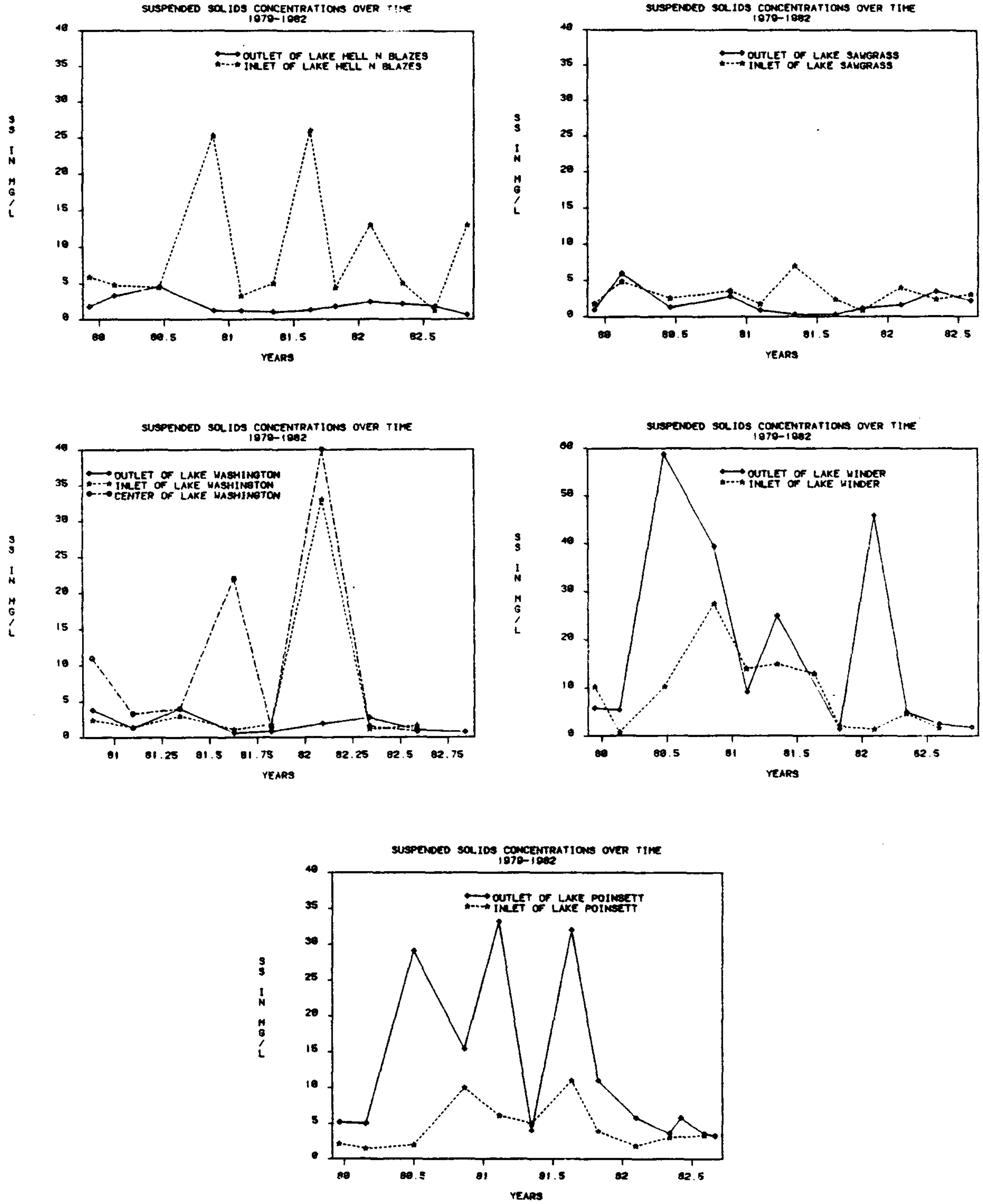


Figure 20. Variation in the concentration of suspended solids over the sampling period at the outlets and inlets of the lakes north of the Fellsmere Grade.

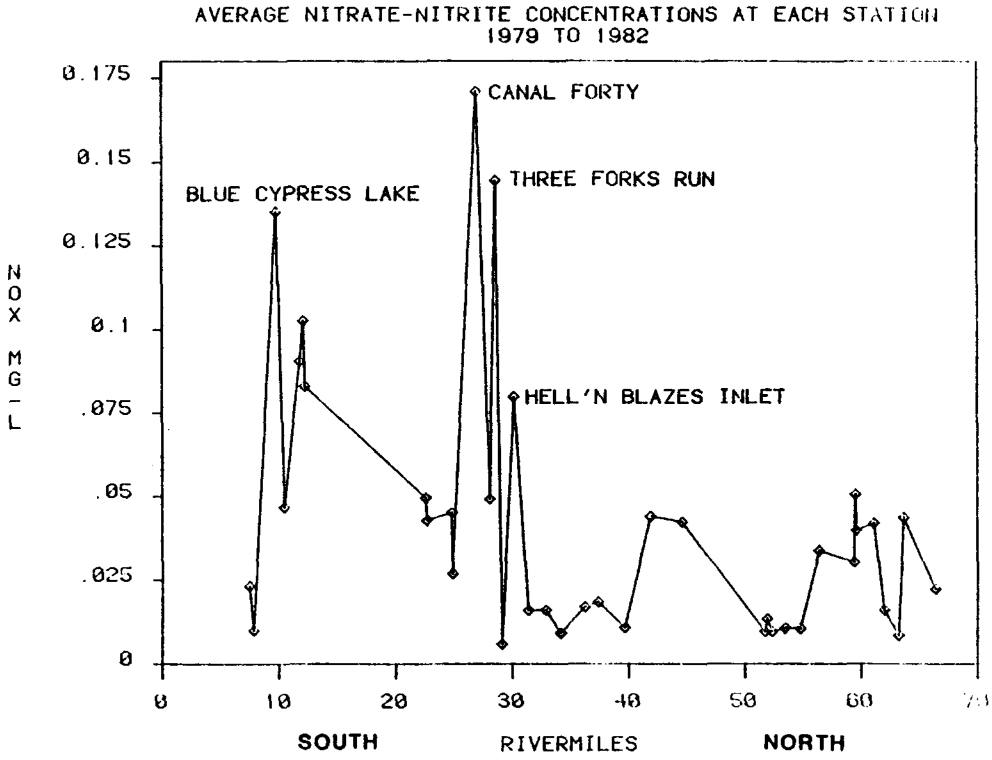


Figure 21. Variation in the mean concentration of nitrate-nitrite nitrogen along the length of the Upper St. Johns River.

(Figure 21). Mean values for other lakes ranged from 0.02 - 0.05 mg N/l and for other canals from <0.01 - 0.09 mg N/l.

An area which had generally poor water quality was Rockledge Creek, a tributary to Lake Poinsett which receives effluent from the Silver Pines sewage treatment plant in addition to urban and agricultural runoff. The creek had consistently high levels of total phosphorus with a mean concentration of 0.352 mg P/l and a range of 0.097 - 1.1 mg P/l. In addition, it had concentrations of chlorophyll-a generally higher than those found in other areas of the basin and often supported extensive algae blooms. The most severe of these blooms occurred during the drought in May - August 1981. At this time, the creek and the southeast shore of Lake Poinsett exhibited a green cast and noxious odor. _Anabena_ sp., a blue green alga capable of fixing atmospheric nitrogen, was the dominant species. Chlorophyll-a levels of 603 ug/l and phaeophytin-a levels of 0 ug/l were recorded on May 12, causing a low transparency (5 inches) and a high pH (9.9). Patches of foam, often associated with algae blooms, were noted and the concentration of dissolved oxygen was very high (18.8 mg/l). Nitrate-nitrite nitrogen levels were less than 0.01 mg N/l and the concentration of orthophosphate was 0.301 mg P/l. Samples collected in August still had high concentrations of chlorophyll-a (258 ug/l) and orthophosphate (0.18 mg P/l). Nitrate and phaeophytin-a levels were still negligible. Because of the low water conditions the bloom remained in the creek and along the shoreline and was not flushed into the lake.

Those parameters which exhibited systematic temporal variation were the major minerals and dissolved oxygen. Variation in levels of major minerals stemmed from their negative correlations with river or lake stage. For example, at US 192, the relationship between water elevation and chloride concentration was statistically significant and linear (mg Cl/l = 1193-75.7(elevation ft. NGVD), $F_{1,14}=75.6$, $p<0.001$, $R^2=0.84$) for elevations from 11 to 15 ft. NGVD (Figure 22). Thus, mineral concentrations increased during the drought. Due to interaction of the effects of river stage and the downstream gradient in mineral concentration, the impact of the drought on mineral concentrations was most intense at the downstream stations. While chloride concentration in Blue Cypress Lake doubled from November 1979 to August 1981, concentration in lakes Sawgrass and Poinsett increased by factors of 4 and 10, respectively (Figure 23). When river stage rose in response to increased rainfall in late summer 1981, chloride concentrations fell rapidly north of the Fellsmere Grade. In lakes Poinsett and Sawgrass, concentrations fell approximately 900 and 200 mg/l, respectively, to levels lower than those recorded in November 1979 following Hurricane David. In Blue Cypress Lake, however, levels increased through May 1982 despite above-normal rainfall and stages. Structure S1 was opened in May 1982 and by August chloride level in the lake had dropped to 63 mg/l, a level twice that observed following Hurricane David.

In order to more rigorously examine the relationship between chloride levels and river stage, additional data on chloride levels in Lake Washington, collected by personnel of the Lake Washington Water Treatment Plant, were obtained. These data indicate that a highly significant, log-linear relationship ($\log_{10}$ (mg Cl/l) = 3.32-0.103(stage in ft. NGVD), $F_{1,294}=243$, $p<0.001$, $R^2=46.2$) between stage and chloride concentration holds over a wide range in stage (Figure 24). These data

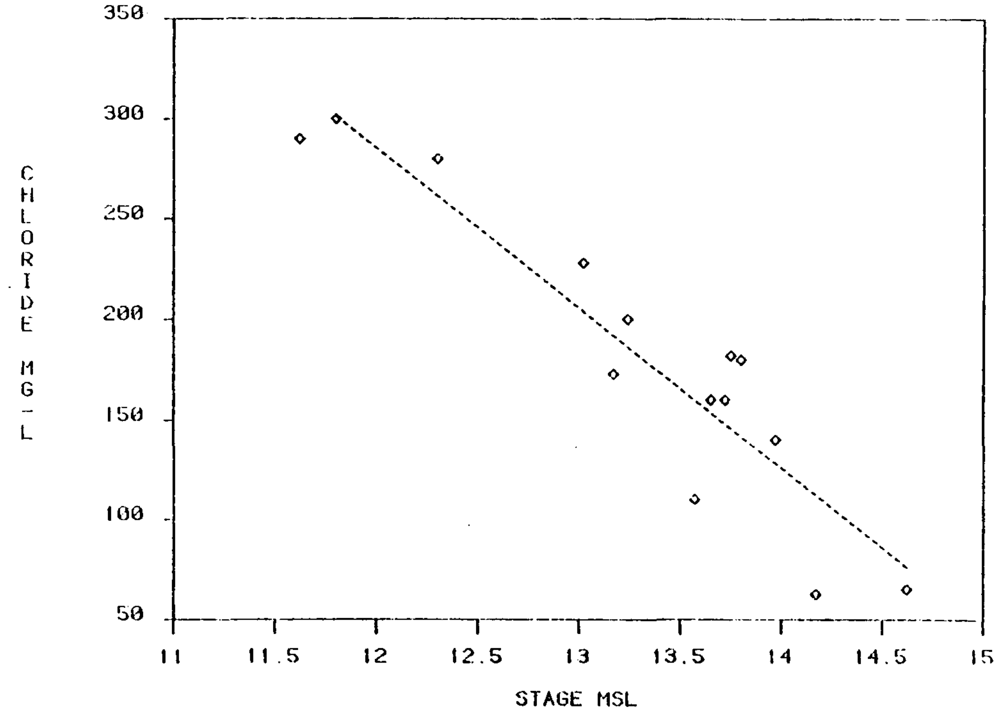


Figure 22. The relationship between river elevation and chloride concentration at U.S. 192 when water levels are falling.

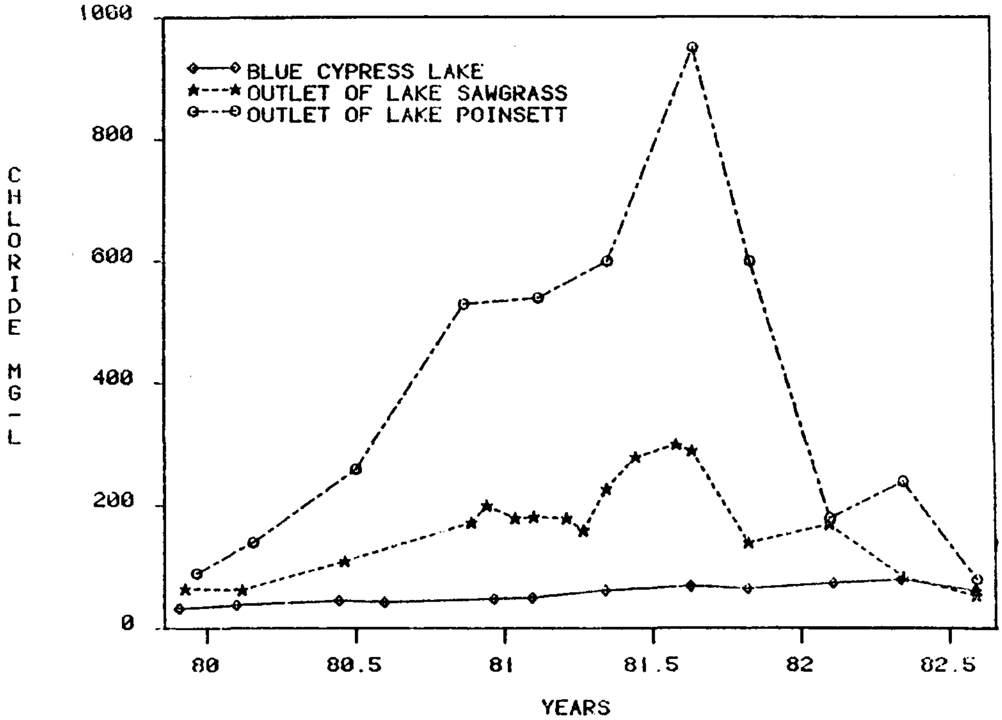


Figure 23. Variation in chloride concentration over the sampling period in the three segments of the Upper St. Johns River.

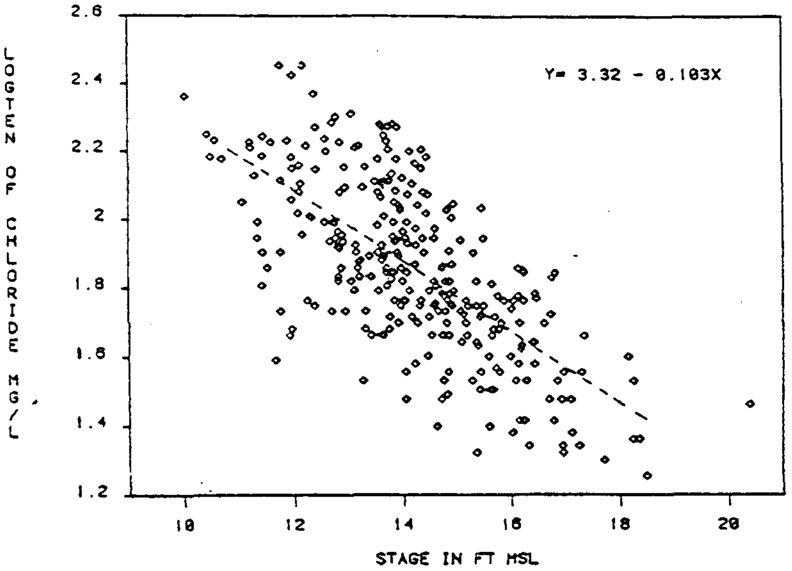


Figure 24. The relationship between chloride concentration and lake elevation at Lake Washington from 1952 - 1983.

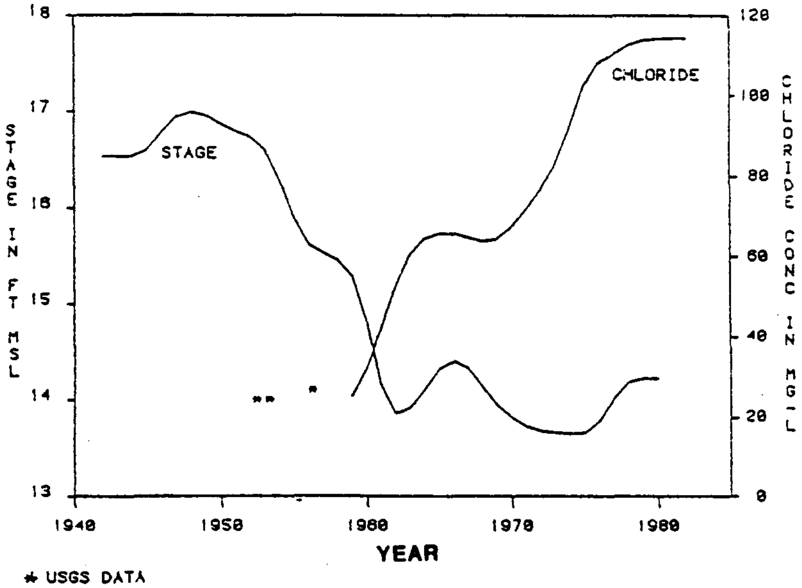


Figure 25. Variation in annual mean concentration of chloride and annual mean lake elevation at Lake Washington from 1940 - 1982. Time series were smoothed using running medians (Velleman and Hoaglin, 1981).

also indicate that the mean level of chloride has risen since 1959 (Figure 25).

Dissolved oxygen concentrations varied seasonally with high levels occurring in winter and low levels in summer (Figure 26). The seasonal cycle probably resulted from seasonal variation in temperature and river stage. Dissolved oxygen also responded to the rise in stage following the drought. As stages rose, oxygen levels declined and declined most severely in those areas where previously dry wetland was once again inundated. For example, in August 1982 stages increased by 2 feet or more and dissolved oxygen levels were very low (mean = 0.95 mg/l) compared to the levels of May (mean = 3.8 mg/l). Furthermore, at that time, levels exceeded 1 mg/l only in lakes Washington and Poinsett.

As expected, dissolved oxygen levels also exhibited diel variation (Figure 27). Ft. Drum Marsh had a relatively large diel fluctuation in dissolved oxygen, ranging from approximately 2 mg/l each morning to 6 mg/l in the afternoon. This increase was nearly twice that observed at the river's entrance to Lake Washington (5-7 mg/l) and at Zig-zag canal (<1-2 mg/l). At Lake Sawgrass, the diel range (5 mg/l) was similar to that of Ft. Drum marsh although concentrations were generally higher (4-9 mg/l). Examination of morning, noon, and afternoon oxygen concentrations for all sites from 1979 - 1980 indicates that diel variation in the dissolved oxygen concentration of marsh sites was typically twice that for canal, river, lake, or hardwood swamp sites. More recent data obtained by the YSI recorder station indicates that in the summer diel variation in oxygen concentration can result in concentrations very near 0 mg/l for a large portion of each day (Figure 28). Perhaps even more importantly, day to day variation in

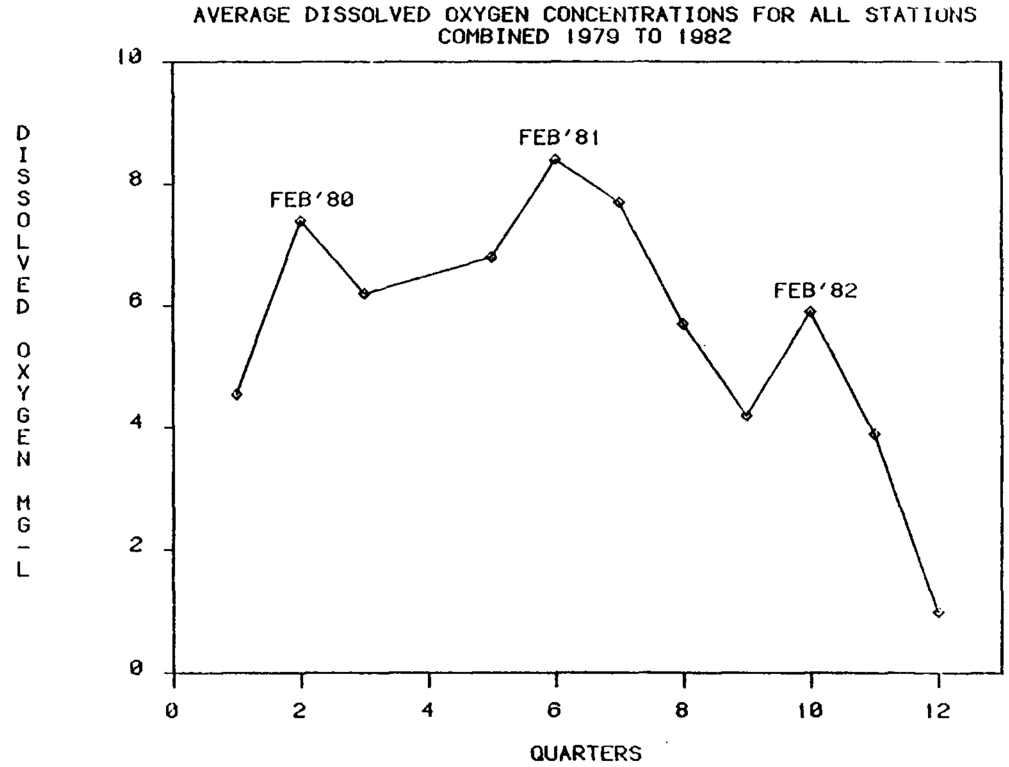


Figure 26. Variation in the mean concentration of dissolved oxygen over the sampling period.

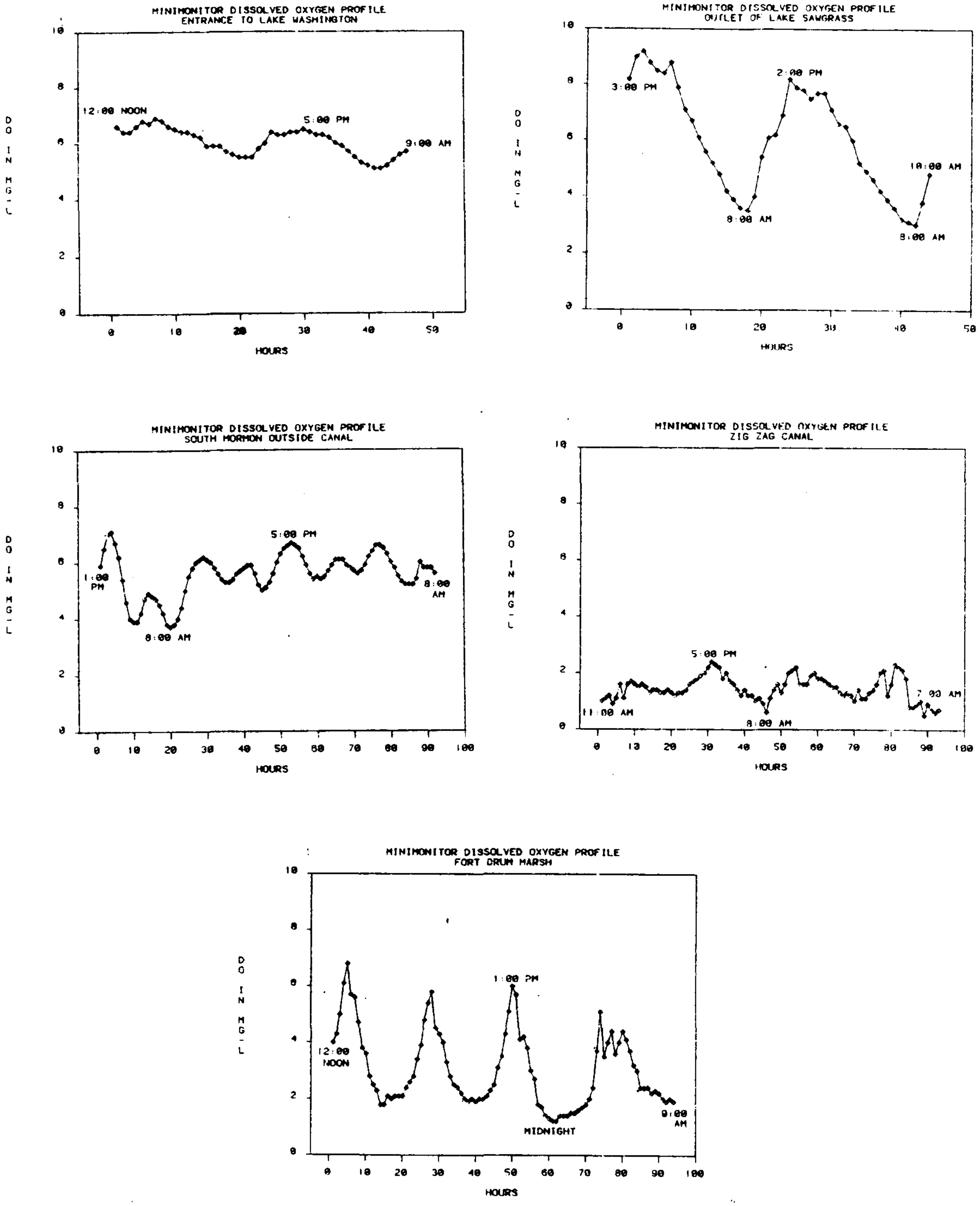


Figure 27. Diel variation in the concentration of dissolved oxygen at different locations in the Upper Basin in August, 1981.

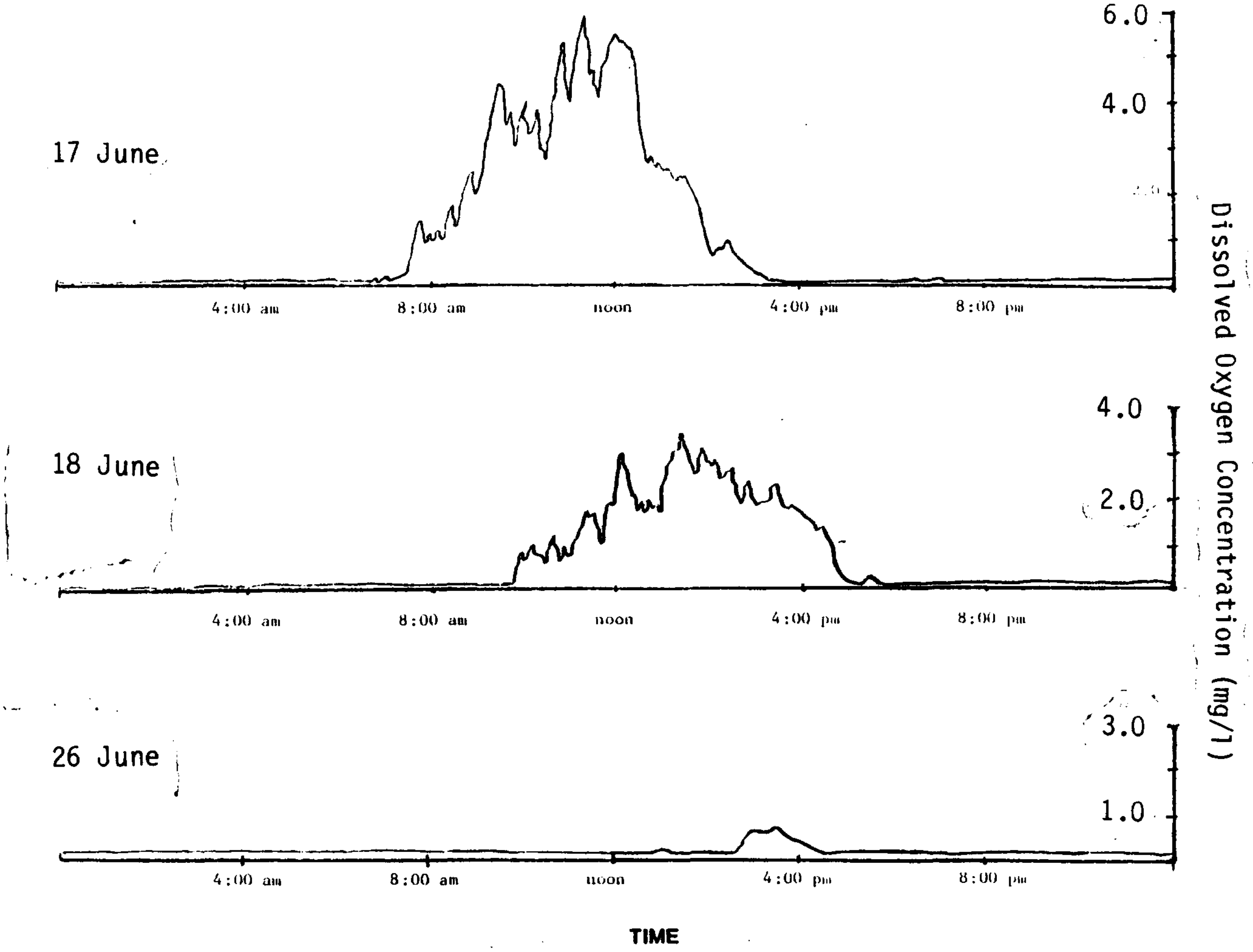


Figure 28. Diel variation in the concentration of dissolved oxygen in the marsh near Bulldozer Canal in June 1983.

oxygen concentrations can be such that levels are near zero for several consecutive days. Such large diel and daily fluctuations in oxygen concentrations are probably of greater ecological importance than seasonal fluctuations.

A significant portion of the variation in water quality can be more aptly attributed to differences in the nature of the water bodies sampled than to spatial or temporal trends. Concentrations of major minerals were highest in canals and lowest in natural tributaries (Table 19). The mean concentration of chloride, for example, ranged from 240 mg/l for canals to 34 mg/l for tributaries. Mineral concentrations were also higher in altered (mean [Cl]=162 mg/l) than in natural (Jane Green Marsh, JGM, mean [Cl]=45 mg/l) marsh. There were also differences between canals and tributaries and between altered and unaltered marshes in concentrations of total dissolved solids, hardness, sulfate, alkalinity, and conductivity.

Dissolved oxygen levels were typically highest in sections of the river without nearby influences (mean=7.4 mg/l) and in lakes (mean=7.2 mg/l) and lowest in altered marsh (mean=3.8 mg/l). During the high water conditions following Hurricane David, deep canals stratified. Oxygen levels at this time were lowest in Zigzag Canal, ranging from 1.3 mg/l at the surface to 0.5 mg/l at the bottom. Levels in Lateral-M Canal were higher at the surface but still dropped to 0.5 mg/l at the bottom. Under the same conditions, Blue Cypress Lake was well mixed with oxygen concentrations ranging from 5.0-5.4 mg/l.

Mean total phosphorus levels were highest in altered marsh (0.11 mg/l) and in canals (0.10 mg/l) and lowest for river stations without nearby influences (0.04 mg/l). Canals exhibited very high levels

Table 19. Mean concentrations of water chemistry consituents for all stations within each of seven types of water body found in the Upper St. Johns River Basin.

	Water Body Type						
Parameter	River	River*	Tributary	Lake	Canal	Marsh	JGM**
Water Temp. (.C)	22.7	22.0	21.0	23.5	22.0	22.0	17.0
Turbidity (NTU)	1.5	1.6	1.2	3.0	3.3	3.3	1.0
Secchi Depth (in)	38	41	25	31	30	20	10
Conductivity (umhos/cm)	1100	1000	151	860	1150	760	24
Diss. Oxygen (mg/l)	6.0	7.4	3.0	7.2	5.0	3.8	4.7
pH	6.9	7.0	5.8	7.0	6.7	5.6	5.7
Color (cpu)	105	81	173	111	95	174	210
B.O.D. (mg/l)	1.9	1.8	2.1	1.8	2.3	4.1	3.8
Susp. Solids (mg/l)	5.9	6.8	2.7	10	12	21	6.2
Diss. Solids (mg/l)	670	700	170	580	730	540	180
Chloride (mg/l)	225	240	34	201	240	162	45
Sulfate (mg/l)	65	69	5.3	58	76	67	8.4
Alkalinity (mg/l)	105	110	28	80	117	39	20
Hardness (mg/l)	255	270	54	220	285	170	83
Magnesium (mg/l)	21	23	6.3	19	24	11	2.3
Calcium (mg/l)	60	60	22	51	69	32	15
Potassium (mg/l)	3.9	3.8	1.3	3.9	4.2	2.3	0.7
Sodium (mg/l)	115	105	44	97	113	82	17
Iron (ug/l)	140	112	450	197	225	310	--
Orthophosphate (mg P/l)	0.03	0.02	0.05	0.03	0.05	0.06	0.03
Tot. Phosphorus (mg P/l)	0.06	0.04	0.08	0.07	0.10	0.11	0.08
Ammonia-N (mg N/l)	0.06	0.06	0.03	0.05	0.09	0.14	0.06
Nitrate-Nitrite N (mg N/l)	0.04	0.03	0.04	0.04	0.05	0.03	0.03
TKN (mg/l)	1.2	1.1	1.1	1.3	1.1	2.3	1.5
Chlorophyll-a, uncorrected,(ug/l)	7.7	8.8	3.2	10	19	39	20
Number of Obs.	89-251	31-129	7-40	40-218	77-369	15-75	3-9

*River Stations excluding other nearby influences

**Marsh (historical) baseline station

following storm events. In February 1980, for example, North Morman Outside Canal had 0.44 mg P/l, approximately 8 times higher than the background level (Figure 19).

The mean concentration of total nitrogen was also highest in altered marsh (2.33 mg N/l) but varied little among other water body types (1.13 - 1.53 mg N/l). Mean chlorophyll-a concentration was highest in altered marsh and lowest in tributaries (3.2 ug/l), probably because of shading.

Table 20. Mean concentrations of various chemical and physical indicators of trophic state for lakes of the Upper St. Johns River. Trophic state indices (TSIs) are those of Huber et al., 1982.

	Lake					
Parameter	Blue Cypress	Hell 'n Blazes	Sawgrass	Washington	Winder	Poinsett
TKN (mg N/l)	1.40	1.30	1.35	1.51	1.45	1.47
Ammonia N (mg N/l)	0.04	0.04	0.03	0.04	0.06	0.06
Nitrate-Nitrite N (mg N/l)	0.13	0.02	0.02	0.05	0.03	0.02
Total N (mg N/l)	1.53	1.32	1.37	1.56	1.48	1.49
Orthophosphate (mg P/l)	0.07	0.02	0.02	0.02	0.05	0.04
Total P (mg P/l)	0.11	0.05	0.04	0.05	0.10	0.09
Secchi Depth (m)	0.79	0.81	1.04	1.07	0.56	0.64
Chlorophyll-a, uncor. (ug/l)	6.5	5.3	7.5	3.7	11.0	18.0
Inorg. N/Total N	0.11	0.05	0.04	0.06	0.06	0.05
Inorg. N/Inorg. P	2.43	3.00	2.50	4.50	1.80	2.00
Total N/Total P	13.91	26.40	34.25	31.20	14.80	16.56
TSI(TN)	64.42	61.50	-	-	63.80	63.76
TSI(TP)	69.03	68.52	63.26	68.52	67.26	65.30
TSI(NUTR)	66.73	65.01	63.26	68.52	65.53	64.53
TSI(SD)	67.19	66.21	58.79	58.05	77.45	73.62
TSI(CHA)	43.75	40.82	45.81	35.64	51.33	58.42
TSI(AVG)	59.22	57.35	55.95	54.07	64.77	65.52

*Determined from data obtained by the District, the Florida Game and Freshwater Fish Commission, Brevard County, Florida Institute of Technology, and the Lake Washington Technical Advisory Committee.

Trophic state indices (TSI's) for total nitrogen (TSI(TN)) and phosphorus (TSI(TP)) indicate that all the lakes are eutrophic according to the nomenclature of Kratzer and Brezonik (1981) (Tables 20 and 21). The ratio of total nitrogen to total phosphorus for lakes Blue Cypress, Hell 'n Blazes, Winder and Poinsett suggests co-limitation of primary production while the relatively high ratio (>30) for lakes Sawgrass and Washington indicates phosphorus limitation (Huber et al., 1982). Although nutrient levels indicate eutrophy, the TSIs for chlorophyll-a (TSI(CHA)) suggest that phytoplankton productivity lies within the mesotrophic range for all lakes except Washington (oligotrophic).

Table 21. The association between numerical values of trophic state indices and trophic state nomenclature proposed by Kratzer and Brezonik, 1981.

TSI	Trophic State
0 - 19	Ultraoligotrophic
20 - 39	Oligotrophic
40 - 59	Mesotrophic
60 - 79	Eutrophic
80 -100	Hypereutrophic

Agricultural Pumpage

Water quality varied considerably among pumps (Table 22) but all pumps typically discharged water of poorer quality than that of the receiving water body (Tables 23 and 24). The highest concentrations of nutrients, particularly nitrogen, were associated with the most intense land uses such as row crops. Mary-A pump, which drained land recently converted from pasture to row crops (Table 15), discharged water of

Table 22. Mean concentrations of water chemistry constituents in discharges of agricultural pumps in the Upper St. Johns River Basin.

	Pump					
Parameter	Zig-zag	Lateral-M	Mary-A	Bulldozer #2	Bulldozer #1	N. Mormon
Dissolved Oxygen (mg/l)	2.9	5.8	3.3	1.7	4.2	4.0
B.O.D. (mg/l)	2.3	4.1	5.5	2.1	4.2	3.6
Color (cpu)	306	180	525	221	472	240
Susp. Solids (mg/l)	8.5	20.5	84	10.6	10.0	20
Turbidity (NTU)	4.4	7.3	18.1	7.0	14.0	1.8
Chloride (mg/l)	106	137	88	127	21	120
Sulfate (mg/l)	71	106	28	55	9.3	76
Total Phosphorus (mg P/l)	0.14	0.13	0.79	0.15	0.14	0.18
Orthophosphate (mg P/l)	0.10	0.04	0.52	0.08	0.06	0.06
TKN (mg/l)	2.4	-	3.9	1.8	2.7	-
Ammonia (mg/l)	0.15	-	0.31	0.09	0.19	-
Nitrate-Nitrite N (mg/l)	0.60	0.10	0.48	0.04	<0.02	<0.01
Chlorophyll-a (ug/l)	4.2	8.5	20	8.5	10.9	24

poorer quality than any other pump. Total phosphorus concentrations in Mary-A discharge averaged 0.79 mg P/l, over ten times the average concentration of the total phosphorus in the receiving water body, Canal Forty. Concentrations exceeded 1 mg P/l in two of the three sample events. Average total phosphorus concentrations for other pumps ranged from 0.13 - 0.18 mg P/l while their receiving waters averaged less than 0.10 mg P/l. Approximately 60 - 70 percent of the total phosphorus discharged by the pumps was in a dissolved form.

Mary-A discharge also had high levels of nitrogen with average TKN of 3.9 mg N/l, ammonia of 0.31 mg N/l, and nitrate-nitrite of 0.48 mg N/l. High levels of nitrate-nitrite nitrogen were also discharged by

Zig-zag pump (mean=0.60 mg N/1). Seventy to eighty percent of the TKN discharged by the pumps was dissolved.

Table 23. Mean concentrations of water chemistry constituents in the receiving waters of agricultural pumps in the Upper St. Johns River Basin.

	Receiving Water Body				
Parameter	Zig-zag Canal	Lateral-M Canal	Canal Forty	Bulldozer Canal	N.Mormon Canal
Dissolved Oxygen (mg/l)	3.9	5.1	4.6	6.5	4.7
B.O.D. (mg/l)	2.2	2.2	1.9	2.8	3.3
Color (cpu)	117	126	119	57	101
Suspended Solids (mg/l)	7.2	2.8	3.1	5.2	19.1
Turbidity (NTU)	2.4	1.7	1.6	2.0	5.4
Chloride (mg/l)	76	68	148	171	226
Sulfate (mg/l)	23	18	37	65	58
Total Phosphorus (mg P/l)	0.08	0.09	0.07	0.05	0.17
Orthophosphate (mg P/l)	0.04	0.05	0.02	0.02	0.10
TKN (mg/l)	1.0	-	1.4	0.70	-
Ammonia Nitrogen (mg/l)	0.06	-	0.14	0.04	-
Nitrate-Nitrite N (mg/l)	0.07	0.10	0.14	0.02	0.01
Chlorohyll-a (ug/l)	7.6	9.9	7.6	9.8	35.0
No. of Obs.	14	13	11	12	12

Levels of color were also highest in Mary-A discharge (mean = 525 CPU) but were elevated in other pumps as well (means of 180 - 472 CPU) relative to background concentrations (means of 56 - 126 CPU). Concentrations of suspended solids, turbidity and BOD were typically higher in pumped discharge than in receiving waters. For Mary-A, concentrations of suspended solids exceeded background levels by 81 mg/l on average. Despite the aeration provided by the pumps, dissolved oxygen levels were typically lower in discharges than in receiving waters.

Table 24. Difference between mean concentrations of water chemistry constituents in pumped discharges and their receiving waters (difference = pump - receiving) in the Upper St. Johns River Basin.

	Pump					
Parameter	Zig-zag	Lateral-M	Mary A	Bulldozer #2	Bulldozer #3	N.Mormon
Diss. Oxygen (mg/l)	-1.0	0.7	-1.3	-4.8	-2.3	-0.7
B.O.D. (mg/l)	0.1	1.9	3.6	-0.7	1.4	0.3
Color (cpu)	189	54	406	164	415	139
Susp. Solids (mg/l)	1.3	17.7	80.9	5.4	4.8	1.0
Turbidity (NTU)	2.0	5.6	16.5	5.0	12.0	-3.6
Chloride (mg/l)	30	69	-60	-44	-150	-106
Sulfate (mg/l)	48	88	- 9	-10	55.7	18
Total Phosphorus (mg P/l)	0.06	0.04	0.72	0.10	0.09	0.01
Orthophosphate (mg P/l)	0.06	-0.01	0.50	0.06	0.04	-0.04
TKN (mg/l)	1.4	-	1.5	1.1	2.0	-
Ammonia (mg/l)	0.09	-	0.17	0.05	0.15	-
Nitrate-Nitrite N (mg/l)	0.53	0.00	0.34	0.02	0.00	0.00
Chlorophyll-a (ug/l)	-3.4	-1.4	12.4	-1.4	1.1	-11.0
Canal	Zig-zag	Lat.-M	C-40	Bulldozer	Bulldozer	N. Mormon

Loading rates for total nitrogen and phosphorus were calculated for Zig-zag, Mary-A, and Bulldozer pumps (Figure 29). The rates for Mary-A were very high with maxima of 2995 and 681 kg/day for nitrogen and phosphorus, respectively. By comparison, the phosphorus loading rate for US 192, one week earlier, was 506 kg/day. The Bulldozer pump had the lowest loading rates with maxima of 238 kg/day for nitrogen and 12 kg/day for phosphorus.

Information on the temporal and spatial variation of water quality during a pump event was obtained for Zig-zag pump. Nutrient concentrations tended to decrease downstream of the discharge point (Figure 30). Generally, both ammonia and nitrate-nitrite nitrogen decreased but decreases in ammonia were sometimes associated with

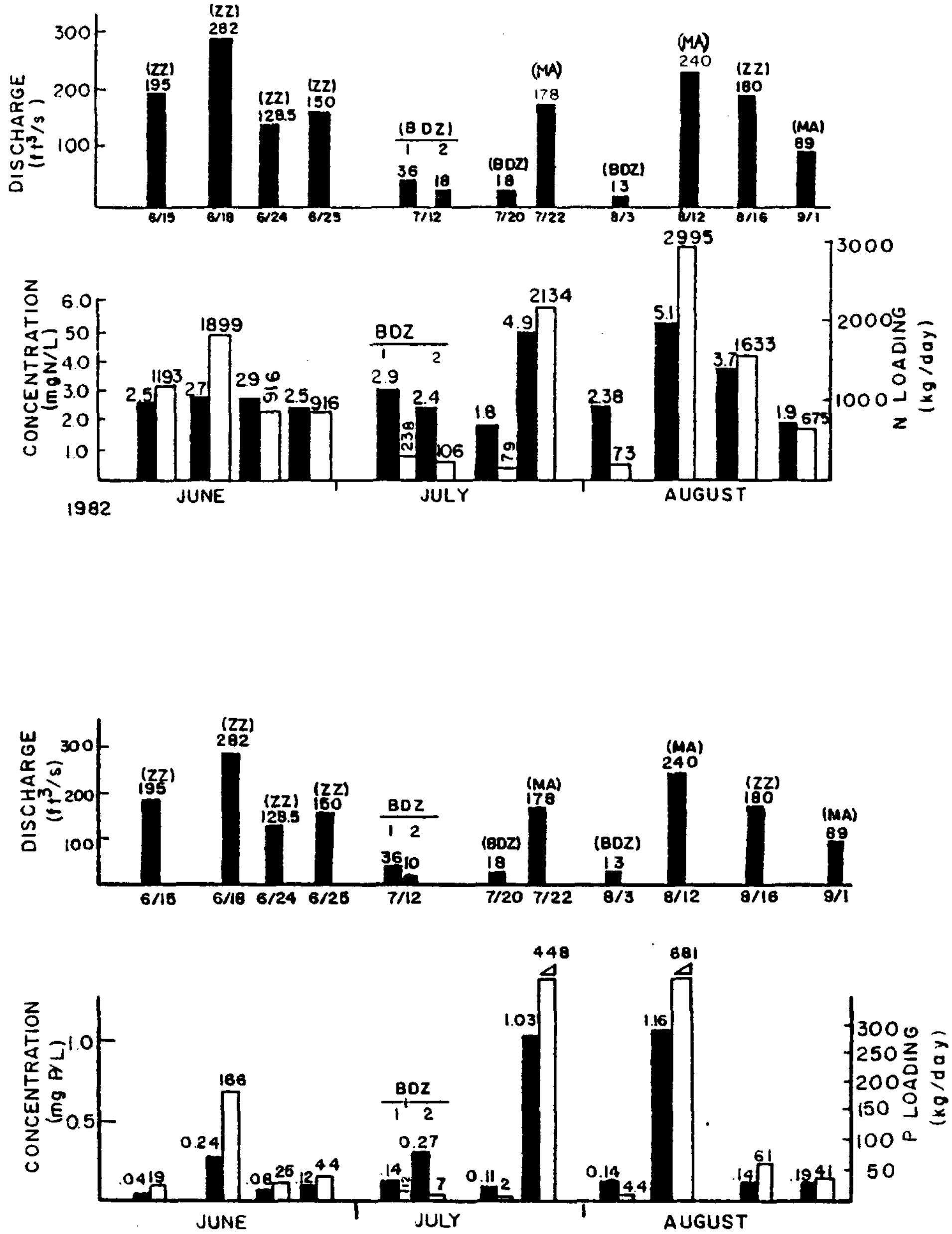


Figure 29. Concentrations and loading rates of nitrogen and phosphorus for the discharges of agricultural pumps. ZZ = Zigzag pump, BDZ = Bulldozer pump, MA = Mary-A pump. Open bars indicate loading rates (from Belanger et al., 1983).

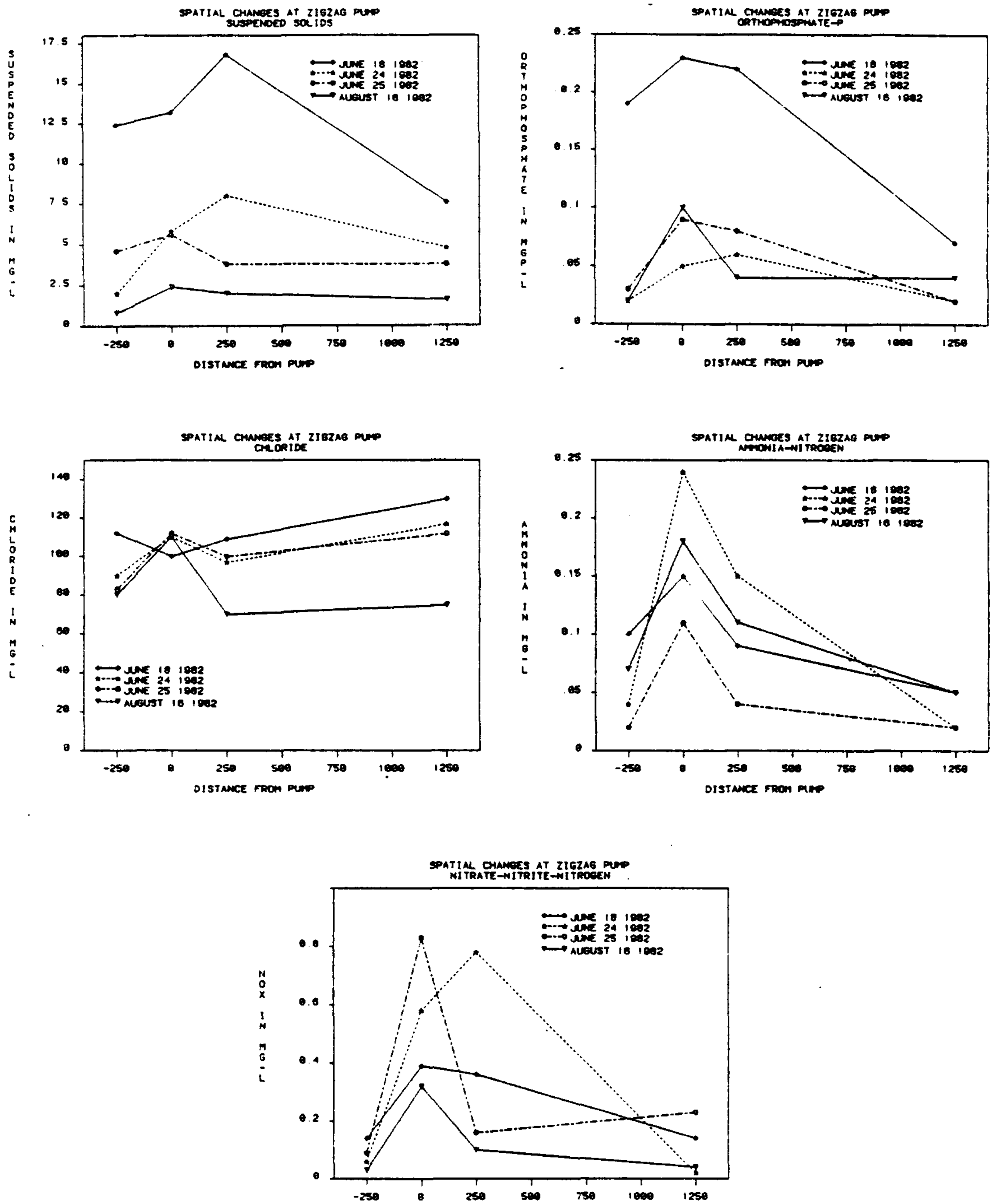


Figure 30. Variation in the concentrations of selected water chemistry constituents in agricultural pump discharges with distance (in meters) from the pump. Negative distance indicates distance upstream of the pump.

increases in nitrate-nitrite nitrogen. The decline in nitrogen levels with distance from the source was apparently not caused by dilution because the concentrations of conservative constituents, such as chloride, remained constant. Likewise, the decline in orthophosphate concentration was apparently not due to a settling of particulate phosphorus in that suspended sediment levels remained the same or increased downstream of the discharge. Thus, the spatial pattern in nutrient concentration was, apparently, biologically mediated.

During one pump event (June 24-25), samples were collected immediately after the pumps were turned on and at 15 and 24 hours thereafter. The concentrations of chlorides, total dissolved solids and color changed little over the sampling period (Table 25). Total phosphorus and nitrate-nitrite nitrogen concentrations, however, approximately doubled, rising from 0.08 to 0.17 mg P/l and from 0.58 to 1.08 mg N/l, respectively. The increase in levels of nitrate-nitrite nitrogen may have been partly due to nitrification since ammonia nitrogen levels declined from 0.24 to 0.07 mg N/l while dissolved oxygen concentration increased from 1.2 to 4.6 mg/l.

To place the quality of agricultural discharges in perspective it is necessary to consider the quality of other water body types. In this context, it is apparent that the concentrations of nutrients in discharges were very high (Table 26). The mean concentrations of total phosphorus (0.26 mg P/l), orthophosphate (0.16 mg P/l), nitrate-nitrite nitrogen (0.31 mg N/l) and TKN (2.5 mg N/l) in discharges were much greater than in canals, tributaries, marshes, or lakes. Pumped discharges also had the highest mean levels of color (307 CPU) and turbidity (9.2 NTU).

Table 25. Concentrations of water chemistry constituents in discharge from the Zigzag agricultural pump over a 24 h. period.

Parameter	Time and Date 5:00 P.M. 6-24-1982	 8:40 P.M. 6-25-1982	 4:30 P.M. 6-25-1982
Chloride (mg/l)	110	112	97
Total Dissolved Solids (mg/l)	546	560	593
Color (cpu)	284	342	286
Total Phosphorus (mg/l)	0.08	0.12	0.17
Orthophosphate (mg/l)	0.05	0.09	0.10
Total Kjeldahl Nitrogen (mg/l)	2.3	2.4	1.8
Nitrate-Nitrite Nitrogen (mg/l)	0.58	0.83	1.08
Ammonia Nitrogen (mg/l)	0.24	0.11	0.07
Dissolved Oxygen (mg/l)	1.2	2.5	4.6
B.O.D. (mg/l)	1.4	3.0	2.6

Furthermore, mean levels of chloride in discharges (130 mg/l) were approximately four times greater than the levels found in natural marsh (Jane Green Marsh) or tributaries. This is unexpected in that for other water body types concentrations of chloride and color were inversely related. In terms of both BOD and dissolved oxygen, pumped discharges had levels within the range found for other water body types.

Sedimentation

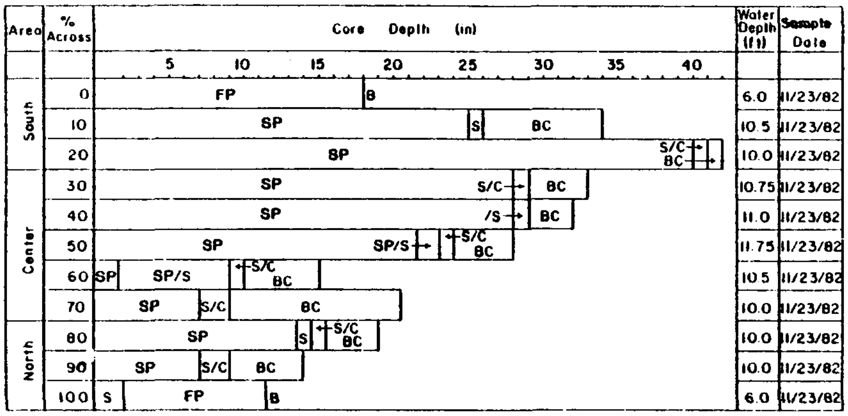
All the lakes had organic sediments in the form of sedimentary or fibrous peat (sensu Brady, 1974) (Figures 31 - 36). These organic strata were typically underlain by sand and the sand by clay. Sedimentary peat was the predominate form of organic sediment in all lakes except Hell 'n Blazes and Sawgrass, where fibrous peat predominated. The mean total thickness of organic strata and the mean thickness of sedimentary peat were both greatest in Blue Cypress Lake (24.5 and 22.9 inches, respectively) and lowest in Lake Winder (3.4 and

Table 26. Mean concentrations of water chemistry constituents during the wet season (June - October) for all stations within each of six types of water body.

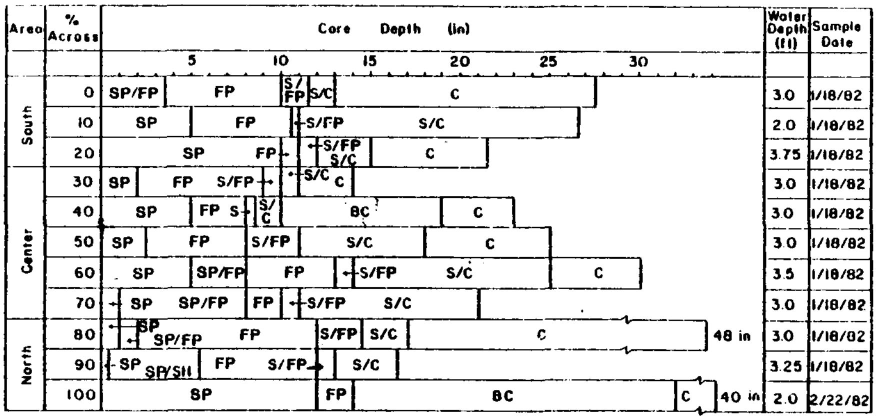
	Water Body Type					
Parameter	Discharge	Canal	JGM*	Tributary	Lake	Marsh
Water Temp. (C)	29.0	25.0	23.5	23.0	24.5	26.0
Turbidity (NTU)	7.9	3.3	2.2	1.3	2.6	4.8
Conductivity (umhos/cm)	820	1193	190	145	835	910
Dissolved Oxygen (mg/l)	3.2	4.4	2.3	2.1	6.9	2.8
pH	7.0**	6.9	5.4	5.9	7.0	5.7
Color (cpu)	307	108	228	200	121	190
B.O.D. (mg/l)	3.2	2.6	5.2	2.2	1.8	4.7
Suspended Solids (mg/l)	24	10	54	3.7	8.9	23
Dissolved Solids (mg/l)	580	700	168	186	530	655
Chloride (mg/l)	130	238	32	30	190	201
Sulfate (mg/l)	59	68	1.8	6.5	50	43
Alkalinity (mg/l)	135	106	22	28	73	50
Orthophosphate (mg P/l)	0.16	0.06	0.05	0.06	0.04	0.08
Total Phosphorus (mg P/l)	0.26	0.11	0.19	0.11	0.08	0.16
Ammonia N (mg N/l)	0.16	0.10	-	0.02	0.04	0.17
Nitrate-Nitrite N (mg N/l)	0.31	0.05	0.01	0.04	0.04	0.02
TKN (mg N/l)	2.5	1.1	-	0.9	1.5	1.0
Chlorophyll-a, uncor. (ug/l)	9.9	24	75	3.9	10	50

* Undisturbed marsh

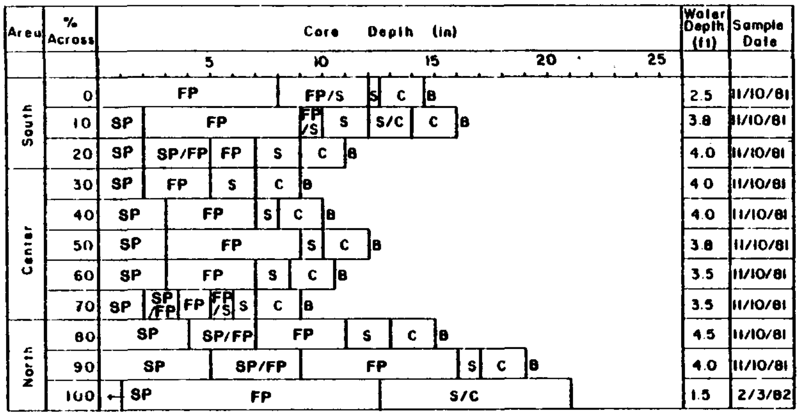
** Arithmetic mean of untransformed pH, not of hydronium ion concentration



Blue Cypress Lake – Longitudinal Transect.

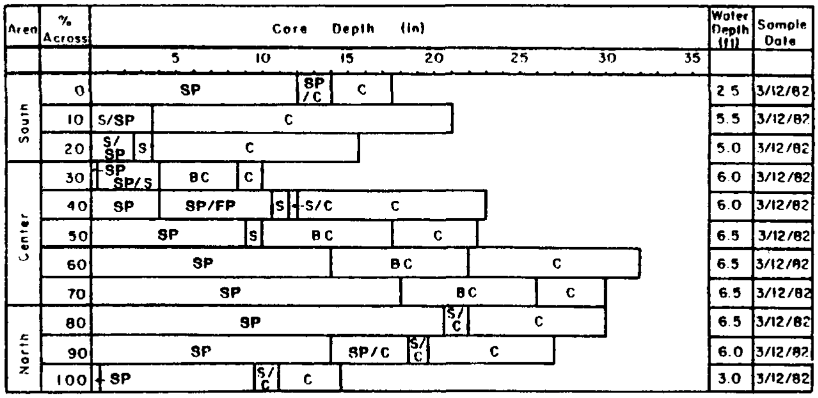


Lake Helen Blazes - Longitudinal Transect.

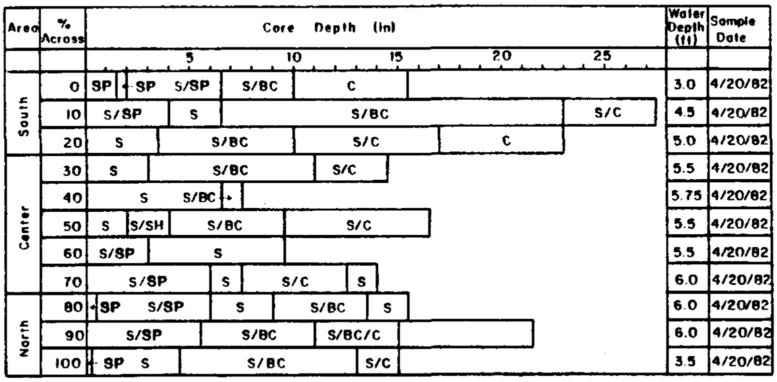


Lake Sawgrass – Longitudinal Transect.

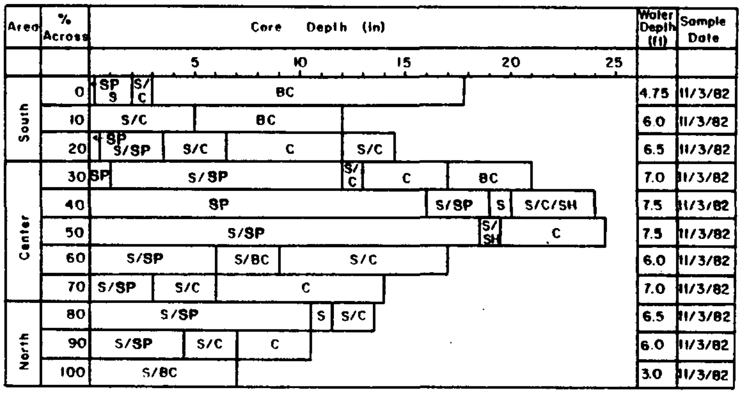
Figure 31. Sediment stratigraphy and depth along longitudinal transects of lakes Blue Cypress, Hell 'n Blazes, and Sawgrass. S = sand, C = clay, BC = black clay, SH = shells, FP = fibrous peat, SP = sedimentary peat, B = bottom of core, did not reach hardpan (modified from Belanger et al., 1983).



Lake Washington – Longitudinal Transect

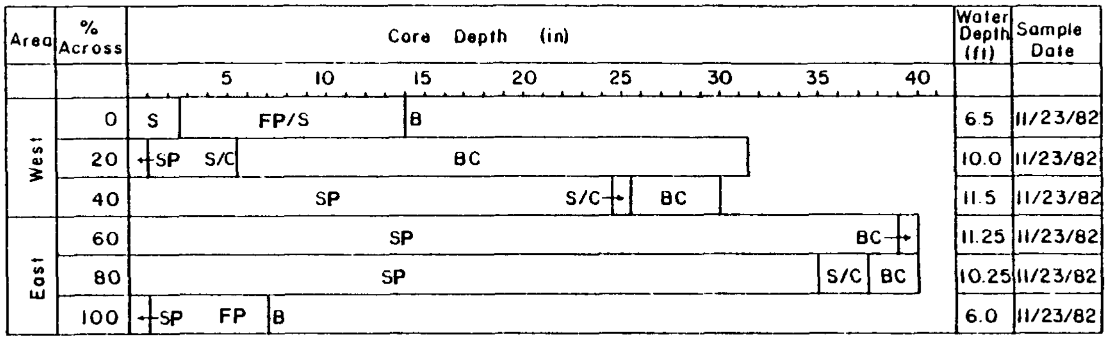


Lake Winder – Longitudinal Transect

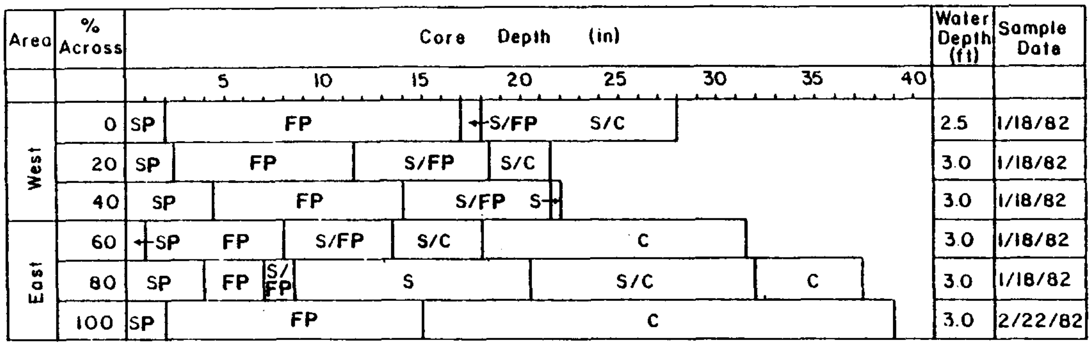


Lake Poinsett – Longitudinal Transect

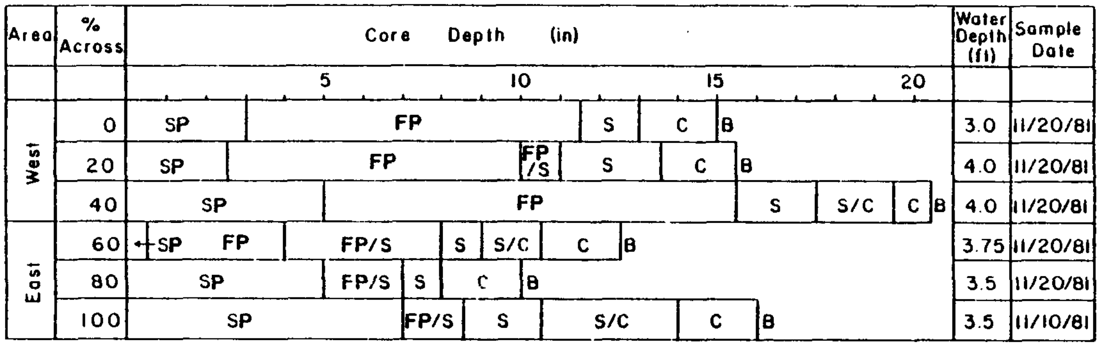
Figure 32. Sediment stratigraphy and depth along longitudinal transects of lakes Washington, Winder and Poinsett. Abbreviations explained in figure 31. Modified from Belanger et al., 1983.



Blue Cypress Lake - Transverse Transect.

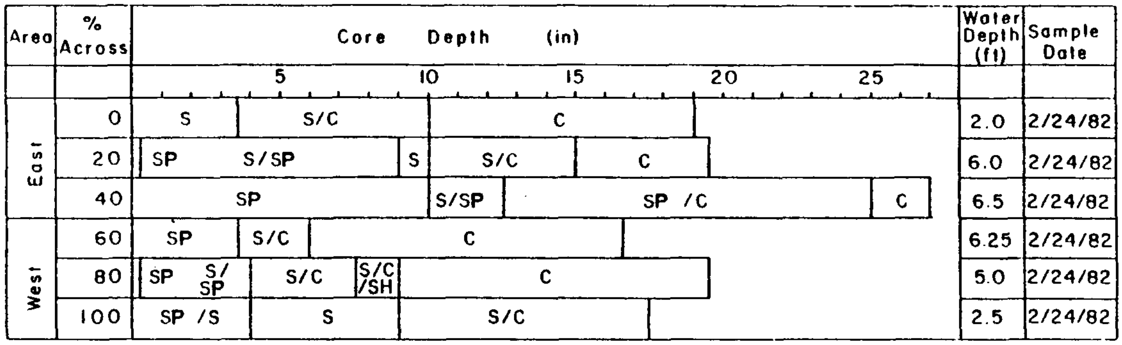


Lake Helen Blazes - Transverse Transect.

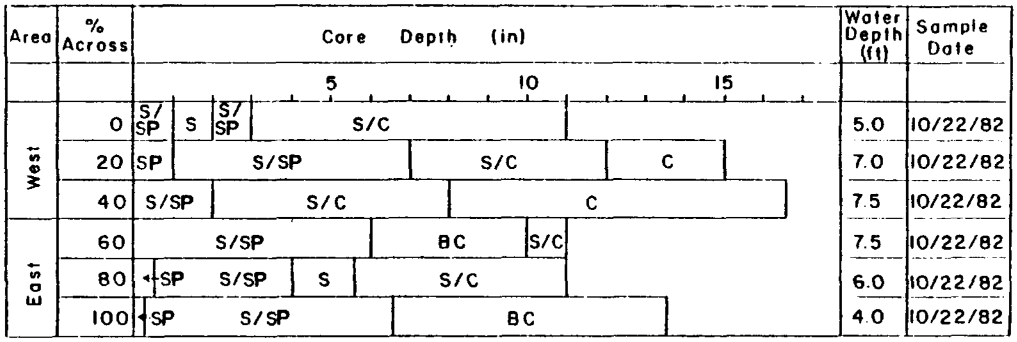


Lake Sawgrass - Transverse Transect.

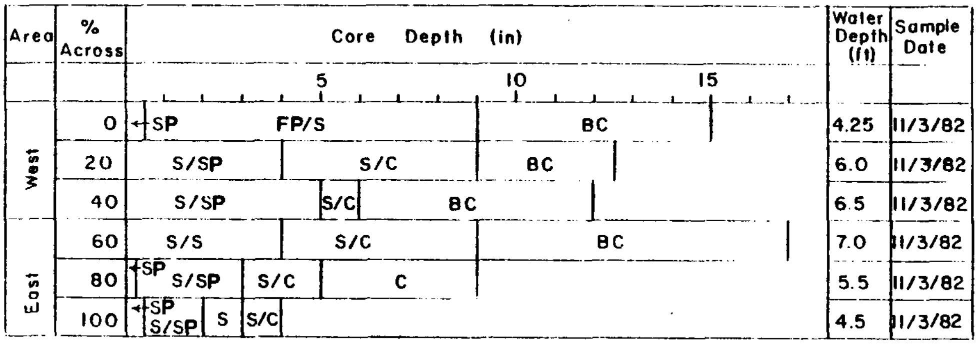
Figure 33. Sediment stratigraphy and depth along transverse transects of lakes Blue Cypress, Hell 'n Blazes, and Sawgrass. Abbreviations explained in figure 31. Modified from Belanger et al., 1983.



Lake Washington – Transverse Transect.

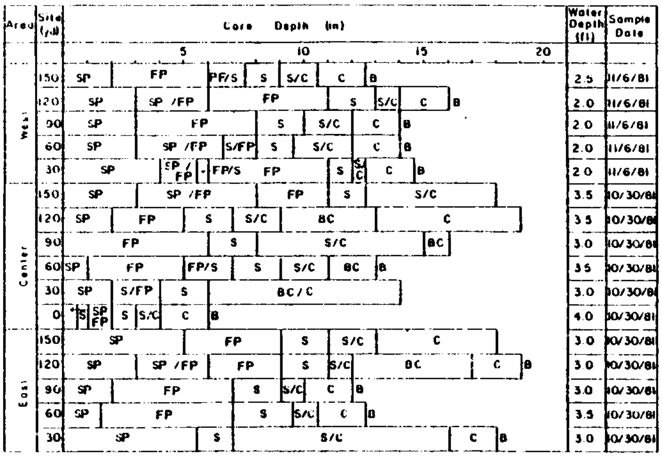


Lake Winder – Transverse Transect.

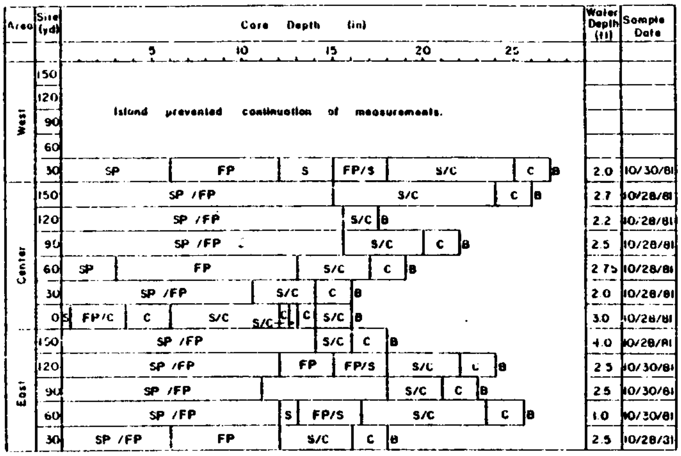


Lake Poinsett – Transverse Transect

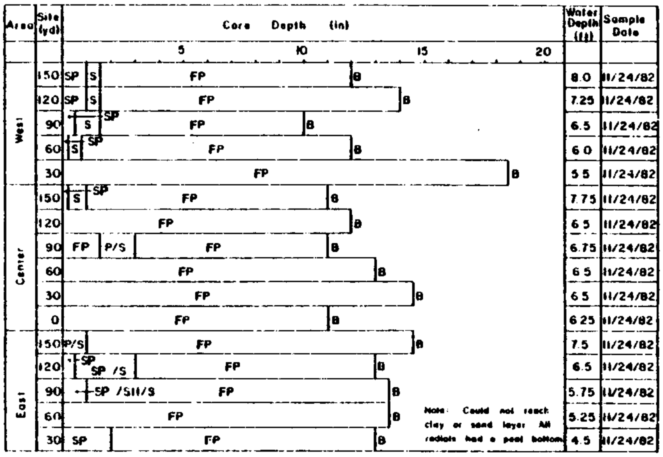
Figure 34. Sediment stratigraphy and depth along transverse transects of lakes Washington, Winder and Poinsett. Abbreviations explained in figure 31. Modified from Belanger et al., 1983.



Lake Sawgrass - Inlet Radials.

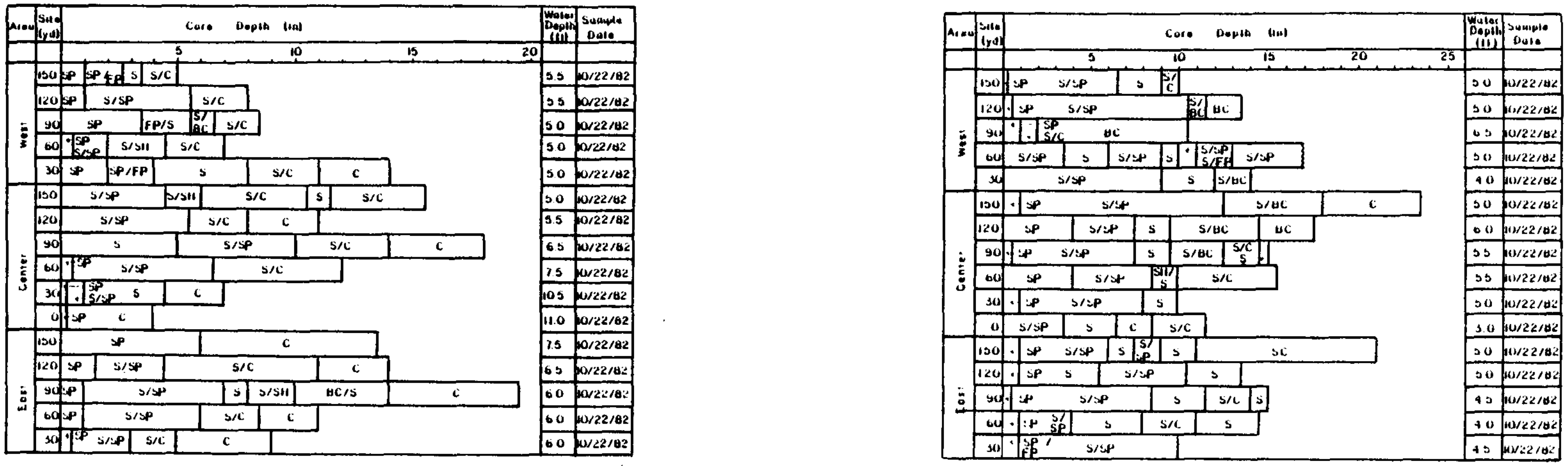


Lake Hellen Blazes - Inlet Radials.



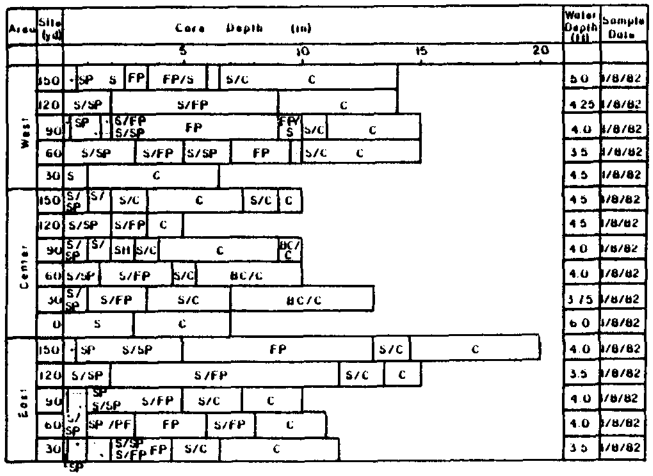
Blue Cypress Lake - Inlet Radials.

Figure 35. Sediment stratigraphy and depth along radial transects at the river's entrance into lakes Blue Cypress, Hell 'n Blazes, and Sawgrass. Abbreviations explained in figure 31. Modified from Belanger et al., 1983.



Lake Poinsett Inlet Radials

Lake Winder Inlet Radials



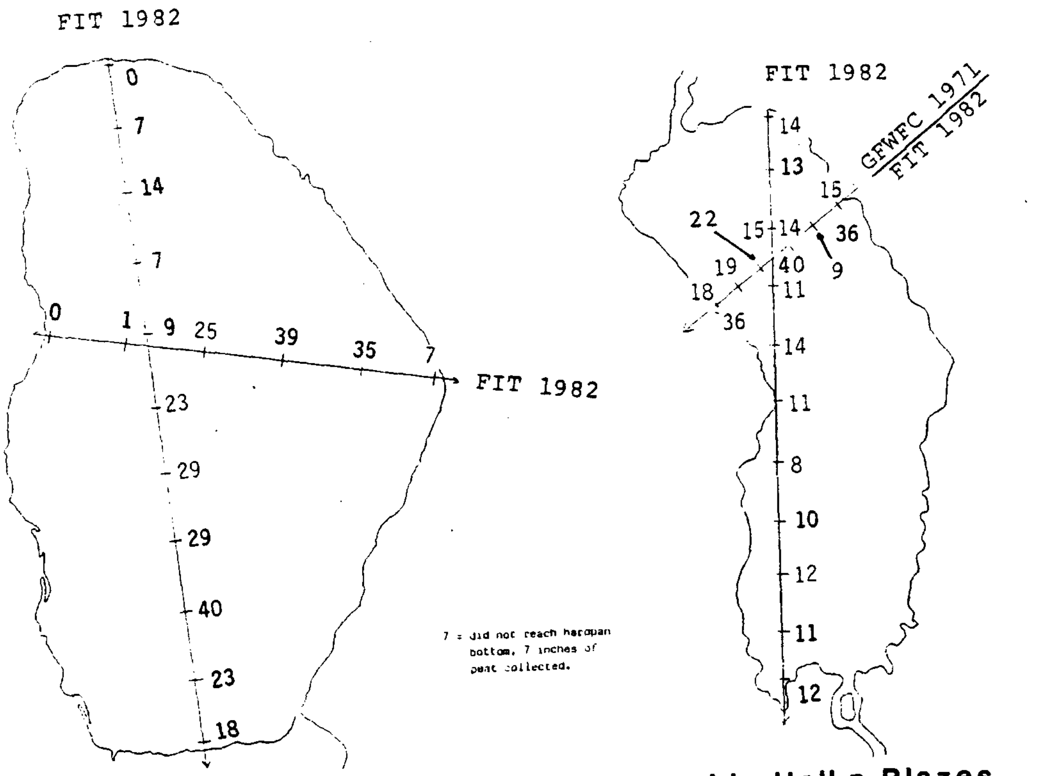
Lake Washington – Inlet Radials

Figure 36. Sediment stratigraphy and depth along radial transects at the river's entrance into lakes Washington, Winder, and Poinsett. Abbreviations explained in figure 31. Modified from Belanger et al., 1983.

1.6 inches, respectively) (Tables 27a - c). Fibrous peat deposits were most extensive in Lakes Hell 'n Blazes and Sawgrass (Table 27c).

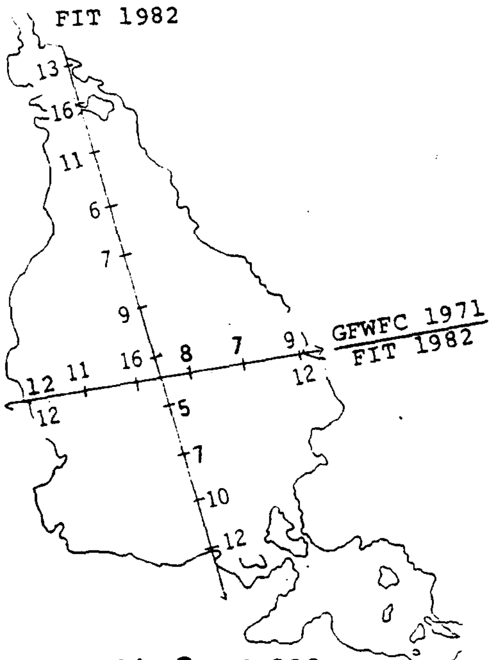
For several lakes there was a distinct pattern of sediment distribution. In Blue Cypress Lake, the thickness of the sedimentary peat stratum generally declined from south to north from a maximum of 40 in. to a minimum of 0 inches. The thickness of organic strata in Lake Washington, however, increased from south (minimum of 2.5 inches) to north (maximum of 20.5 inches). In Lake Poinsett the thickness of organic stata was greatest near the center of the lake both longitudinally and transversely. In other lakes organic sediments were nearly evenly (Hell 'n Blazes, Sawgrass) or erratically (Winder) distributed.

The transverse transects closely duplicated transects sampled earlier by Cox et al. (1976) (Figures 37 and 38). Comparison of data from the two studies indicates the the thickness of organic strata in Lake Hell 'n Blazes decreased by approximately 20 inches from 1971 - 1982. The change was apparently due to the loss of a stratum described as unconsolidated silt and plant detritus by Cox et al. (1976) (Figure 39). All the other lakes, except Sawgrass, however, generally had thicker layers of organic strata in 1983 than in 1971. This was especially true of Lake Washington, where an organic stratum was generally lacking in 1971 but was as thick as 25 inches in 1983. The increased thickness of the organic stratum in lakes Washington, Winder and Poinsett was due to greater amounts of sedimentary peat, deposits of fibrous peat being of little importance.



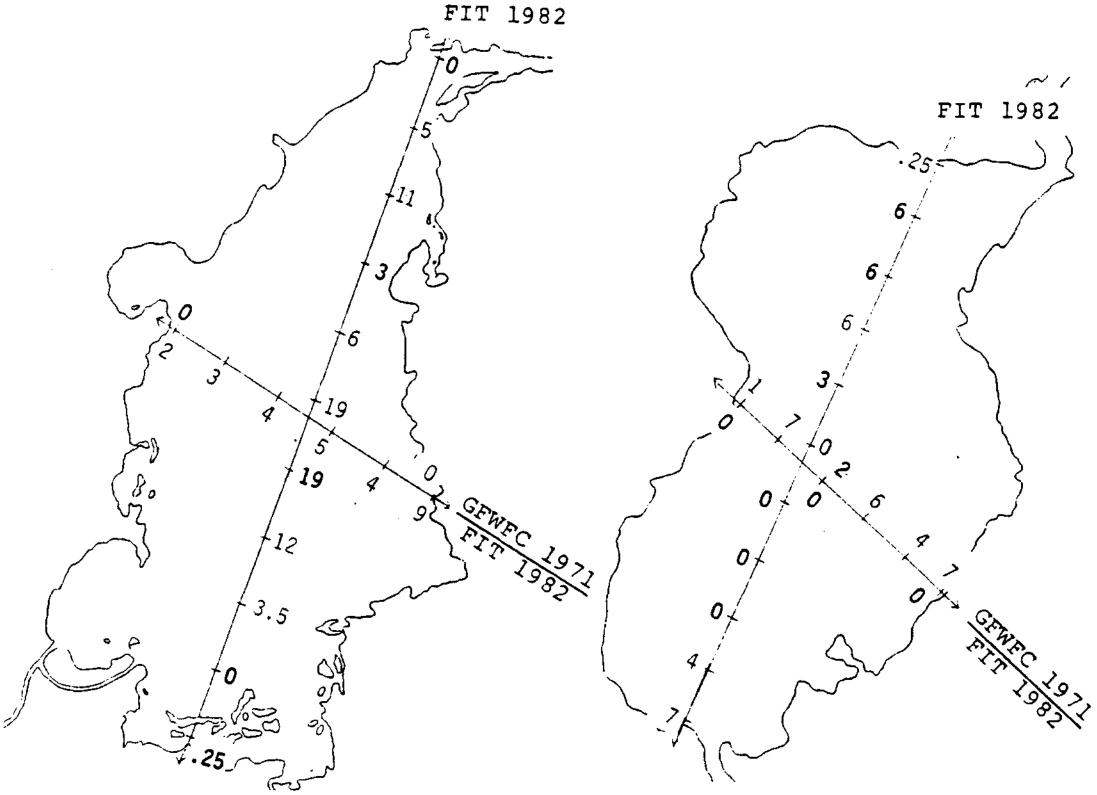
Blue Cypress Lk.

Lk. Hell,n Blazes



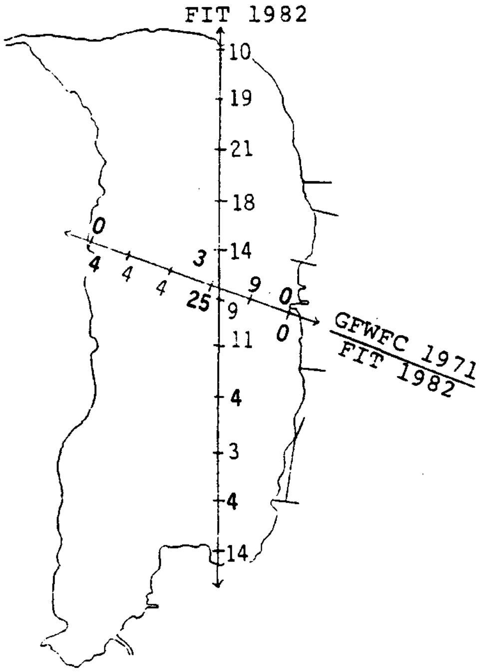
Lk. Sawgrass

Figure 37. Depth (inches) of organic sediment along longitudinal and transverse transects sampled by FIT and along transverse transects sampled by Cox et al. (1976) in lakes Blue Cypress, Hell 'n Blazes, and Sawgrass (from Belanger et al., 1983).



Lk. Poinsett

Lk. Winder



Lk. Washington

Figure 38. Depth (inches) of organic sediment along longitudinal and transverse transects sampled by FIT and along transverse transects sampled by Cox et al. (1976) in lakes Washington, Winder and Poinsett (from Belanger et al., 1983).

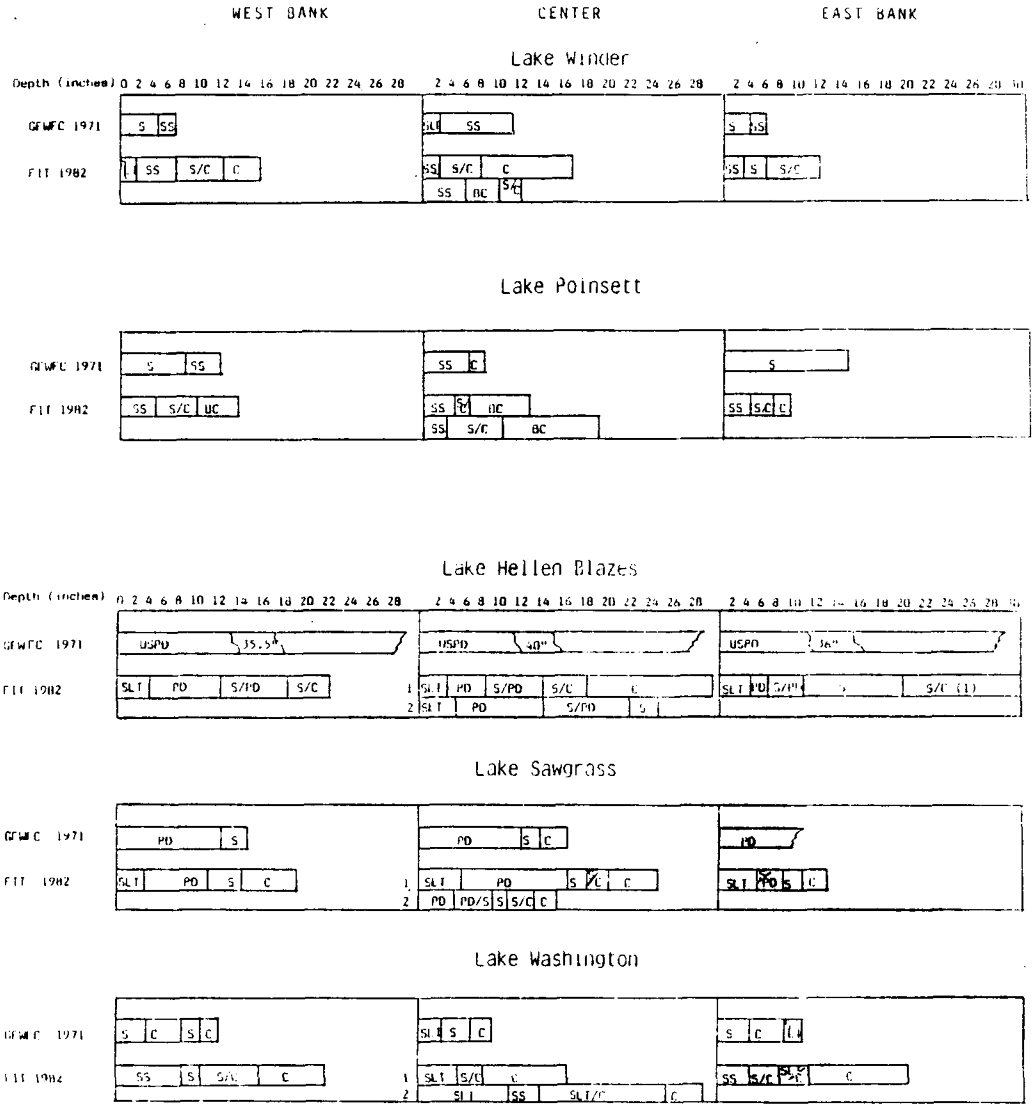


Figure 39. Depth and stratigraphy of sediment cores obtained by FIT and Cox et al. (1976) along transverse transects of lakes Hell 'n Blazes, Sawgrass, Washington, Winder, and Poinsett (from Belanger et al., 1983). S = sand, C = clay, SLT = silt (sedimentary peat), PD = plant detritus (fibrous peat), USPD = unconsolidated silt and plant detritus.

]

Table 27. Variation in the thickness (inches) of strata in sediments of lakes in the Upper St. Johns River. The percentages of lake dimensions increase from south to north for length and from east to west for width. (a) Strata containing visable organic material, (b) strata primarily composed of sedimentary peat, (c) strata primarily composed of fibrous peat.

(a). Thickness (inches) of strata containing visable organic material.

	Position of Core																
	Percentage of Lake's Length											Percentage of Lake's Width					
Lake	0	10	20	30	40	50	60	70	80	90	100	0	20	40	60	80	100
Blue Cypress	>18.0	25.0	40.0	28.5	29.0	23.0	9.0	7.0	13.5	7.0	0.0	>14.0	1.0	24.5	39.0	35.0	1.0
Hell 'n Blazes	11.5	11.0	12.0	10.0	8.0	11.0	14.0	11.0	14.5	13.0	14.0	18.0	18.5	21.5	13.5	8.5	15.0
Sawgrass	12.0	10.0	7.0	5.0	7.0	9.0	7.0	6.0	11.0	16.0	12.5	11.5	11.0	15.5	8.0	7.0	8.5
Washington	14.0	3.5	2.5	4.0	11.5	9.0	14.0	18.0	20.5	18.5	9.5	0.0	9.0	25.0	3.5	4.0	4.0
Winder	6.5	4.0	0.0	0.0	0.0	0.0	3.0	6.0	6.0	5.5	0.5	1.0	7.0	2.0	6.0	4.0	6.5
Poinsett	0.5	0.0	3.5	12.0	19.0	18.5	6.0	3.0	10.5	4.5	0.0	9.0	4.0	5.0	4.0	3.0	2.0

(b). Thickness (inches) of strata composed of sedimentary peat.

	Position of Core																
	Percentage of Lake's Length											Percentage of Lake's Width					
Lake	0	10	20	30	40	50	60	70	80	90	100	0	20	40	60	80	100
Blue Cypress	0.0	25.0	40.0	28.5	28.0	21.5	1.5	7.0	13.5	7.0	0.0	0.0	1.0	24.5	39.0	35.0	1.0
Hell 'n Blazes	0.0	5.0	10.0	2.0	5.0	2.5	5.0	1.0	1.0	0.5	12.0	2.0	2.5	4.5	1.0	4.0	2.0
Sawgrass	0.0	2.0	2.0	2.0	3.0	3.0	3.0	2.0	4.0	5.0	1.0	3.0	2.5	5.0	0.5	5.0	7.0
Washington	12.0	3.5	2.5	0.5	4.0	9.0	14.0	18.0	20.5	14.0	9.5	0.0	9.0	12.5	3.5	4.0	0.0
Winder	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.5	1.0	1.0	2.0	6.0	4.0	6.5
Poinsett	0.5	0.0	3.5	12.0	16.0	18.5	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.5	0.5

(c). Thickness (inches) of strata composed of fibrous peat.

	Position of Core																
	Percentage of Lake's Length											Percentage of Lake's Width					
Lake	0	10	20	30	40	50	60	70	80	90	100	0	20	40	60	80	100
Blue Cypress	>18.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hell 'n Blazes	11.5	6.0	2.0	8.0	3.0	8.5	9.0	10.0	13.5	7.5	2.0	16.0	16.0	17.0	12.5	4.5	13.0
Sawgrass	12.0	8.0	5.0	3.0	4.0	6.0	4.0	4.0	7.0	11.0	11.5	8.5	8.5	10.5	7.5	2.0	1.5
Washington	0.0	0.0	0.0	0.0	6.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Winder	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Poinsett	0.0	0.0	0.0	0.0	0.0	8.0	0.0	0.0	0.0	0.0	0.0	8.5	0.0	0.0	0.0	0.0	0.0

The sediments of Lake Hell 'n Blazes had high concentrations of water down to a depth of about 50 cm from the soil-water interface (Figure 40) where organic sediments graded into a sand/clay mixture. Levels of organic material were well above 50 percent of the dry weight down to a depth of 35-40 cm beyond which levels were generally below 50 percent (Figure 41). Below 50 cm, levels of organic material rapidly declined to near 0 percent. Pb-210 activity varied in an erratic manner along the sediment profile. In an ideal system, with constant rates of Pb-210 supply and sedimentation and without mixing, Pb-210 activity would decline exponentially with distance from the sediment surface. If the sedimentation rate had not been constant, a strong possibility in the upper basin, changing sedimentation rates coupled with nearly constant loading of P-210 to the system (Robbins, 1978) would cause dilution of Pb-210 during years of high sedimentation and concentration of Pb-210 during years of low sedimentation. The following results assume that varying rates of sedimentation did, in fact, account for the variation in Pb-210 activity along core 2.

The deepest organic sediments in Lake Hell 'n Blazes were deposited more than 150 years ago (Figure 42) (150 years is the detection limit of the methodology). The dry weight of sediment deposited per year increased from the base to the surface of the core (Figure 43) indicating that the rate of sediment deposition increased over the past 150 years (Figure 44). There appears to have been five distinct periods of deposition. From 1832 - 1882 (the precision of data varies from ±2-3 years near the surface to ±8-10 years near the base, M. Binford, personal communication), the sedimentation rate was constant with roughly 50 percent of the sediment being organic. Sedimentation

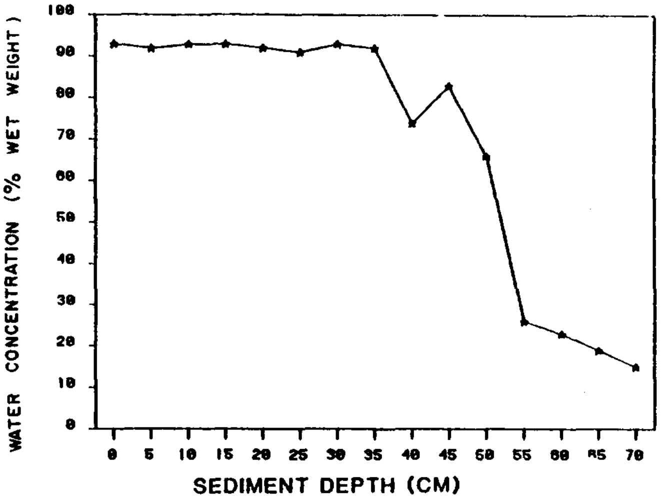


Figure 40. Variation in the concentration of water with sediment depth in the sediments of Lake Hell 'n Blazes.

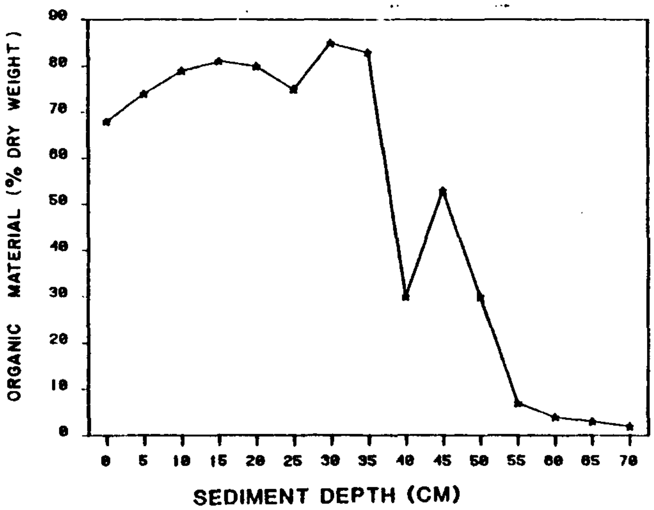


Figure 41. Variation in the concentration of organic material with sediment depth in the sediments of Lake Hell 'n Blazes.

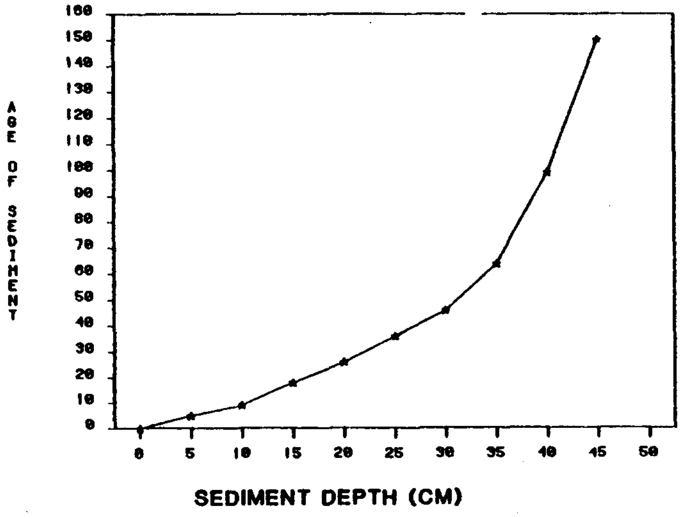


Figure 42. The relationship between sediment depth and age in the sediments of Lake Hell 'n Blazes as determined by Pb-210 dating.

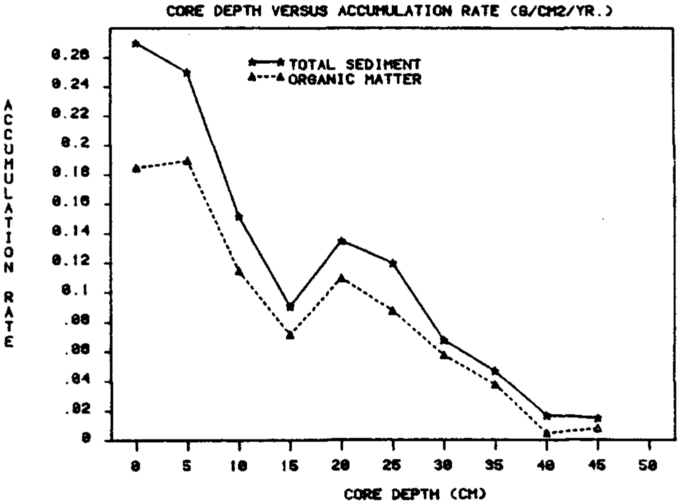


Figure 43. The relationship between sediment depth and accumulation rate in Lake Hell 'n Blazes.

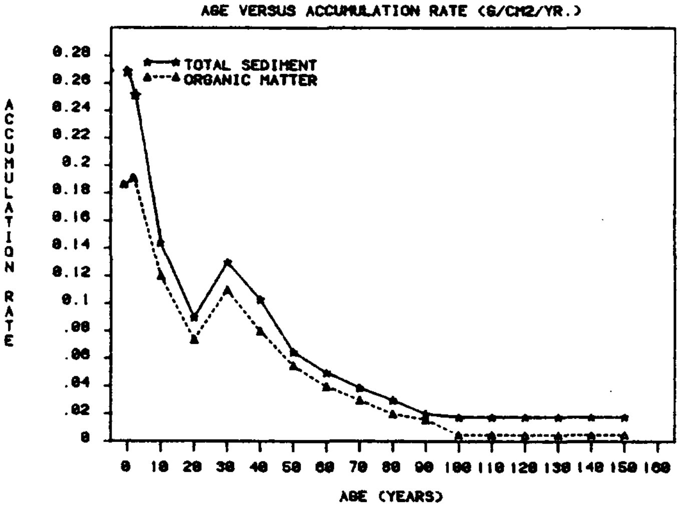


Figure 44. The relationship between sediment age and accumulation rate in Lake Hell 'n Blazes.

accelerated at a roughly constant rate from approximately 1882 to 1935 then accelerated at a greater rate from approximately 1935 - 1952. During these two periods the acceleration in sedimentation was primarily due to the deposition of organic material, consequently the level of organic material in the sediment increased. Following a brief deceleration of sedimentation rates during the early fifties and sixties, sedimentation accelerated again and at a rate greater than that of earlier periods. This last period of sedimentation was also distinct from previous periods in that there was a decline in the proportion of organic sediments.

Floodplain vegetation and hydrology

In the early period (1943), communities with a significant shrub component occurred primarily in transition areas, such as lake shores or marsh margins, and disturbed areas, such as along the Fellsmere Grade (Figures 45 - 48). In the later periods (1980 or 1981), shrubs were more widely distributed, particularly near Lake Hell 'n Blazes (site 2) and near the Fellsmere Grade (site 3) where the percentage of low marsh acreage declined from 50 to 10 percent and from 98 to 8 percent, respectively (Table 28). Northeast of Lake Hell 'n Blazes (site 2), the acreage of low marsh declined not only due to shrub expansion but also due to expansion of high marsh. Southeast of Blue Cypress Lake (site 4) and southeast of Lake Poinsett (site 1), the change in acreages and positions of marsh communities changed very little between the time periods sampled.

LAKE POINSETT MARSH

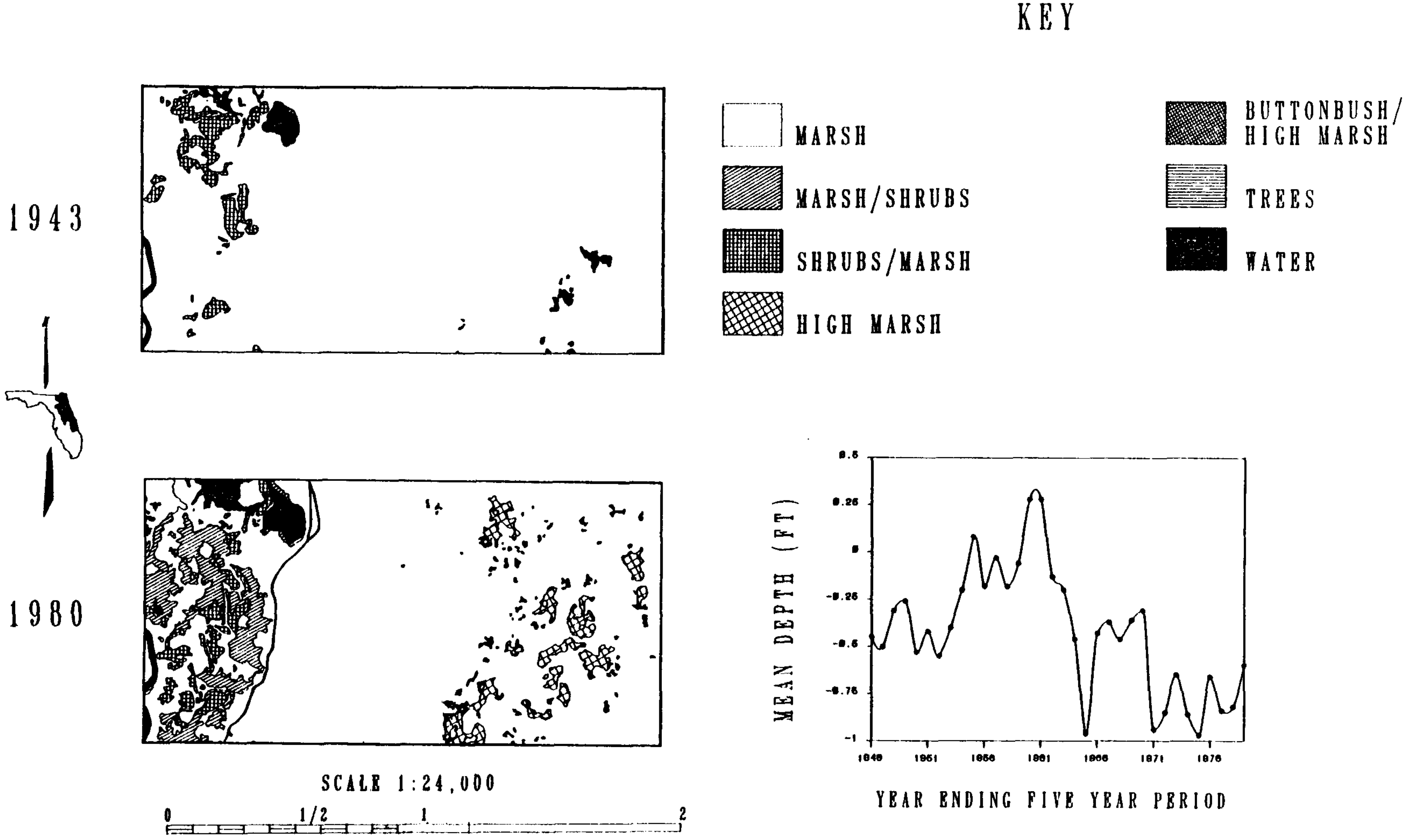


Figure 45. Changes in the distribution and composition of floodplain plant communities near Lake Poinsett (site 1) and ten-year moving averages for mean depth of inundation of the floodplain at 12.5 ft. NGVD.

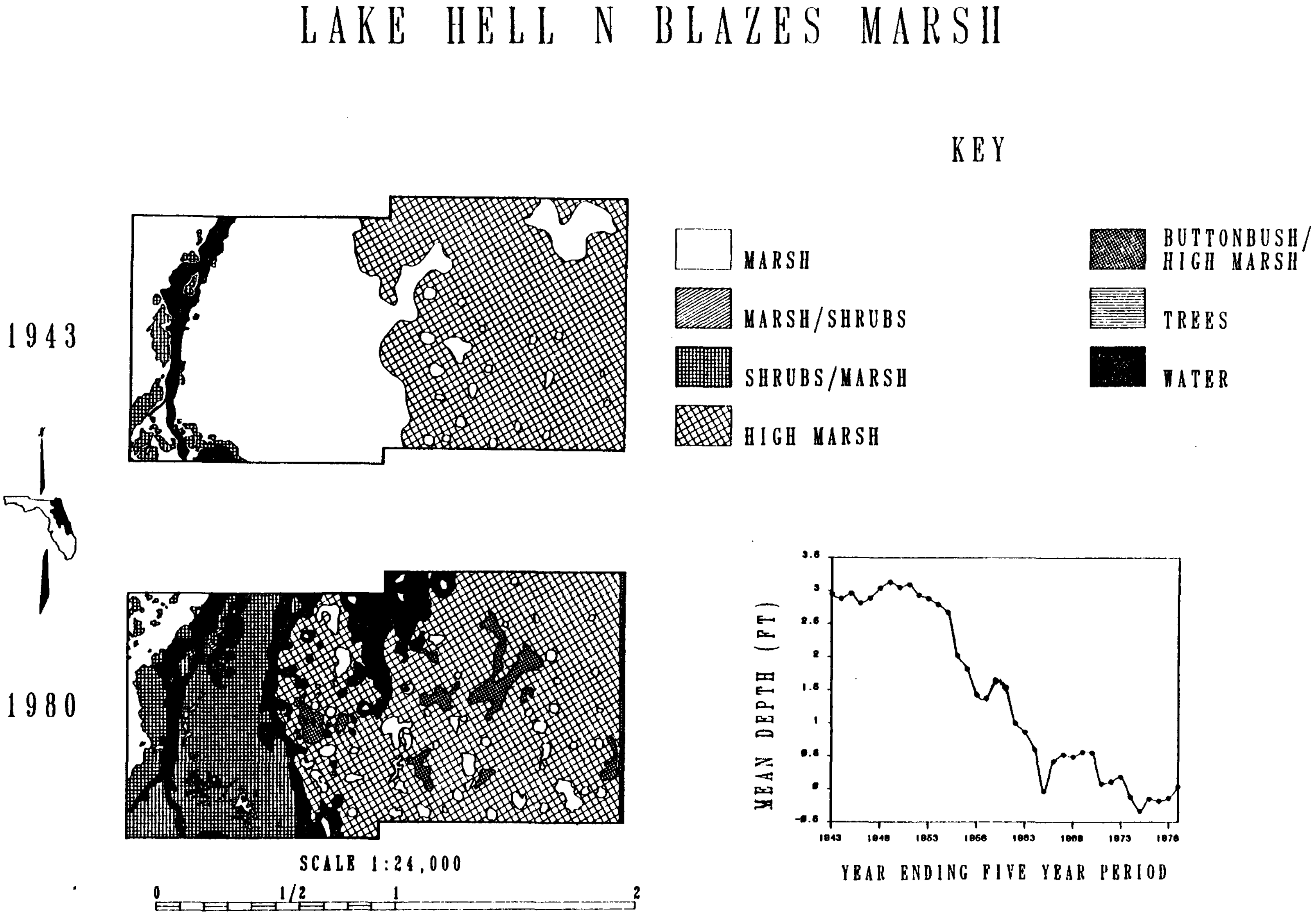


Figure 46. Changes in the distribution and composition of floodplain plant communities near Lake Hell 'n Blazes (site 2) and ten-year moving averages for mean depth of inundation of the floodplain at 14 ft. NGVD.

FELLSMERE GRADE MARSH

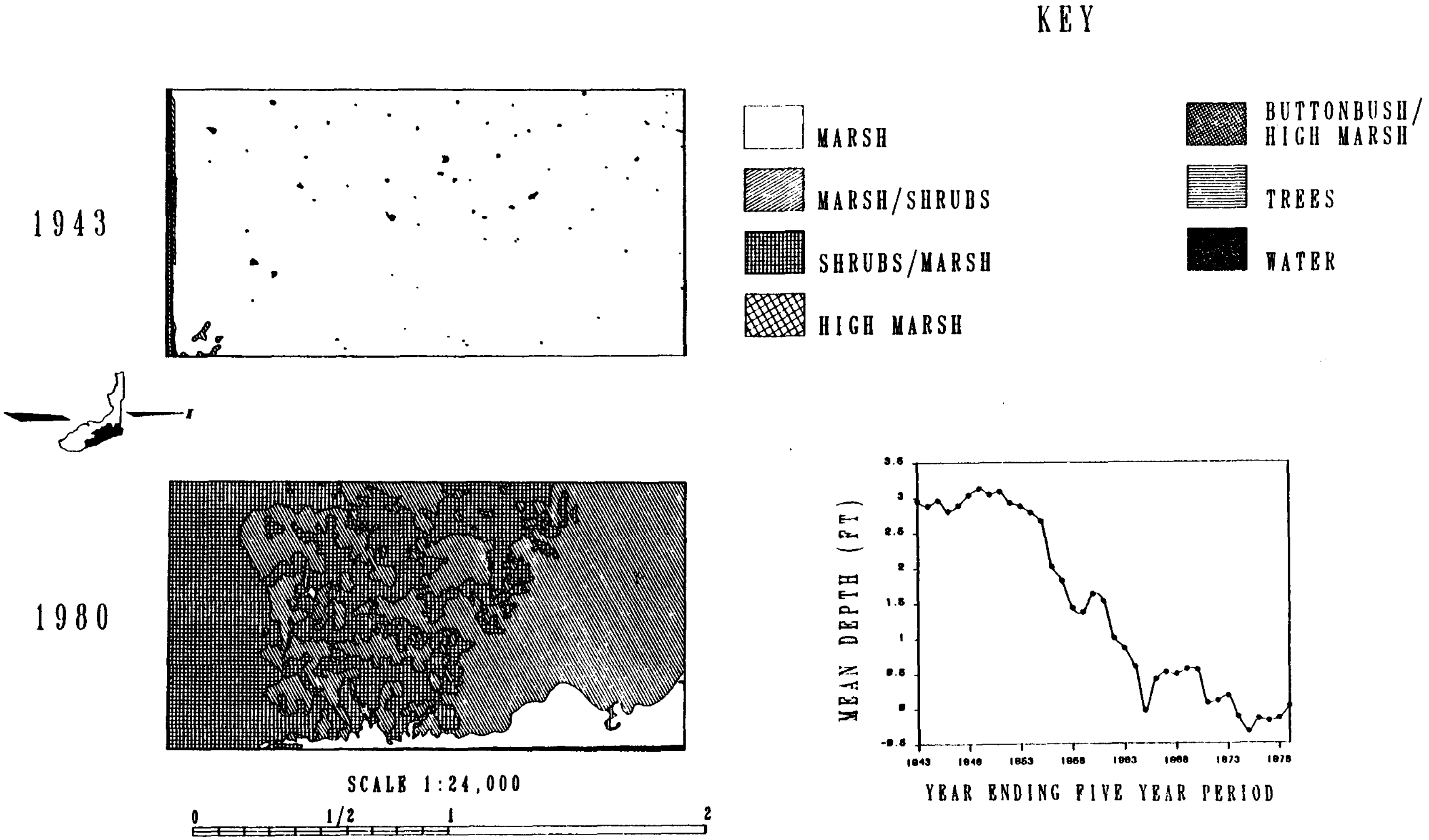


Figure 47. Changes in the distribution and composition of floodplain plant communities just north of the Fellsmere Grade (site 3) and ten-year moving averages for mean depth of inundation of the floodplain near Lake Hell 'n Blazes (location of nearest stage recorder) at 13.5 ft. NGVD.

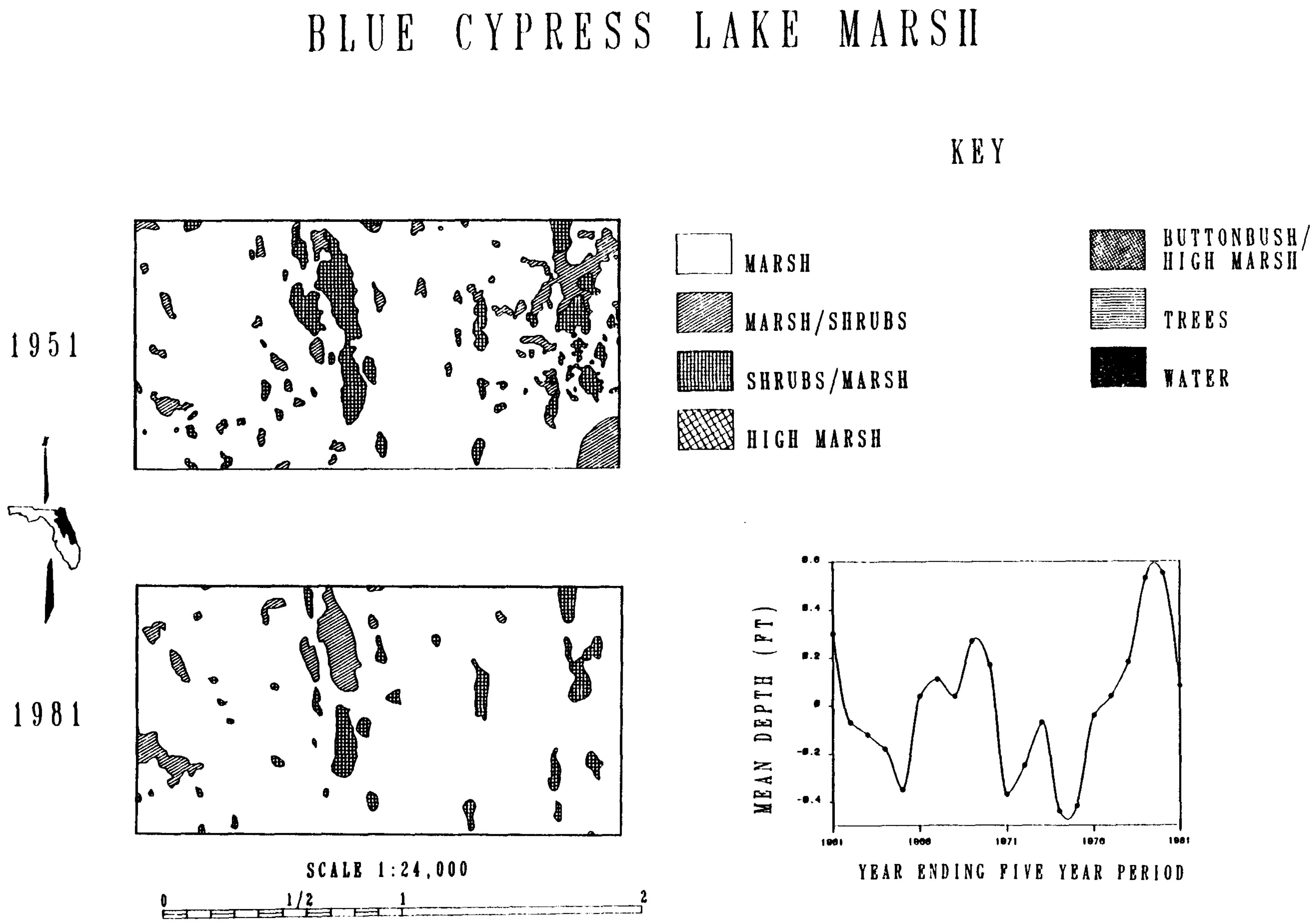


Figure 48. Changes in the distribution and composition of floodplain plant communities near Blue Cypress Lake (site 4) and ten-year moving averages for mean depth of inundation of the floodplain at 23.0 ft. NGVD.

At site 2 the changes in vegetation were concomitant with a progressive reduction in inundation of the marsh (Figure 46). Although hydrologic data were not available for site 3, it's position relative to the Fellsmere Grade strongly suggests a similar or more intense reduction in inundation. The only sites which were vegetatively stable, sites 1 and 4, were also the only sites which experienced no strong temporal trends in their inundation characteristics (Figures 45 and 48).

Table 28. Relative cover (percent) of different types of vegetation and of open water during 1943 and 1980 or 1981 of selected sites in the Upper St. Johns River Basin as indicated by aerial photographs.

	Area and Date							
	1		2		3		4	
Vegetation Class	1943	1980	1943	1980	1943	1980	1943	1981
low marsh	95	84	50	10	98	8	84	91
low marsh/shrub	<1	7	0	1	1	48	7	4
shrub/low marsh	3	3	3	20	1	43	9	5
high marsh	0	0	44	50	0	0	0	0
high marsh/shrub	0	0	0	4	0	0	0	0
trees	0	0	0	1	0	<1	0	0
open water	2	3	3	15	1	1	0	0
Total area (ha.)	526	518	535	532	522	527	520	520

Associated with the reduction in inundation of the floodplain, was a long-term decline in rainfall as indicated by five-year running means of total annual rainfall (Figure 49). The average annual total for the 52 year period of record is 136 cm (53.4 in.), however, the average for the recent 22 year period (1961 - 1982) is only 126 cm (49.6 in.), about 10 cm (3.8 in.) below the 52 year average. Moreover, seven of the last ten years were below average, with two extreme droughts in 1980 and 1981 when total rainfall was 41 cm (16.1 in.) and 34 cm (13.3 in.) below average, respectively.

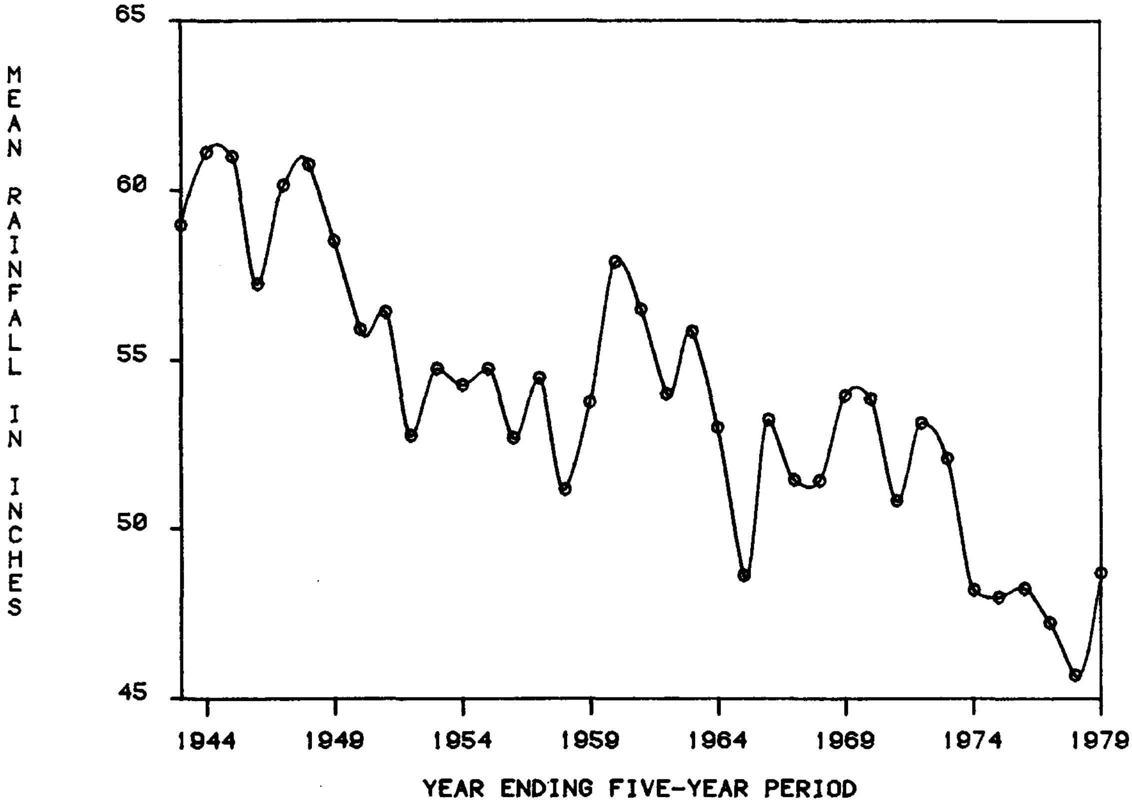


Figure 49. Five-year moving average of total annual rainfall in the Upper Basin.

DISCUSSION

The Upper St. Johns River is, apparently, an ecosystem under stress. This is evidenced by an alteration of floodplain vegetation over large areas; a marked decline in the abundance of waterfowl and other wetland birds, including the endangered wood stork and Everglades kite; and a decline in the standing stocks of game fish coupled with an increase in the frequency of fish kills. What environmental factors were the probable primary causes of these biological changes? The literature review and new data provided by this study allow a consideration of several broad possibilities: declining water quality, increasing sedimentation rates in lakes, habitat destruction, alteration of basin hydrology through development, and declining rainfall.

There are, apparently, no persistent, pervasive and severe water quality problems in the lakes and streams of the Upper Basin. Mean levels of total phosphorus (0.04 - 0.11 mg P/l) and nitrogen (1.32 - 1.56 mg N/l) indicate that much of the basin is eutrophic based on current criteria (Tables 20 and 21) but this is probably a natural eutrophy (Canfield, 1981). This is supported by data from a relatively pristine area, Jane Green Marsh, where mean levels of total phosphorus (0.08 mg P/l), nitrogen (1.53 mg N/l) and chlorophyll-a (20 ug/l) are also in the eutrophic range (Table 19).

A primary problem associated with eutrophy, especially cultural eutrophy, is blooms of noxious algae (Provasol, 1969). Algal production in the lakes of the Upper Basin, however, is low by world standards (Belanger et al., 1982) and algae blooms are not typically implicated as the cause of one of the basin's most serious ecological problems, fish kills. The low algal production is probably caused by low water transparency, due to high color, and competition for nutrients with large standing crops of macrophytes and periphyton.

Although the high nutrient concentrations of the basin have not caused obvious environmental harm in the past, they do indicate a potential for detrimental algae blooms. Consequently, any cultural eutrophication should be minimized. Further drainage of peat soils, with subsequent subsidence (Stephens, 1974) and concomitant release of nitrogen and phosphorus (Hortenstine and Forbes, 1972; Terry, 1980) should be curtailed. Futhermore, pumped agricultural discharges should be eliminated or managed in such a manner that their high loads of inorganic nitrogen and orthophosphate are diminished before entering the river channel.

Water quality factors which present serious, but intermittent, problems are dissolved oxygen and the total concentration of minerals. High mineral concentrations are less an ecological than an economical problem because levels to date have been well within the tolerances of freshwater organisms. Chloride levels in Lake Washington, however, did exceed the state standard for potable water supplies (250 mg/l) in July and August of 1981. Considering the long-term trend towards higher mean levels of chlorides, mineralization of the water column could become a significant problem in the future. Although some of the long-term increase in chloride levels was apparently caused by progressive reductions in river stage (Figures 24 and 25), most of the increase occurred after the period when stages declined most dramatically. This suggests a change in the relationship between river stage and chloride concentration, a change which was probably caused by an increase in mineral loading due to channelization of the marsh and agricultural runoff and discharges (Table 26).

Seasonal and diel fluctuations in levels of dissolved oxygen apparently present significant ecological problems. During the summer, when temperature and river stage increase, and particularly following droughts, oxygen levels may fall to levels below 1 mg/l nearly basin-wide. In addition, oxygen levels may fluctuate widely as a diel cycle in the marshes, a situation particularly injurious to growth of juvenile largemouth bass (Stewart et al., 1967). Yearly, seasonal and diel fluctuations in dissolved oxygen are undoubtedly natural, but the relationship between oxygen concentration and fluctuations in river stage suggests that the alteration of hydrology caused by basin development (i.e. decrease in mean stage and increase in yearly range in stage) may have intensified yearly and seasonal fluctuations in oxygen concentration. Normalization of the depth and duration of inundation of the floodplain marshes would probably result in a greater stability in concentration of dissolved oxygen.

It is not clear whether excessive sedimentation has occurred in Upper Basin lakes. Two quite different temporal patterns of sedimentation are suggested by current information. The decline in sediment thickness in Lake Hell 'n Blazes from 1971 - 1982 indicates a dynamic process in which sediments accrue in upstream lakes and later are transported downstream as suspended solids or bed load. Sediment stratigraphy and radiological aging of the strata, on the other hand, suggests that sediments have slowly accumulated over long periods of time with the rate of deposition increasing markedly over the last century. Conclusive determination of which of these two possibilities is most accurate requires data on sediment dynamics not yet available. Circumstantial evidence does exist, however, which can help in analysis of the dichotomy.

Several observations suggest that resuspension and subsequent transport of sediments downstream is rare in Lake Hell 'n Blazes: 1) flow rates are low due to the low hydraulic gradient and due to the presence of macrophytic benthic vegetation, 2) wave action is slight due to the small surface area (short fetch) and protective border of marsh vegetation, 3) sediments would not be easily resuspended because the bottom is well vegetated with aquatic macrophytes (primarily Vallisneria and Potamogeton) and because sediments below the top 10 - 15 cm are very cohesive (M. Binford, personal communication), and 4) the presence of recognizable and spatially consistent strata in the sediments suggests that deep mixing has not occurred. In the larger lakes of the Upper Basin, resuspension and transport of sediments does seem likely. Wave action in these lakes can be substantial. Moreover, suspended solids levels are often higher at their outlets than at their inlets (Figure 20), a situation which is uncommon for lakes Hell 'n Blazes and Sawgrass. In Hell 'n Blazes suspended solids levels averaged 9.0 mg/l at the inlet but only 2.3 mg/l at the outlet. By comparison, outlet concentrations in the larger lakes averaged 9.4 - 18.2 mg/l.

Thus the preponderance of circumstantial data indicates that sediment mixing and transport, athough potentially high in the large lakes, is low in Lake Hell 'n Blazes. The strongest argument against this is the large decline in thickness of organic sediments from 1971 - 1982. It is difficult, however, to conceive of such a large volume of sediment being transported downstream given our current understanding of the hydrodynamics of Lake Hell 'n Blazes. A more reasonable interpretation may be that the unconsolidated sediments found by Cox et al. (1976) were recently deposited (perhaps due to the sinking of a large, herbicide treated, hyacinth raft) and in an early state of decay. These unconsolidated plant fragments would rapidly oxidize and thus be absent at a later date. Credibility is added to this interpretation by the observation that the category of sediments lost in Hell 'n Blazes, unconsolidated silt and plant detritus, was found in only 1 of 33 transects sampled by Cox et al. (1976).

If it is accepted that sediments have accrued in a relatively undisturbed manner in Lake Hell 'n Blazes, it is interesting to examine the relationship between sedimentation and basin development.

During the first period of sediment deposition (1832-1882), no significant development of the basin occurred (Table 29) and the rate of sediment accumulation was very low ($<0.02 g/cm^2/year$). In the next two periods (1882-1952), sedimentation increased rapidly to a maximum of approximately $0.13 g/cm^2/year$. Considerable biological and physical modification of the basin also occurred during this time including the introduction of water hyacinth, the construction of roads across the marsh, and the construction of drainage canals. The sedimentation rate declined somewhat during the next period (1952-1962) despite the construction of some major drainage canals. This was probably caused by removal of the jams in 1952-1953 with a concomitant increase in mean flow rate and decline of mean river stages in lakes Washington, Hell 'n Blazes and Sawgrass. Construction of the Lake Washington weir during the next period (1962-1982) along with continued channelization and drainage may account for a rapid acceleration of sedimentation rates to a maximum of 0.27 $g/cm^2/year$.

Table 29. A chronology of biological and physical modification of the Upper St. Johns River and its floodplain.

Date	Modification
1890	Water hyacinth introduced into St. Johns River.
1911 - 1912	Construction of Fellsmere Grade and Fellsmere Canal.
1912 - 1914	Construction of Lateral-M, Zigzag, Lateral-O Canals.
1920 - 1930	Construction of State Road 60 and C-40 Canal.
1949 - 1952	Drainage, diking, and channelization of additional floodplain north of Fellsmere Grade.
1952 - 1960	Construction of Bulldozer, Mary-A, and North and South Morman Outside Canals. Destruction of floating islands of peat and vegetation (the "Jams") lodged in river channel north of Lake Washington.
1961 - 1981	Closure of gap in the Fellsmere Grade. Construction of Lake Washington Weir. Impoundment of Taylor Creek. Construction of C-54, C-40 Extension and South Morman Extension Canals. Construction of St. Johns Drainage District Reservoir. Initiation of maintenance chemical control of water hyacinth.

As with the problem of progressive mineralization, the high levels of suspended solids in the canals indicates that a reduction in sedimentation rates could be accomplished by elimination of canal flow. Plugging of canals would also decrease the area of connected habitat favorable for growth of water hyacinth, which may be a major contributor to the organic sediments of the basin (Gerry, 1982), and provide for mitigation of the generally poor water quality of agricultural discharges.

Neither the decline in water quality nor the possible increase in sedimentation rates seems sufficiently severe or pervasive to have caused the observed declines in fish and wildlife populations. It is well known, however, that floodplain feeding and breeding areas are vital for fish, waterfowl, and wading birds. Thus, alteration of the timing, frequency, depth or duration of inundation of the floodplain; alteration of the species structure of the vegetation; or reduction of the area of the floodplain would be expected to have far-reaching ecological consequences. The ecological decline of the Upper Basin probably stems primarily from alteration and loss of floodplain habitat. These two factors are interrelated in that reduction in floodplain area through drainage and diking reduces storage and thus causes hydrologic changes which, in turn, cause further alteration of the remaining floodplain habitat.

A major issue to be resolved in understanding the chronic water deficit of areas of the floodplain is whether the change in hydrology stemmed primarily from natural or anthropogenic events. The most significant natural event has been a decline in rainfall since about 1960 (Figure 49). If this was the major cause of the decline in inundation of the floodplain then the observed ecological changes have probably been experienced by the marsh before and are reversible in a reasonable length of time. Regression analysis, however, indicates a significant change in the relationship between total annual rainfall and annual mean surface water elevation at US 192 occurred in the mid-50s (Figure 50). The change was twofold: 1) for an equivalent rainfall event in the early (1939-1955) and recent (1956-1979) periods the concomitant expected mean annual surface water elevation is lower for

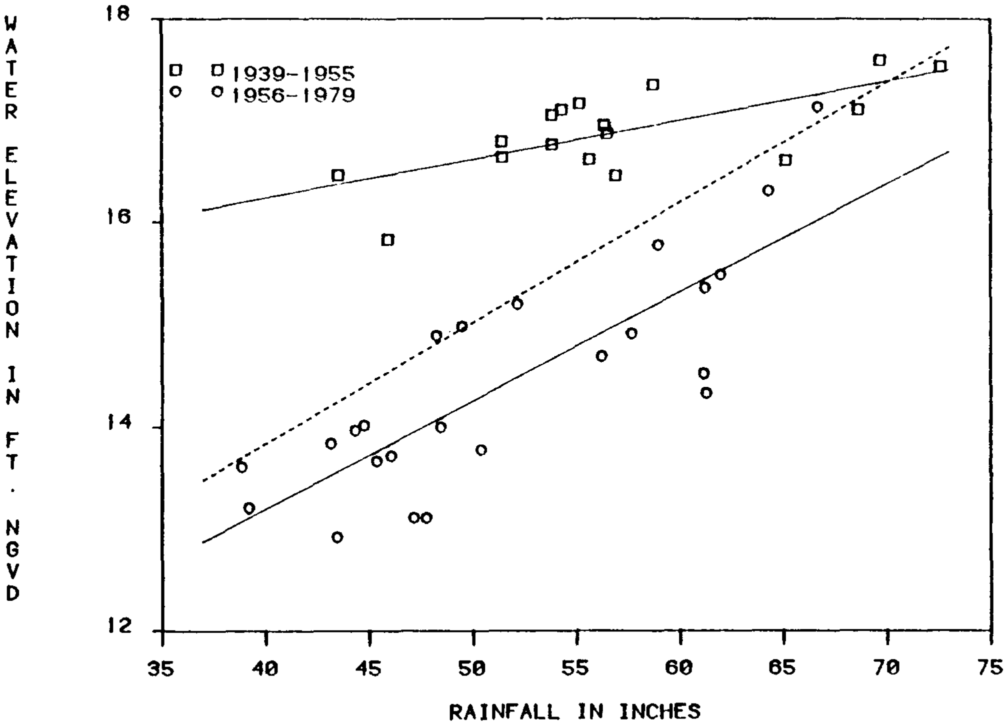


Figure 50. The relationship between annual mean surface water elevation at U.S. 192 and total annual rainfall in the Upper Basin.

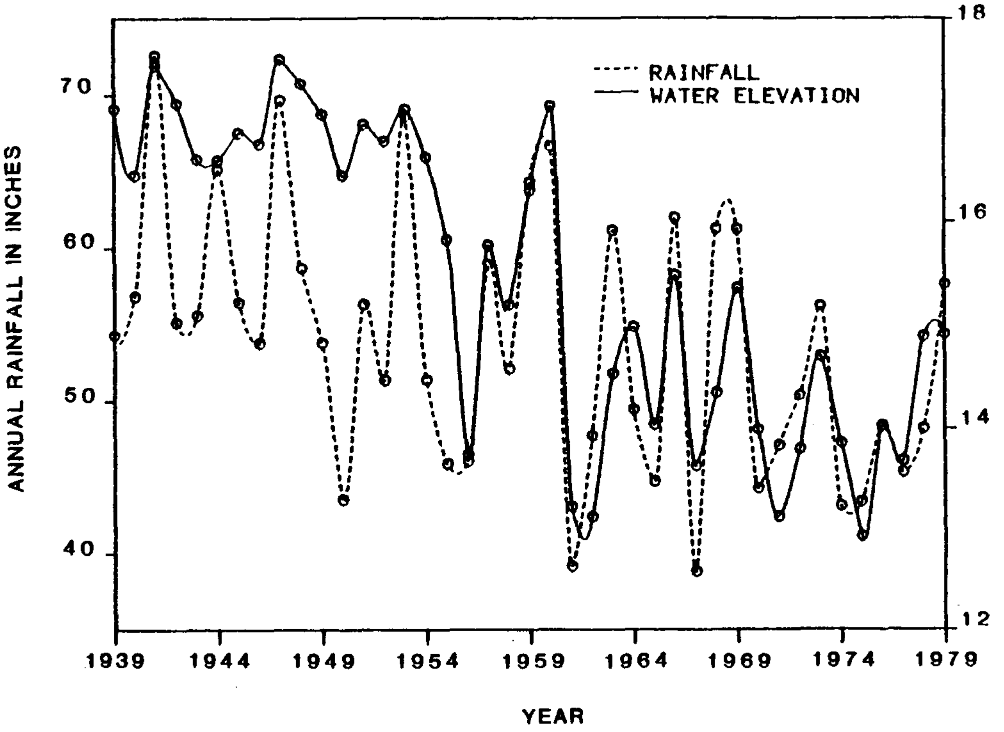


Figure 51. Variation in total annual rainfall and surface water elevation at U.S. 192 from 1939 - 1979.

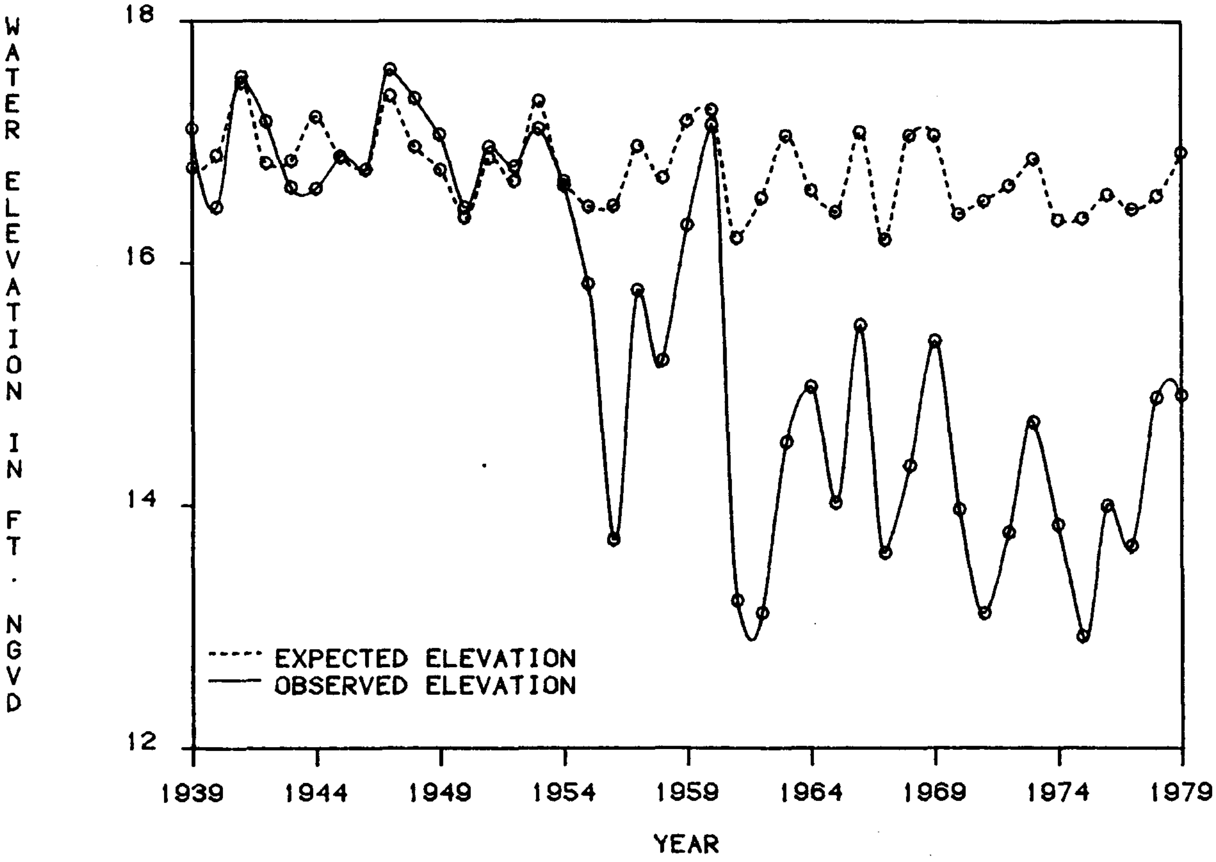


Figure 52. Observed and expected relationship between rainfall and surface water elevation at U.S. 192 from 1939 - 1979.

the recent period, and 2) the correlation between rainfall and surface water elevation is higher in the recent (r=0.82) than in the early (r=0.70) period (Figure 51). How important was this change relative to the decline in elevations expected due to reduced rainfall? Modeling of stages expected from recent period rainfall using the early period regression equation strongly suggests that the decline in rainfall was insufficient to have caused the decline in mean surface water elevation (0.74 m) and that in the absence of extensive floodplain alterations mean elevation would have changed very little (Figure 52). The abruptness of the change in the surface water-rainfall relationship suggests that a single event was the major cause of the decline in surface water elevations. This event may have been the destruction of the jams in about 1952. In the absence of their impounding effect it is reasonable to expect both reduced stages and increased correlation with rainfall. An intensifying factor in this time period would be completion of the major north-south canals south of Lake Hell 'n Blazes.

As with the water quality and sedimentation problems, plugging of the canals which rapidly drain the floodplain would reduce the water deficit of the floodplain wetlands. Elimination or extenuation of diversion of water to the coast and restoration of lost floodplain storage would further reduce the deficit. It is important, however, that additional storage be obtained through restoration of natural wetland habitat. Without some restoration of lost habitat it seems doubtful that the river system will support the remarkable abundance of fish and wildlife indicated by Howell (1932) and by the historical research of Cox et al. (1981).

LITERATURE CITED

Aggus, L. R. and G. V. Elliott, 1975. Effects of cover and food on year-class strength of largemouth bass. In: Black bass biology and management, Henry Clepper (Editor), Sport Fishing Institute, Washington, D.C., pp. 317-322.

American Public Health Association et al., 1975. Standard methods for the examination of water and wastewater (14th ed.), New York, 1193 pp.

Baker, J. A., L. A. Pugh III, and K. T. Kimball, 1977. A simple hand corer for shallow water sampling. Chesapeake Science, 18(2) 232-236.

Belanger, T. V., R. D. Arnone and C. J. Bore, 1982. Factors affecting production and dissolved oxygen levels in the Upper St. Johns River, Florida. Unpublished Manuscript, Dept. of Environmental Sciences Engineering, Florida Institute of Technology, Melbourne, 28 pp.

Belanger, T. V., S. D. Van Vonderen and T. J. Carberry, 1983. Analysis of selected water quality factors in the St. Johns River Basin, Report to the St. Johns River Water Management District. Florida Institute of Technology, Melbourne, Florida, 114 pp.

Brady, N. C., 1974. The nature and properties of soils. MacMillan Publishing Co., Inc., New York, 639 pp.

Brevard County Comprehensive Plan, 1981. Potable water sub-element, Brevard County, Florida, 112 pp.

Brown, D. W., W. E. Kenner, J. W. Crooks, and J. B. Foster, 1962. Water resources of Brevard County, Florida: Florida Geological Survey, Report of Investigations No. 28, 104 pp.

Canfield, D. E., 1981. Chemical and trophic state characteristics of Florida lakes in relation to regional geology. Center for Aquatic Weeds, University of Florida, Gainesville, 444 pp.

Carlson, R. E., 1977. A trophic state index for lakes. Limnol. Oceanogr., 22(2): 361-369.

CH_2M Hill, Inc., 1979. Assessment of water quality. Report for Deseret Ranches of Florida, Inc., Gainesville, Florida.

Cox, D. T., H. L. Moody, E. D. Vosatka, and L. Hartzog, 1976. D-J F-25 Stream investigations completion report, Florida Game and Fresh Water Fish Commission, Tallahassee, Florida, 848 pp.

Cox, D. T., E. D. Vosatka, H. L. Moody, and L. L. Conner, 1982. D-J F-33 St. Johns River fisheries resources completion report, Florida Game and Fresh Water Fish Commission, Tallahassee, Florida, 245 pp.

Crain, L. J., G. H. Hughes, and L. J. Snell, 1975. Water resources of Indian River County, Florida. Report of Investigation 80. Tallahassee, Florida, 75 pp.

Dineen, J. W., R. L. Goodrick, D. W. Hallett, and J. F. Milleson, 1974. The Kissimmee River revisited. In Depth Report, Central and Southern Flood Control District, West Palm Beach, Florida, Vol. 2, No. 2, 11 pp.

Eakins, J. D. and R. T. Morrison, 1978. A new procedure for the determination of Lead-210 in lake and marine sediments. J. Applied Radiation and Isotopes, 29: 531-536.

Fall, C., 1982. Water quality monitoring annual report, 1979-1981. Technical Report No. 17, St. Johns River Water Management District, Palatka, Florida, 51 pp.

Florida Game and Fresh Water Fish Commission, 1980. Public lake and stream survey, management-Lake Blue Cypress, Study No. 1, Job No. 3-B, 7 pp.

Florida Game and Fresh Water Fish Commission, 1981. An evaluation of the waterfowl potential on lands of the St. Johns River Water Management District lying between S.R. 60 and U.S. 192, 26 pp.

Gerry, L. R., 1982. The potential effects of water hyacinths on sediments in the Upper St. Johns River Basin. Unpublished Report, St. Johns River Water Management District, Palatka, Florida, 9 pp.

Herke, W. H. and G. Horel, 1958. Fishery evaluation of the U.S. Corps of Engineers proposed Water Storage Plan for the Upper St. Johns River. Project F-8-R, Florida Game and Fresh Water Fish Commission, Tallahassee, Florida, 36 pp.

Hortenstine, C. C. and R. B. Forbes, 1972. Concentrations of nitrogen, phosphorus, potassium and total soluble salts in soil solution samples from fertilized and unfertilized histosols J. Environ. Quality, 1(4): 446-449.

Howell, A. H., 1932. Florida bird life. Florida Department of Game and Fresh Water Fish Commission in Cooperation with the Bureau of Biological Survey, United States Department of Agriculture, Publishers Agents - Coward-McCann, Inc., New York, 579 pp.

Huber, W. C., P. L. Brezonik, J. P. Heaney, R. E. Dickinson, S. D. Preston, D. S. Dwornik, and M. A. DeMaio, 1982. A classification of Florida lakes. Water Resources Research Center, Publication No. 72, Dept. Env. Eng. Sci., Univ. Florida, Gainesville

Kratzer, C. R. and P. L. Brezonik, 1981. A Carlson-type trophic state index for nitrogen in Florida lakes. Water Resources Bulletin, 17(4): 713-715.

Kushlan, J. A. and D. A. White, 1977. Nesting wading birds in southern Florida. Florida Scientist, 40(1): 65-72.

Lowe, E. F., 1983. Distribution and structure of floodplain plant communities in the Upper Basin of the St. Johns River, Florida. St. Johns River Water Management District, Technical Report No. 26, 65 pp.

Marella, R. and B. Ford, 1980. Current and projected population of the St. Johns River Water Management District. Information Circular No. 8, St. Johns River Water Management District, Palatka, Florida, 30 pp.

Marella, R., 1982. Annual water use survey, Technical Report No. 27, St. Johns River Water Management District, Palatka, Florida, 97 pp.

Martin, D. B., L. J. Mengel, J. F. Novotny and C. H. Walburg, 1981. Spring and summer water levels in a Missouri River reservoir: Effects on age-0 fish and zooplankton. Trans. Am. Fish. Soc., 110: 370-381.

Nesbitt, S. A., J. C. Ogden, H. W. Kale, B. W. Patty, and L. A. Rowse, 1982. Florida atlas of breeding sites for herons and their allies: 1976-1978, U. S. Fish and Wildlife Service, Office of Biological Services, FWS/OBS-81/49, 449 pp.

Ploskey, G. R., 1982. Effects of water level changes on reservoir ecosystems, fish, and fishing, with guidance on reservoir operations to enhance fisheries. U. S. Department of the Interior, Fish and Wildlife Service, Fayetteville, AR., 83 pp.

Provasoli, L., 1969. Algal nutrition and eutrophication. In: Eutrophication: causes, consequences, correctives. National Academy of Sciences, Washington, D.C., pp. 574-593.

Ryan, T. A., Jr., B. L. Joiner, and B. F. Ryan, 1982. Minitab reference manual. Minitab Project, Statistics Dept., Penn. State Univ., University Park.

Sincock, J. L., 1958. Waterfowl ecology in the St. Johns River valley as related to the proposed conservation areas and changes in the hydrology from Lake Harney to Fort Pierce, Florida. Florida Game and Fresh Water Fish Commission, Tallahassee, Florida, 224 pp.

Smith, S. K., 1982. Pojections of Florida population by county, 1983 - 2020. Univ. Florida, Bureau of Economic and Business Research, Population Program, Bull. No. 60.

St. Johns River Water Management District, 1979. Upper St. Johns River Basin Surface Water Management Plan, Volume 1., 394 pp.

St. Johns River Water Management District, 1980. Upper St. Johns River Basin Surface Water Management Plan, Volume 2., 500 pp.

Stephens, J. C. 1974. Subsidence of organic soils in the Florida Everglades - a review and update. In: Gleason, P. J. (ed.), Environments of South Florida, Memoir 2, Miami Geological Society, Miami, Florida., 452 pp.

Stewart, N. E., D. L. Shumway, and P. Duodoroff, 1967. Influence of oxygen concentration on the growth of juvenile largemouth bass. J. Fish. Res. Bd. Can., 24(3): 475-494.

Sullivan, J. H., 1979. Effects of agricultural discharges in the upper St. Johns River. Unpublished Manuscript, Water and Air Research, Inc., P. O. Box 1121, Gainesville, Florida, 19 pp.

Tai, C. C. and D. V. Rao, 1982. Hydrologic change due to floodplain impoundment and encroachment by agricultural activities. In: Environmentally sound water and soil management, E. G. Kruse, C. R. Burdick, and Y. A. Yousef (Editors), American Society of Civil Engineers, New York, pp. 193-200.

Terry, R. E., 1980. Nitrogen mineralization in Florida Histosols. Soil Sci. So. Am. J., 44: 747-750.

U. S. Department of Commerce, 1982. 1980 National survey of fishing, hunting, and wildlife-associated recreation, 156 pp.

U. S. Department of Agriculture, Soil Conservation Service, 1974. Soil survey of Brevard County, Florida, 123 pp.

Velleman, P. F. and D. C. Hoaglin, 1981. Applications, basics and computing of Exploratory Data Analysis. Duxbury Press, Boston, Ma., 354 pp.

Wegener, W. and V. Williams, 1974. Fish population responses to improved lake habitat utilizing an extreme drawdown. Proceedings S.E. Assoc. Game and Fish Commission, 28: 144-160.

White, W. A. 1970. The geomorphology of the Florida peninsula. Florida Bureau of Geology, Geological Bulletin No. 51, 164 pp.