SOUTH FLORIDA WATER MANAGEMENT DISTRICT

VOLUME I
KISSIMMEE RIVER RESTORATION STUDIES



Establishing a Baseline: *Pre-restoration Studies of the Channelized Kissimmee River*

November 2005
Technical Publication ERA 432

Edited by:
Stephen G. Bousquin
David H. Anderson
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Photo Researcher: Monica L. Daeumler and Brent C. Anderson
Majority of Photos by: Patrick Lynch

Suggested Citation Format:

Bousquin, S. G., D. H. Anderson, G. E. Williams, and D. J. Colangelo, editors. 2005. Establishing a baseline: pre-restoration studies of the channelized Kissimmee River. South Florida Water Management District, West Palm Beach, Florida, USA. Technical Publication ERA #432.

Printed in the United States of America

Additional copies or CDs of this volume are available from the below listed address:

Kissimmee Division
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ESTABLISHING A BASELINE: PRE-RESTORATION STUDIES OF THE CHANNELIZED KISSIMMEE RIVER

CONTENTS

ACKNOWLEDGEMENTS

The editors would like to extend their thanks to Bonnie Rose for her assistance and coordination of the various maps used in these volumes; to Monica L. Daeumler and Brent C. Anderson for formatting both volumes and developing an associated literature database; to R. Harper Carroll IV, Rene Hartless, Kim Kanter, Jennifer Zimmerman, Christine Carlson, Lou Toth, and Jose Valdes, who have made significant contributions to the Kissimmee River Restoration Evaluation program; to former Kissimmee Division Directors Seán Sculley, Larry Gerry, Paul Whalen, and Patricia Strayer, and Jennifer Jorge, current Division Director, for program leadership and support. Previous versions of these chapters were improved by comments from an external review panel that included Drs. Arthur C. Benke, Arnold G. van der Valk, Clifford N. Dahm, James R. Karr, and Milton W. Weller. We appreciate the panel's encouragement to improve and publish these chapters.

CHAPTER 1

INTRODUCTION TO BASELINE STUDIES OF THE CHANNELIZED KISSIMMEE RIVER

Stephen G. Bousquin, David H. Anderson, David J. Colangelo, and Gary E. Williams

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The Kissimmee River in central Florida (Figure 1-1) is the subject of one of the largest river restoration projects in the world. The goal of the project is to restore ecological function and biological communities to much of the river and floodplain, primarily by restoring lost hydrologic drivers. The project is expected to affect over 150 km^2 of former floodplain, and will restore approximately 80 km^2 of floodplain wetlands and over 70 km of meandering river channel. The restoration effort has a projected cost of $578 million and will take an estimated 15 years to complete. With a project of this size and scope, evaluation of success is of considerable interest. This volume collects the results of studies conducted prior to the start of restoration of the Kissimmee River to establish a benchmark for comparison with the restored system. These baseline studies will be used to evaluate changes resulting from the restoration, and ultimately, to evaluate the success of the project in meeting its primary goal, restoration of ecological integrity.

As part of the Central & Southern Florida (C&SF) flood control project, canal C-38 was excavated along the entire length of the Kissimmee River floodplain. The canal was designed to provide a high level of flood protection for surrounding communities and agricultural interests after a number of disastrous hurricanes and floods in the early half of the 20^{th} century. The primary flood protection strategy in the design of C-38 was to contain all flow that the river and floodplain had formerly carried. The canal effectively eliminated flow in the river and ended seasonal overbank flow and inundation of the floodplain. This highly successful engineering project was decried for its environmental impacts even before it was completed in 1971.

The project resulted in the loss of almost 8000 ha of wetlands; drastic declines in bird, fish, and other animal populations that depended on the wetlands; and substantial reductions in water quality. Initial studies explored ways to restore some portion of the river and floodplain while retaining the level of flood protection provided by the C&SF project. Modeling and evaluation in the early stages of feasibility planning indicated that adequate flood protection could be sustained by a combined strategy of property acquisition and backfilling of over one-third of the canal's 56 mile reach. The main engineering/construction efforts of the restoration project are to backfill a substantial portion of C-38, which bisects the Kissimmee River's floodplain, and to recarve and reconnect disconnected (remnant) river channels where necessary to restore continuity to the river channel. These efforts, when coupled with system operation strategies, will result in the restoration of flow to the river and seasonal inundation to the floodplain. Reestablished flow and inundation are expected to restore ecosystem function, and provide the impetus for recovery of native communities.

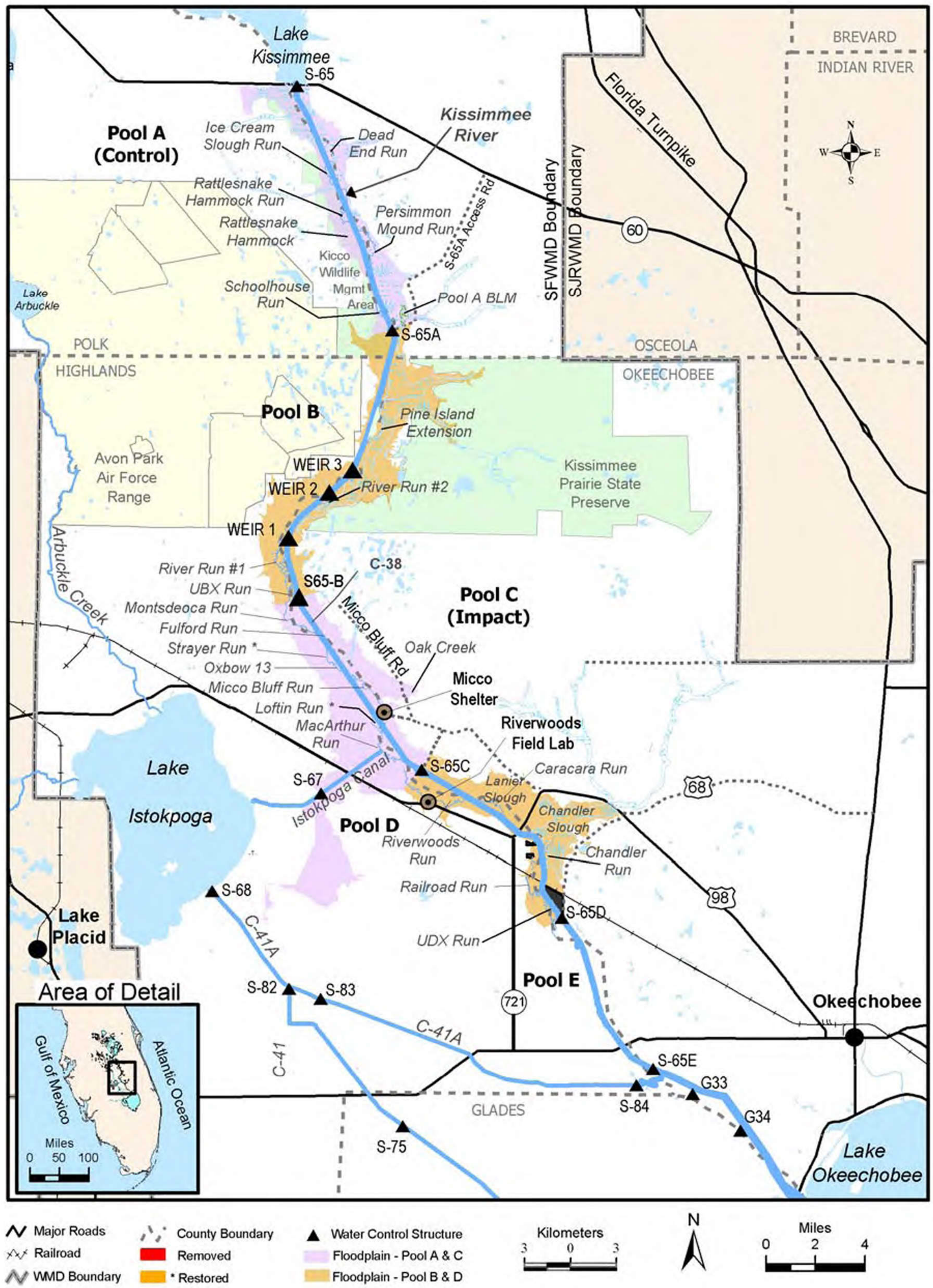

Figure 1-1. Map of the Kissimmee River from Lake Kissimmee to Lake Okeechobee.

This volume presents a diverse set of ecological studies designed and conducted specifically for the purpose of restoration evaluation. The scope of data collection in these studies has rarely been implemented for a channelized river. Primary goals of the volume are to collect what has been learned about the impacts of channelization on the river ecosystem and to establish a baseline for future comparisons with post-restoration data. In addition to presentation of baseline data, most of the chapters also predict specific responses of biological or abiotic attributes of the restored river and floodplain based on estimates of conditions prior to channelization. A companion volume (Anderson et al. 2005) summarizes these predictions in the form of restoration performance measures called expectations.

The specific objectives of this chapter are to provide:

(1) a brief overview of the history of impacts to the Kissimmee River,
(2) a summary of the plan to restore the river,
(3) a presentation of the approach and logic of the Kissimmee River Restoration Evaluation Program (KRREP),
(4) an overview of the 13 baseline chapters, each of which encompasses one or more studies conducted for the restoration evaluation program.

HISTORICAL BACKGROUND

Regional Setting

The Kissimmee River watershed is located in south-central Florida (Figure 1-1). The watershed is typically subdivided into the upper basin, which encompasses 4135 km^2 and consists of approximately 24 lakes varying in size from 0.5 to 152 km^2; and the 1731 km^2 lower Kissimmee Basin, located between Lake Kissimmee and Lake Okeechobee, which contains the Kissimmee River and tributaries. The river and floodplain slope to the south from an elevation of 15.5 m at Lake Kissimmee to 4.6 m at Lake Okeechobee (approximately 0.07 m km^{-1}) (Koebel 1995).

The regional climate is humid sub-tropical, with nearly equal-length wet and dry seasons and an average yearly rainfall of 121 and 114 cm in the upper and lower basins, respectively (Figure 1-2). Air temperature ranges from 5° C to 30° C (Figure 1-3). Soils of the lower Kissimmee basin consist mostly of the Manatee-Delray-Okeelanta soil association (McCollum and Pendleton 1971). These soils typically have a surface layer high in organic matter content overlying mineral subsoil (Toth et al. 1995).

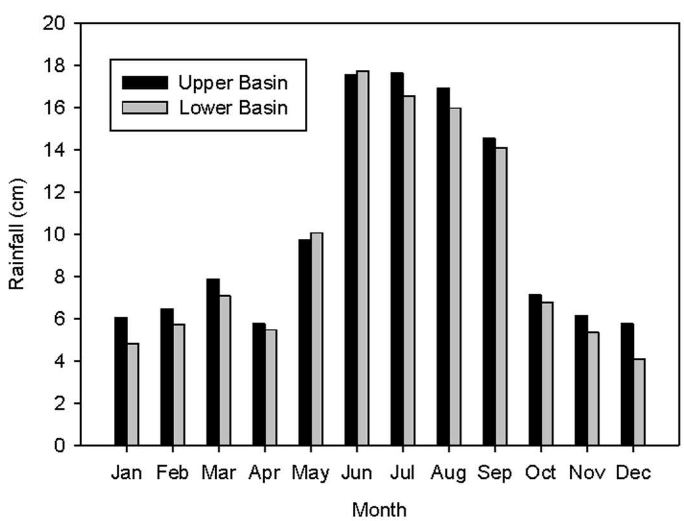

Figure 1-2. Average monthly rainfall in the upper and lower Kissimmee Basins (1971-2000).

Early Changes in the Kissimmee River Basin

A number of events altered the Kissimmee River Basin and affected conditions within the river (Table 1-1). These events began with the Seminole Wars, which opened the basin for settlement by European-Americans. The Second Seminole War (1835 - 1842) ended with the Seminoles confined to a temporary reservation that included the Peace River, the head of Lake Istokpoga and the Kissimmee River southward to Lake Okeechobee (Tebeau 1971). During this war, the military made several expeditions into the basin including a canoe trip up the Kissimmee River and around some of the lakes (Preble 1945). Fort Gardiner was constructed between Lake Hatchineha and Lake Kissimmee in the upper basin and Fort Basinger was constructed in the lower basin. The conclusion of the Third Seminole War (1855 - 1858) pushed the Seminoles south of Lake Okeechobee into the Everglades and the Big Cypress area, and opened the Kissimmee Basin to settlement. Ranchers and farmers settled in the basin and began to drain swampland, which further opened the area to development.

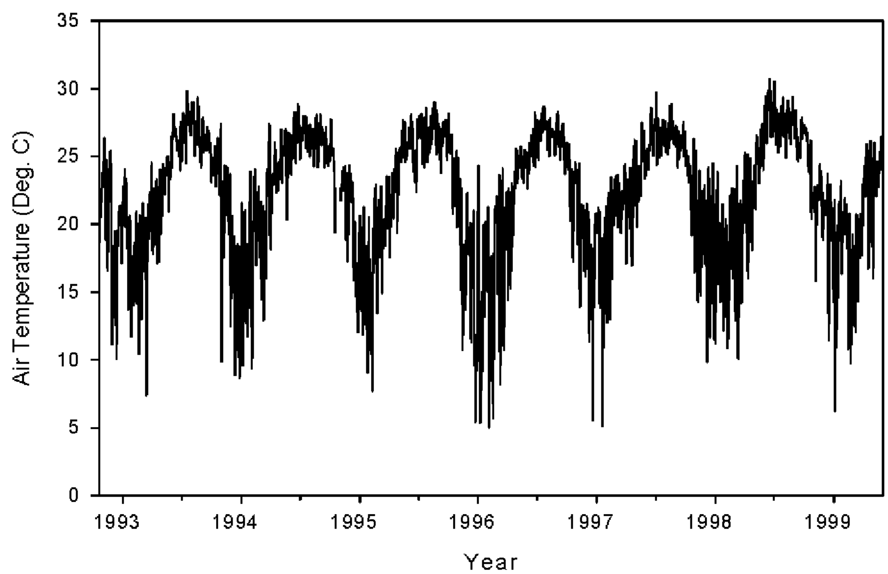

Figure 1-3. Mean daily temperature at water control structure S-65C from October 1992 - June 1999 (SFWMD DBHYDRO Database).

A major factor in the development of the basin was the passage by the U. S. Congress of the Swamp and Overflowed Land Act of 1850. This act allowed the state legislatures to transfer the ownership of swamp and overflowed lands to private entities with the stipulation that proceeds be used to reclaim the land through drainage and levee projects (Blake 1980). Hamilton Disston, a wealthy businessman from Pennsylvania and co-heir to the Disston Saw Company, was interested in draining wetlands to facilitate agricultural and residential development. In 1881, he negotiated a contract with the State of Florida's Internal Improvement Fund that would allow him to keep half of the land that he drained in and around Lake Okeechobee and the Kissimmee River. To conduct this drainage project, Disston formed the Atlantic and Gulf Coast Canal and Okeechobee Land Company and became its largest stockholder.

Between 1881 and 1884, the Gulf Coast Canal and Okeechobee Land Company dredged a canal to connect Lake Okeechobee with the Caloosahatchee River, the Southport Canal to connect Lake Tohopekaliga and Lake Cypress, and the St. Cloud Canal between Lake Tohopekaliga and East Lake Tohopekaliga. Additional dredging of and snag removal from existing river channels and lakes created a navigable waterway between St. Cloud on East Lake Tohopekaliga to Fort Myers on the Gulf Coast. As a result of this project, water levels in the Kissimmee Upper Basin dropped (U. S. Army Corps of Engineers 1991) and the area was opened to steamboat traffic. To maintain a navigable channel, clearing and snagging operations were conducted along the Kissimmee River. A 1901 map constructed by the U. S.

Army Corps of Engineers (Map of the Kissimmee River from Lake Kissimmee to Lake Okeechobee, surveyed under the direction of Captain T. H. Rees), shows a number of channel remnants that were probably cut-off by Disston's dredging activities. Commercial steamboat traffic between the towns of Kissimmee and Fort Myers on the coast began in 1885 and continued into the 1920s. Steamboats as large as 75 feet in length could now carry supplies from the coast to settlers in the interior of the state, and products such as oranges, hides, resin, wood, fish, and turpentine could be transported on the return trip.

In 1902, Congress authorized the Corps of Engineers to construct a navigation channel three feet deep and 30 feet wide extending from the city of Kissimmee to Fort Basinger in what is now Pool D. A similar channel was constructed in Istokpoga Creek, which connects Lake Istokpoga to the Kissimmee River. These projects were completed in 1909. The newly dredged channels were used until the 1920s when railroads became the primary transport for commercial products. The last federal navigational maintenance of these channels occurred in 1927 (U. S. Army Corps of Engineers 1991).

By 1938, the lower reach of the Kissimmee River was modified during construction of the 34 foot tall Herbert Hoover Dike around the south shore of Lake Okeechobee. This flood control project was initiated in response to extreme flooding and loss of life during the September 1928 hurricane (Blake 1980). A minor feature of this project was the construction of a 6.5 mile levee along the east side of the Kissimmee River below what is now Pool E (U. S. Army Corps of Engineers 1956). The material used to construct the levee was excavated from an eight mile long borrow canal. When part of the flow to Lake Okeechobee was diverted through the borrow canal, it became known as Government Cut and the section of river channel that was cut off became known as Paradise Run. Paradise Run continued to receive some flow and fluctuating water levels (Perrin et al. 1982).

Channelization and the Central & Southern Florida Flood Control Project

In response to severe droughts and hurricanes which caused extensive, prolonged flooding in the Kissimmee Basin in the mid 1940's, Congress authorized the Central & Southern Florida (C&SF) flood control project in 1948. This project would increase flood protection, allow for better control over the flow of water throughout the upper and lower Kissimmee Basins, and significantly alter the hydrology of the area. The Kissimmee River portion of the C&SF project was authorized by Congress in 1954 and was designed between 1954 and 1960. The river was channelized between 1962 and 1971. The project included excavation of a 90 km long, 10 m deep, 100 m wide flood control canal (C-38) that bisected the Kissimmee River floodplain from Lake Kissimmee to Lake Okeechobee, and construction of six water control structures (S-65 - S-65E) along the length of the canal (U. S. Army Corps of Engineers 1991).

In addition to channelizing the Kissimmee River, the C&SF project enlarged some of the canals connecting lakes in the upper basin, and installed water control structures to regulate lake levels and the movement of water between the lakes. The S-65 structure installed at the outflow from Lake Kissimmee regulated the releases to the Kissimmee River.

The C&SF project also modified the routing of water between Lake Istokpoga and the Kissimmee River. Prior to the C&SF project, Istokpoga Creek connected the lake with the river. By 1949, a local drainage district had excavated the Istokpoga Canal parallel to Istokpoga Creek and installed a sheet pile weir (G-85 structure) to regulate flow to the river. The principal outflow from the lake continued to involve overflowing the southwestern end, creating sheetflow across the Indian Prairie toward Lake Okeechobee. After the C&SF project, the principal outflow from Lake Istokpoga was through the S-68 structure at the southeastern end of the lake. This structure discharged into the C-41 canal, which branched to form the C-41A and C-40 canals. The C-41A canal discharged directly into the Kissimmee River just south of the S-65E structure.

Impacts of the Central and Southern Florida Flood Control Project

Following channelization of the Kissimmee River, the lower basin was transformed into a series of impounded reservoirs (pools) separated by water control structures. The 2-3 m deep original river channel was intersected by C-38, leaving stagnant, remnant river channel sections on either side of the canal. Channelization stabilized water levels, permanently inundating the southern end of each pool and permanently draining approximately two thirds of the floodplain at the northern end. Discharges, which under pre-channelization conditions were conducted by both the river channel and floodplain, were now contained entirely within the canal. Disconnected remnant river channels received virtually no discharge.

Due to the regulation schedules of the Kissimmee Upper Basin lakes, there were frequent episodes of zero discharge into the lower basin (Anderson and Chamberlain 2005, Toth et al. 1995).

Table 1-1. Timeline of events that led to changes in the Kissimmee Basin.

Year	Changes	Source
1837	Fort Gardiner built.	
	Fort Basinger built on the Kissimmee River.	
Late 1830s	Fort Kissimmee constructed.	
1856	Yates family is first family to settle in Shingle Creek.	Hetherington 1980
1881	February 26, Hamilton Disston contracts with the State of Florida to drain lands in exchange for ownership of half the reclaimed land.	
1882	January – Disston's company completes canal to connect lake Okeechobee with the Caloosahatchee River.	
	July – Disston's company completes Southport Canal between Lake Tohopekaliga and Lake Cypress .	
1883	January – Disston's company begins work on St. Cloud Canal between Lake Tohopekaliga and East Lake Tohopekaliga.	
1884	September – St. Cloud Canal completed. Over a 30 day period, water levels drop approximately 3 feet exposing a sand beach between the cypress and the new waterline.	
1883	Settlement of Allendale becomes Kissimmee City.	Mueller 1966
1884	Canal from Lake Tohopekaliga to East Lake Tohopekaliga completed; East Lake Toho stages fall 36 inches in 30 days. Canal from Lake Tohopekaliga to Lake Cypress completed. Kissimmee River was streamlined by cutting off number of bends. Snag boat in operation on the river.	Mueller 1966
1885	June 5 - Regular steamship service from Fort Meyers to Kissimmee begins.	Casselberry 1984
1902-09	Corps of Engineers completes navigation project to dredge a 3 foot navigation channel in the Kissimmee River to Istokpoga Creek; snag removal operations.	USACE 1969
1921	Completion of railroad to Fort Meyers brings steamship era to an end.	Casselberry 1984
1927	Last Federal maintenance for Kissimmee River navigation authority.	
	Last steam boat operation on the upper basin lakes.	
1938	During the Herbert Hoover Dike Project for Lake Okeechobee, Corps of Engineers creates a 6.5 mile levee from Lake Okeechobee along the east side of the Kissimmee River. Part of the flow was diverted through the eight mile barrow canal. The canal became known as Government Cut and the remnant river channel as Paradise Run.	U. S. Army Corps of Engineers. 1969
	Istokpoga Creek dredged to create Istokpoga Canal.	
1948	1948 Flood Control Act authorizes Central and Southern Florida Flood Control Project.	
1962-71	Excavation of the C-38 canal.	Abtew 1992
1963	S-59 installed to regulate outflow from East Lake Tohopekaliga.	Guardo 1992
1963	S-61 installed to regulate outflow from Lake Tohopekaliga.	Guardo 1992
1964	S-65 installed in August to regulate the outflow from Lake Kissimmee.	Guardo 1992
1965	Installation of the S-68 on Lake Istokpoga in Dec 14.	
1970	C&SF construction completed in the upper basin lakes and interim operating schedules adopted for water control structures.	USACE 1996
1971	Lake Tohopekaliga drawdown (Feb-Nov) 1971 Governor's Conference On Water Management recommends	Dierberg and Williams 1989

Year	Changes	Source
	restoration of the river.	
1976	Adoption of regulation schedules outlined in Report to the Governing Board on Regulatory Levels in the upper Kissimmee Basin.	USACE 1996
1977	Lake Kissimmee drawn down (Jan-Dec).	
1978-85	First Federal Feasibility Study for the Kissimmee River restoration.	
1979	Lake Tohopekaliga drawdown (Jan-May).	
1982	In April revised regulation schedules were implemented.	USACE 1996
1983	Coordinating Council recommends the backfilling plan.	
1984	Sheet pile Weir 3 installed (Oct 1 – Nov 6) for Pool B Demonstration Project.	Toth 1991
1984-90	Kissimmee River Demonstration Project.	
1985	Sheet pile Weir 2 installed (Feb 5 – Mar 16) for Pool B Demonstration Project.	Toth 1991
	Sheet pile Weir 1 installed (May 2 – Jun 9) for Pool B Demonstration Project.	Toth 1991
	Pool B stage fluctuation initiated on October 28 . (Note: Obeysekera and Loftin 1990 use September 1985).	Toth 1991
1987	Lake Tohopekaliga drawdown (Jan-Sep) with muck removal.	
1988	Kissimmee River restoration symposium adopts the ecological integrity goal.	
1990-95	Second Federal Feasibility Study recommended the level II backfilling plan	
1990	Drawdown in East Lake Tohopekaliga.	
1992	Water Resources Act authorizes the Kissimmee River restoration project.	
1994	Drawdown in Lake Jackson. Project Cooperative Agreement between the District and the U. S. Army Corps of Engineers. Test backfill constructed and high flow tests.	
1995-99	Baseline sampling conducted.	
1995	Drawdown in Lake Jackson.	
1996	Drawdown in lake Kissimmee.	
1997	Drawdown in Lake Jackson.	

The physical effects of channelization, including alteration of the system's hydrologic characteristics, drastically reduced the extent of floodplain wetlands, and severely degraded fish and wildlife resources of the basin (U. S. Army Corps of Engineers 1991). Approximately 8000 ha of floodplain wetlands were drained, covered with spoil material, or converted into canal (Carnal and Bousquin 2005). No-flow regimes in remnant channels encouraged extensive growth of floating vegetation, which impeded navigation (Toth 1990). Senescence and death of encroaching vegetation produced large amounts of organic matter that covered the shifting sand substrate, greatly increasing the biological oxygen demand of the system (Anderson et al. 2005).

The effects of channelization and other disturbances, such as invasion by exotic vegetation and grazing, have significantly altered plant communities of the river channel and floodplain. Wetland types such as broadleaf marsh, wet prairie, and wetland shrub communities that were dominant prior to channelization were replaced by pasture and other upland vegetation (Carnal and Bousquin 2005).

By the late 1970s, floodplain use by wintering waterfowl had plummeted. Diverse and abundant wading bird populations declined and were largely replaced by cattle egret (*Bubulcus ibis*), a species generally associated with upland, terrestrial habitats (Perrin et al. 1982).

The largemouth bass (*Micropterus salmoides*) fishery was decimated, while fish species tolerant of low dissolved oxygen, such as Florida gar (*Lepisosteus platyrhincus*) increased (Perrin et al. 1982). Aquatic invertebrate taxa of the channelized system were typical of those found in lakes and reservoirs rather than riverine systems (Harris et al. 1995). Stabilized water levels greatly reduced river-floodplain interactions, disrupting critical food web linkages dependent on seasonal flooding and protracted floodplain recession rates (Harris et al. 1995). Other impacts on the Kissimmee River Basin that may or may not be related to channelization include an increase in human population growth (Figure 1-4); changes in land use (South Florida Water Management District 2000); and invasion by exotic species of plants, fish, birds, and invertebrates (Ferriter et al. in press).

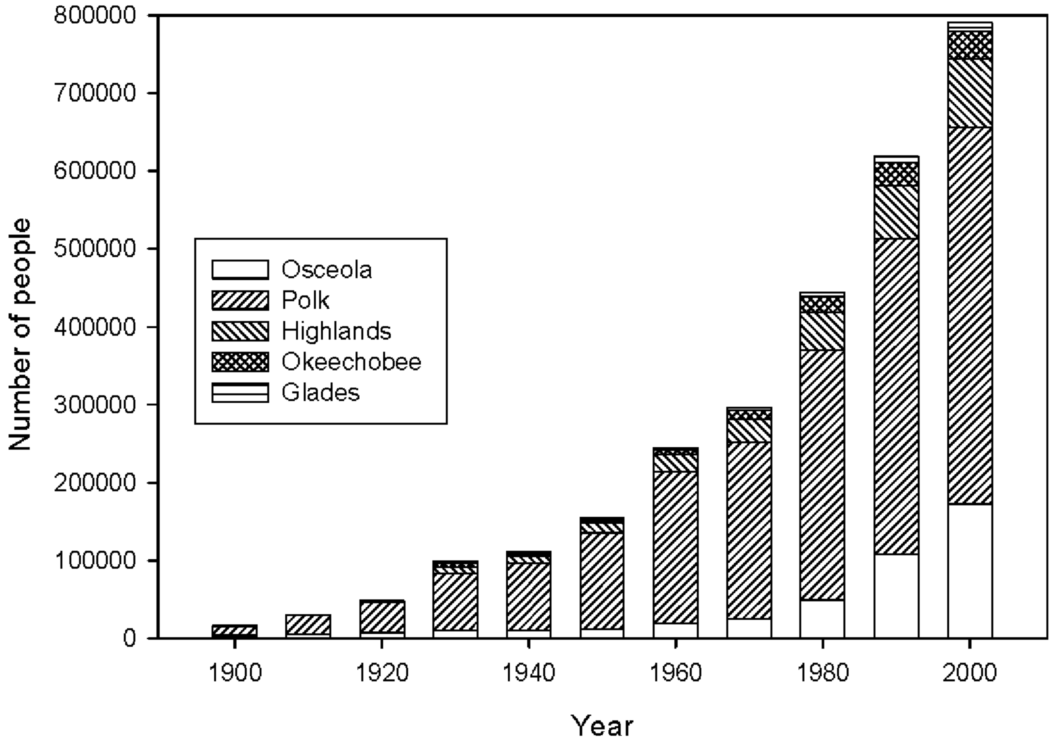

Figure 1-4. Population size for five counties that overlap the Kissimmee Basin. Population estimates were obtained from the State of Florida web page.

THE RESTORATION PROJECT

Mandate and Authorization for Restoration

Even before construction of the C&SF Project began, its potential for ecological damage was recognized (USFWS 1959). During construction (1962-1971), a grassroots movement formed with the goal of restoring the Kissimmee River (Loftin et al. 1990, Koebel 1995). In 1972, just one year after construction was completed, the Central and Southern Florida Flood Control District (now known as South Florida Water Management District) held the first public hearing on the potential for restoration of the Kissimmee River (Loftin et al. 1990). Concern over the effects of the Kissimmee Project eventually led the Florida legislature to pass the Kissimmee River Restoration Act (Section 373.1965, Florida Statutes), which mandated the creation of the Kissimmee River Coordinating Council (KRCC). The KRCC was tasked with developing measures to improve water quality in the Kissimmee River Valley and Taylor Creek/Nubbin Slough Basin, with specific goals that included restoration of natural seasonal water level fluctuations in Upper Basin lakes and the Kissimmee River floodplain, and re-creation of conditions that would lead to reestablishment of wetland flora and fauna. Following the creation of the KRCC, three restoration evaluation and planning studies (the first federal feasibility study, the Kissimmee River Demonstration Project, and the second federal feasibility study) were initiated by the U. S. Army Corps of Engineers and/or the South Florida Water Management District (SFWMD).

The first federal feasibility study (1978 - 1985) was authorized via resolutions of the Committee on Public Works and Transportation of the U. S. House of Representatives, and the Committee on

Environment and Public Works of the U. S. Senate on April 25, 1978. The primary purpose of the study was to "evaluate the feasibility of modifying the existing flood control system for purposes of improving water quality and enhancing fish and wildlife resources" (U. S. Army Corps of Engineers 1985, Koebel 1995). As a result of the study, various restoration plans were developed.

The first feasibility study concluded that the best opportunities for meeting the above goal involved (1) pool stage manipulations, (2) restoration of wetland values to Paradise Run near Lake Okeechobee, and (3) implementation of best management practices (U. S. Army Corps of Engineers 1985). The study did not endorse federal support for these projects because they did not meet federal water and land resource planning guidelines requiring a net contribution to the nation's economic benefit. It did recommend that state and local interests use the report to develop a framework for long term management of the Kissimmee Basin. Eventually, because of overwhelming public support for the plans that called for backfilling C-38 over other restoration alternatives, the KRCC endorsed backfilling as the preferred option (Koebel 1995).

The Demonstration Project (1984-1990) was initiated by the SFWMD to assess the feasibility of the backfill plan. This project had four components: (1) reestablishing seasonal floodplain inundation in the project area, (2) construction of three navigable weirs along C-38 to divert flow through remnant river channel sections, (3) creating a flow-through marsh system, and (4) performing hydrologic and hydraulic modeling studies (Toth 1991, Koebel 1995). Additionally, physical and biological monitoring was performed to evaluate the feasibility of recreating the river's pre-channelization structure and function. Results of the Demonstration Project indicated increased floodplain inundation and reestablishment of some of the biological communities that existed before channelization, suggesting that restoration of the structure and function of the system was feasible (Toth 1991, 1993).

The purpose of the second federal feasibility study (1990 - 1991) was to determine how the backfilling plan would be implemented and how much federal participation would be granted. The 1986 Water Resources Development Act authorized the U. S. Army Corps of Engineers to modify existing Corps projects to enhance environmental quality in the public interest and to calculate the benefits of such enhancements as being equal to other costs (Woody 1993). This change removed the barrier that prevented the first feasibility study from recommending federal support for the restoration project. The results of this study, which included extensive value engineering, led to the adoption of the modified Level II Backfilling Plan as the recommended restoration plan. This plan called for continuous backfilling of approximately 47 km of C-38 from the middle of Pool B to Pool E and included removal of structures S-65B, S-65C, and S-65D as well as excavation of river channels that had been destroyed during C-38 construction (U. S. Army Corps of Engineers 1991, Koebel 1995). The plan was further modified following a recommendation in 1992 by Assistant Secretary of the Army, Nancy P. Dorn, to eliminate federal participation in the removal of S-65D and backfilling in Pool E (Assistant Secretary of the Army 1992). Following this recommendation, the geographic scope of backfilling was reduced to approximately 35 km of backfilling of C-38 in Pools B-D. Plans for the removal of S-65D were also discontinued.

Implementation of the Restoration Plan

In 1992, the U. S. Congress jointly authorized ecosystem restoration of the Kissimmee River and the Kissimmee River Headwaters Revitalization Project (Headwaters) via the Water Resources Development Act. Headwaters was authorized primarily because modifications of the Upper Kissimmee Basin were necessary for successful restoration of the Kissimmee River. Specifically, Headwaters was designed to provide the Upper Basin storage and flow characteristics necessary to meet or exceed the needs of the (KRRP), while increasing the quality and quantity of wetland habitat in littoral zones of the Upper Basin lakes (U. S. Army Corps of Engineers 1996). The 1994 cost-sharing Project Cooperative Agreement between the U. S. Army Corps of Engineers and the SFWMD combined the restoration project and Headwaters into the single restoration entity called the Kissimmee River Restoration Project. Because of the large scale of the KRRP and the lack of other similar restoration projects to use as templates, a pilot "test fill" project of C-38 was initiated in April, 1994. The test fill project involved filling a 330 m section of C-38 to evaluate fill consolidation and stability, construction methodologies, water quality impacts, and subsequent colonization of backfill by vegetation (Koebel et al. 1999). This project demonstrated that the planned construction methodology would produce stable soils in the area of backfilling without causing long-term impacts to downstream water quality (Koebel et al. 1999).

Construction of the KRRP was divided into four major phases, the first of which was initiated in 1999. Phase I included removal of the S-65B structure, and backfilling of a small portion of lower Pool B and most of Pool C. Phase II/III will remove S-65C, and will backfill the remainder of Pool C and most of Pool D. Phase IV, which will backfill a section of Pool B north of the Phase I area, is scheduled for completion in 2012. The new headwaters regulation schedule will be implemented following completion of KRRP/Headwaters, which is scheduled for 2010 (Figure 1-5).

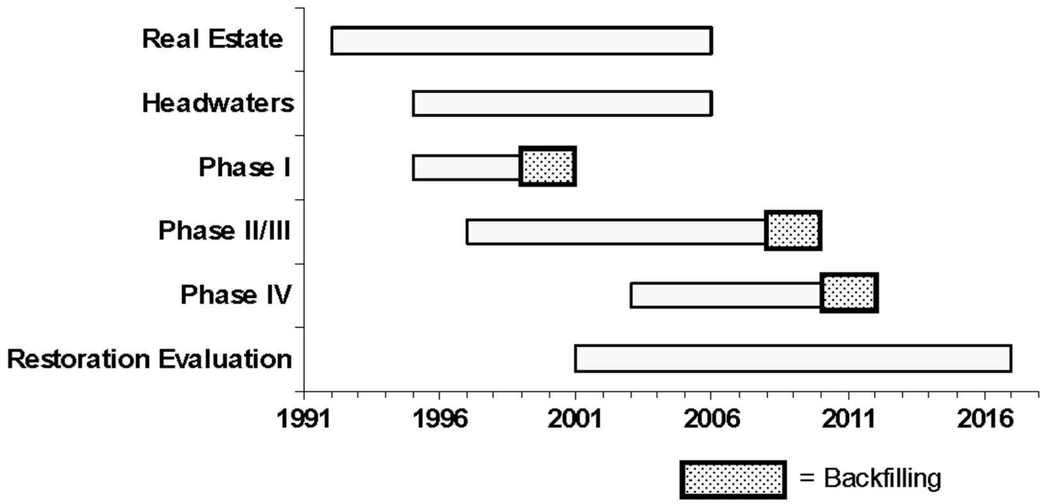

Figure 1-5. Timeline for completing major components of the KRRP including real estate acquisition, headwaters revitalization, Phases I through IV of backfilling and construction, and restoration evaluation.

THE RESTORATION EVALUATION PROGRAM

The Goal of Ecological Integrity

The goal of the KRRP is to restore ecological integrity to the Kissimmee River and its floodplain. Ecological integrity is a characteristic of ecosystems that are ". . . capable of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region" (Toth 1990 after Karr and Dudley 1981). The restoration project is designed to achieve this goal by reestablishing the natural hydro-geomorphic drivers of the floodplain river ecosystem that were disrupted by channelization. These ecosystem drivers are expected to facilitation of complex ecological interactions that will lead to changes in nutrient cycling and dissolved oxygen levels, and precipitate ecological processes such as re-establishment of a pre-channelization vegetation mosaic, colonization or expansion of aquatic invertebrates, and use and colonization of habitats by native fish and birds. This cascade of abiotic and community assembly responses to restoration is expected to continue even after the completion of the construction phases of the restoration, ultimately resulting in a functional and resilient ecological system.

The Mandate for Monitoring: The Integrated Feasibility Report

The Final Integrated Feasibility Report (IFR) for the KRRP (U. S. Army Corps of Engineers 1991) identified the need for a program to evaluate the success of the restoration project. Restoration evaluation is also specified in the 1994 cost-sharing Project Cooperative Agreement between the U. S. Army Corps of Engineers and the SFWMD, and this agreement assigns responsibility for evaluation to SFWMD.

As specified in the IFR, ecological monitoring should address a suite of goals for monitoring. These goals were to a) assess impacts during the phased construction project and future management of the system, b) signal potential needs for adaptive management, c) assess applicability of the restoration approach to other projects, and d) provide information needed to evaluate project success. These are discussed below.

Construction Monitoring

Two studies were designed specifically to monitor construction impacts. River turbidity will be monitored during all phases of restoration construction, as will nesting territories of Audubon's crested caracara, which is classified as threatened by the U. S. Fish & Wildlife Service. Other studies (Table 1-2) monitored initial responses during or shortly after construction.

Adaptive Management

Adaptive management refers to the use of monitoring results to guide resource management. In the context of KRREP, failure to meet the expectations associated with evaluation project metrics may initiate further study , which may ultimately lead to changes in the operational or construction activities believed to be inhibiting recovery. While the predicted response of the system is the natural recovery of ecological integrity, the U. S. Army Corps of Engineers (1991) report recognized that variable inflow from the headwaters of the Kissimmee River must be maintained in order to initiate and sustain the expected biological and ecological responses. Therefore, a primary function of the restoration evaluation program is providing a feedback loop to guide water control operations and management efforts throughout the watershed.

Applications to Other Restoration Projects

The IFR recognized that the KRRP should yield invaluable insights for future large scale river restoration projects. Applications to other restoration projects are addressed through ongoing comprehensive documentation of the evaluation program. The KRRP has been noted as one of the most well-documented restoration projects in the nation (Bernhardt et al. 2005). Regular publications allow for independent peer review of the evaluation program and help ensure that project science is of the highest quality. The results of the Kissimmee River Restoration Demonstration Project were documented in a SFWMD technical bulletin (Toth 1991), a series of papers in peer-reviewed journals (Toth 1993, Toth et al. 1993, Toth et al. 1998), and in a symposium volume (Loftin et al. 1990). An issue of the journal *Restoration Ecology* (1995, Vol. 3, No. 3) was dedicated to the restoration project. It included papers on the history of the project (Koebel 1995) and conceptual models for major components of the ecosystem including habitat/vegetation (Toth et al. 1995), aquatic invertebrates (Harris et al. 1995), fish (Trexler 1995), water birds (Weller 1995), and the overall ecosystem (Dahm et al. 1995). A pair of separately-published papers outlined the conceptual framework for restoration evaluation (Anderson and Dugger 1998) and the process used to develop restoration expectations to evaluate the success of the project (Toth and Anderson 1998). The current volume documents the baseline-period studies, and a companion volume documents the restoration expectations for the restoration project (Anderson et al. 2005). As the project progresses, other publications will document specific phases of the project, such as impacts during construction for Phase I backfilling (Colangelo and Jones 2005). Annual updates on the project will be included in the South Florida Environmental Report (e.g., Williams et al. 2005). Kissimmee River Restoration Evaluation Program staff presentations at national and regional conferences provide the science community with timely updates on system responses (Anderson 2002, 2003, 2004; Bousquin 2003a, 2003b, 2004; Colangelo and Jones 2000, 2004; Carnal and Bousquin 2003; Colangelo 2003, 2004; Jones 2003a, 2003b; Jones and Colangelo 2003; Koebel 2003; Williams 2004).

The restoration literature suggests that the approach taken by the Kissimmee River Restoration Project has influenced the conceptualization and planning of other river restoration projects (Palmer et al. 2005, Bernhardt et al. 2005). The KRRP was used as a detailed case study of a river restoration project (National Research Council 1992), to illustrate the application of ecological understanding to a large restoration project (MacMahon 1998), as an example of restoration at the landscape level for a general ecology text (Molles 1999), and to demonstrate the importance of replicating natural disturbance regimes such as flooding for ecosystem management (Dodds 2002). The KRRP is frequently used in university courses as an example of ecological restoration (D. H. Anderson, SFWMD, personal communication).

Table 1-2. Relationship of baseline studies to monitoring program components identified in the feasibility report.

	Ecology/Fish and Wildlife						Hydraulics	Sediment	Stability
Baseline Study	Birds	Fish/ Fisheries	Habitat	Water Quality	Threatened/ Endangered Species	Ecosystem Function			
Hydrology			X				X		
Geomorphology			X					X	X
Water Quality			X	X					
Dissolved Oxygen			X	X					
River Channel Vegetation			X						
Vegetation Classification			X						
Vegetation Mapping			X						
Floodplain Vegetation			X			X			
Algae			X			X			
Invertebrates						X			
Fish		X				X			
Reptiles and Amphibians						X			
Birds	X				X				

Much of our knowledge about river restoration of channelized streams is derived from smaller-scale projects, or from streams affected by dissimilar channelization than that found on the Kissimmee River. Extensive documentation of the KRRP will yield valuable insights for future restoration design and implementation.

Restoration Evaluation

Restoration evaluation was to take place primarily through monitoring. The U. S. Army Corps of Engineers (1991) report listed four programs as "necessary basic components" of the monitoring program, to be conducted during and after restoration construction. These included monitoring of (1) ecological (usually referred to as "fish and wildlife"), (2) hydraulic, (3) sedimentation, and (4) stability attributes. The IFR clearly placed all of these components in an ecological context.

Ecological Monitoring. The IFR recognized that restoration of ecological integrity necessitated evaluation of multiple indicators. It specifically identified water quality, habitat, ecological (usually called "fish and wildlife"), birds, fish/fisheries, threatened and endangered species, and ecosystem function as components of a comprehensive restoration monitoring program for the Kissimmee River. These components and others have been integrated into the KRREP (Table 1-2).

Hydraulic, Sedimentation, and Stability Monitoring. Studies of the hydrology of the river and floodplain are fundamental to the restoration project. A hydraulic resistance study is being implemented as specified in U. S. Army Corps of Engineers (1991) to determine the upstream limit of backfilling in Pool B to be reached by the construction project. The flood control needs of surrounding interests will constrain the ultimate size of the restoration project. Hydrologic monitoring and investigations also provide guidance for the operation of the Upper Basin for flood control and water supply. Hydrologic monitoring is needed to assess the status of the five specific hydrologic criteria outlined in U. S. Army Corps of Engineers (1991), which are discussed in detail in Chapter 2. Finally, continuous monitoring of the hydrology of the system is necessary to draw inferences about the relationships between ecological variables and hydrology.

Because the restoration project involves both construction of channels and management of flow through the reconnected river channel and floodplain, the U. S. Army Corps of Engineers (1991) identified the need to monitor sedimentation and channel stability. Of particular concern is the potential for restored flow to erode or deposit sediments in recarved and connector sections of river channel and on graded reaches of backfilled canal. The types of monitoring needed to address these concerns include studies of bank and bed stability and mass transport downstream including suspended and bed loads. Some of these, as well as stability issues, are addressed by the hydrology and geomorphology projects.

The Kissimmee River Restoration Evaluation Program

The Integrated Feasibility Report (U. S. Army Corps of Engineers 1991) broadly identified general monitoring needs. To supplement the IFR's recommendations, the SFWMD sought input on the design of the restoration evaluation program from an advisory panel of external experts on river and wetland ecology and hydraulics (Karr et al. 1992). This guidance helped refine the vision presented in the IFR, yet maintained the spirit of its goals. SFWMD scientists have taken the lead in implementing and expanding these recommendations to create the KRREP. At the core of the KRREP is a group of projects that encompass the four monitoring components outlined by U. S. Army Corps of Engineers (1991). These projects are designed to address these components for the purpose of restoration evaluation. Four important elements of the KRREP are (1) prediction of restoration response, (2) estimation of baseline conditions, (3) sampling designs, and (4) estimation of pre-channelization conditions.

Predictive vs. Monitoring-only Metrics

In addition to collecting data and reporting results, most of the KRREP projects also include restoration expectations, which are formal statements predicting responses of selected metrics to restoration. Expectation development is summarized and archived in an expectation document, which states the expectation and condenses the background, rationale, and data that led to the expectation. The expectation documents are compiled in Anderson et al. (2005). The expectations were developed based on the information presented in this volume's chapters. The baseline studies monitor metrics that fall into one of two main groups, either or both of which may be used in a single study.

Predictive metrics are associated with formal expectations of response to restoration and are used to evaluate the success of the project. Predictive metrics are associated with reference data from the pre-channelized Kissimmee River, from another system judged to be appropriate to represent pre-channelized conditions on the Kissimmee, or that are based on known relationships of the metric to driving variables. Because of their role in restoration evaluation, predictive metrics will be monitored until the necessary drivers have been achieved, and the expected responses have had time to occur. If they have not been achieved by that point, adaptive management actions may be needed, or more detailed study may be necessary to understand why predicted outcomes are not taking place.

Monitoring-only metrics are not associated with expectations but are measured at intervals to evaluate restoration progress. Monitoring-only metrics are also important for evaluation, but lack sufficient information to make the reliable predictions needed for predictive metrics. Some will provide additional information about driver-metric relationships. Both kinds of metrics will be useful in restoration evaluation as the project proceeds, and will likely be used to evaluate achievement of ecological integrity, and to establish any needs for adaptive management.

Estimation of Baseline Conditions

Studies intended to evaluate restoration projects often have failed to include the collection of baseline data prior to the restoration (Anderson and Dugger 1998, Wissmar and Beschta 1998). Without pre-restoration data as a benchmark for comparison with post-restoration data, however, it is not possible to demonstrate that change occurred. Conclusions about achievement of the restoration project's goals, and the success or failure of the project, are enhanced by an ability to prove quantitatively and statistically that the restored system has changed. Change detection relative to a baseline condition is therefore an important aspect of restoration evaluation.

The chapters in this volume describe studies of the channelized river during a baseline period prior to Phase I of the restoration project. The timing of the baseline period varies among studies, but generally ends by June 1, 1999, when construction for the first phase of the project began. For most studies, the baseline period began during or after 1995 and continued for at least one year, although several monitor twice yearly or continuously, and will continue to do so through restoration. A few studies (i.e., hydrology, water quality) were able to use baseline data from existing monitoring programs that were begun in the early 1970s when the channelization project was completed.

Most of the chapters present data for the baseline period of the Phase I project area (most of Pool C and lower Pool B) and make predictions only for Phase I of restoration construction, although some projects extend predictions to other project phases. U. S. Army Corps of Engineers (1991) recognized the primary importance of baseline data from the Phase I area, the need to monitor change until restoration effects stabilize, and the importance of using monitoring results from to Phase I of the project to inform the design and implementation of monitoring of future phases of restoration.

Before-After-Control-Impact Design and Control and Impact Area Sampling

A BACI (before-after-control-impact) approach was used in the design of most of the evaluation studies (Stewart-Oaten et al. 1992). Before-after-control-impact involves measurement of a variable before and after a perturbation both at a location that will be affected by the perturbation (impact) and in an area that will not be affected (control). The approach is analogous to an experimental design in which some subjects receive a treatment and others do not, although true replication in the experimental sense may not be possible. One classic approach to analysis proposed by Stewart-Oaten et al. (1992) is to compare the mean difference between the control and impact area in the before period with the mean control-impact difference in the after period. A significant difference suggests that an effect of the perturbation has been detected.

The KRREP studies used portions of river and floodplain where C-38 would not be backfilled in Phase I (Pools A and/or D) as control areas, and sections that would be restored (most of Pool C, lower Pool B) as impact areas. Pool A is upstream of the restoration project and C-38 will not be backfilled in this pool. Therefore, Pool A should be minimally impacted by the restoration project. In the BACI design, a control site does not have to be identical to the impact area but it should exhibit similar trends over time, as would be expected for locations within the same watershed.

Estimation of Reference Pre-channelization Conditions

Depending on the project and available data, determination of pre-channelization conditions may involve the use of data from the pre-channelization Kissimmee River, areas of the Kissimmee that were judged to be remnant examples of conditions prior to channelization, data available from other systems, or results of experiments. An example of the highest level of reference information, actual pre-channelization data, is the work of Pierce et al. (1982), who used 1952-1954 pre-channelization aerial photography to produce a vegetation map for the entire Kissimmee River and floodplain prior to channelization (this information is used in Chapters 8 and 10 as reference data). Areal data from a digitized version of this map made it possible to estimate the impacts of channelization on floodplain vegetation and to make predictions about the likely results of restoration. The results of Demonstration Project (Toth 1991) studies provided reference data for a number of evaluation studies. Other studies used data collection performed specifically for the study in minimally impacted areas. Estimation of the impact of channelization on study metrics usually involved a comparison of the reference data with baseline data.

Baseline Compendium Volume Overview

The remaining twelve chapters in this volume of baseline studies summarize original data collected on the channelized river and floodplain. Most chapters also present available reference conditions and use the reference information to make inferences about the impacts of channelization and to help develop expectations for the restored ecosystem. The studies were concentrated in Pool C, which contains most of the area of Phase I of the restoration project. This section provides an overview of these chapters.

The first four chapters describe abiotic components of the ecosystem: hydrology, geomorphology, dissolved oxygen, and water quality. Chapter 2 takes advantage of long-term monitoring at permanent stations along the length of the river channel to assess changes in stage and flow characteristics associated with channelization. In Chapter 3, the geomorphology study focuses on the composition of the river channel bed and how it has changed in the absence of flow. In Chapter 4, the concentration of dissolved oxygen is characterized in river channel remnants and compared to values for seven reference streams that also occur in the Lake Okeechobee watershed. Chapter 5 examines a large number of water quality parameters, especially nutrients, and demonstrates the importance of measuring water quality parameters of inflows from upstream and the surrounding watershed. Collectively these four chapters show that abiotic characteristics of the environment were altered by channelization.

Chapters 6 through 10 deal with algal and plant communities. In Chapter 6, algal communities attached to surfaces (periphyton) and suspended in the water column (phytoplankton) are characterized using a combination of community structure and functional groups. Chapter 7 uses permanent transects to evaluate the coverage and structure of aquatic plant communities associated with river channel margins. Chapter 8 examines the structure of floodplain plant communities at permanent plots positioned along elevation gradients across the floodplain to characterize major plant communities. Chapter 9 describes a classification system used in other chapters, which reduces the complexity of plant species data to a smaller, more manageable number of communities. In Chapter 10, the vegetation classification is combined with aerial photography to map the distribution of dominant plant communities across the floodplain. These chapters document changes in algal and plant communities that reflect the impacts of channelization, and illustrate the loss of ecological integrity for a river-floodplain ecosystem. Changes in plant communities have implications for animal communities because plants are an important component of habitat for many animal species.

The final four chapters deal with major groups of animals including aquatic invertebrates, amphibians and reptiles, fish, and birds. In Chapter 11, the community structure, functional feeding and habitat groupings, and secondary production of aquatic invertebrates is quantified for major river and floodplain habitats. Chapter 12 focuses on species richness and other measures of community structure to characterize amphibian and reptile communities associated with degraded floodplain habitats. Chapter 13 summarizes a number of studies of ecological and socioeconomic aspects of fish in both the floodplain and river channel that include assemblage structure, habitat guilds, diet, movement patterns, creel surveys, bioaccumulation of mercury, and physiological responses to hypoxia. Chapter 14 reports studies of wading bird and waterfowl communities and includes the only single species studies in the evaluation program, which examine four federally listed species: wood stork (*Mycteria americana*), bald eagle (*Haliaeetus leucocephalus*), snail kite (*Rostrhamus sociabilis*), and Audubon's crested caracara (*Caracara cheriway*).

CONCLUDING REMARKS

The Kissimmee Basin has undergone numerous changes over the last 150 years. The greatest changes to the Kissimmee River were associated with channelization and flow regulation by the C&SF Project. A major effort is underway to restore ecological integrity to the central portion of the river/floodplain system. The KRREP represents an effort to assess the success of the restoration project and guide future management of the system. The KRREP includes studies of major abiotic components of the ecosystem (hydrology, geomorphology, and water quality) and major biological communities (e.g., plants, invertebrates, fish, and birds). To assess achievement of ecological integrity, the evaluation process will focus on the collection of baseline data as a benchmark against which to evaluate restoration-related change, estimation of the impacts of channelization, development of restoration expectations to predict the effects of restoration, and monitoring of these metrics to assess change as the system responds to restoration.

The 13 chapters that follow perform the initial steps in this evaluation process. Each study reports results of field measurements, usually of multiple metrics, collected during the channelized period. Most were able also to estimate pre-channelization conditions for study metrics and use these data both for estimation of the effects of channelization and prediction of the effects of restoration. Monitoring for the KRREP will proceed through all future project construction phases, and continue for a minimum of five years following project completion.

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CHAPTER 2

IMPACTS OF CHANNELIZATION ON THE HYDROLOGY OF THE KISSIMMEE RIVER, FLORIDA

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ABSTRACT: Long-term records of rainfall, stage, and discharge data for different stations along the length of the Kissimmee River were used to investigate changes in hydrologic characteristics associated with channelization of the river during the 1960s as part of the Central & Southern Flood Control Project. Data were organized by water years (May 1–April 30) for a pre-channelization reference period that ended with Water Year 1962 or earlier to avoid confounding with the channelization project. Data from the reference period were compared to data for a channelized baseline period that could begin as early as 1972 and end as late as 1999 depending on the availability of data for a site. In general, channelization and flow regulation narrowed the range of stage fluctuation, caused more erratic discharge patterns especially increasing the number of days with no flow, shifted the seasonality of flow, and resulted in flow being carried by the C-38 canal and not by the natural river channel. Based on these changes in hydrologic characteristics, five hydrologic expectations were developed for the restored river. These expectations describe the number of days with no flow, the seasonality of flow, stage hydrograph characteristics, stage recession rates, and mean channel velocities.

INTRODUCTION

The ecological integrity of river ecosystems is closely coupled to hydrologic conditions, which can influence the composition of biological communities, the availability and quality of habitat, and connectivity with riparian areas including floodplains (Karr, 1991, Trush et al. 2000). Two of the most common impacts on river hydrology that can degrade ecological integrity involve channelization and flow regulation (e.g., Brookes 1988, Petts 1984), which have become problems of global significance (Benke 1990, Dynesius and Nilsson 1994, Tharme 2003). The importance of recreating natural flow variability for the restoration and management of river ecosystems is captured in the concept of the natural flow regime. This concept characterizes flow in terms of magnitude, frequency, duration, timing and predictability, and rate of change (Poff et al. 1997). While these impacts have been long recognized, they have not always been considered in the cost-benefit analysis of implementing a channelization project (e.g., Arthur D. Little, Inc. 1973). Only recently have there been efforts to restore rivers by managing for the natural flow regime.

One of the most prominent examples of such a restoration project is the Kissimmee River Restoration Project in south-central Florida.

Hydrologic conditions in the Kissimmee River have been altered by more than 100 years of anthropogenic modifications to the channel and flow regime (Appendix 2-1A). By far, the greatest changes are associated with channelization of the river along its entire length from Lake Kissimmee to Lake Okeechobee and with regulation of inflows from Lake Kissimmee during the 1960s as part of the Central and Southern Florida Flood Control Project. Channelization involved excavation of a canal that is much wider and deeper than the natural river channel so that its conveyance capacity (cross-sectional area) was approximately ten-times that of the natural channel. Flow regulation was accomplished by installing gated water control structures at the outflow from Lake Kissimmee and at five downstream locations, which created five terraced pools with nearly level water surfaces. Channelization altered the longitudinal water surface and energy profiles from a continuous, gradually sloping profile, to a discontinuous profile of nearly zero slope with little opportunity for generating flow in remnant river channels. The operation of these structures altered the movement of water through the system by shifting the seasonal distribution of flow volume between wet and dry seasons, increasing the rate of change especially during recession events, and increasing the frequency of periods without flow (Obeysekera and Loftin 1990). Discontinuous flows and the presence of the water control structures have resulted in the flattening of the water surface within a pool so that the upstream portion of each pool remains dry while the downstream portion is inundated permanently. These changes in hydrology have been linked to degradation of river channel habitat and the loss of floodplain wetlands, and changes in biological communities including invertebrates, fish, waterfowl, and wading birds that depend on these habitats (Toth 1990a).

Currently, the South Florida Water Management District and the U. S. Army Corps of Engineers are engaged in the restoration of ecological integrity to the Kissimmee River and its floodplain. The restoration project is guided by hydrologic criteria, which were used to select the most viable among alternative restoration plans (U. S. Army Corps of Engineers 1991). A key feature of this restoration project is an evaluation program, which examines major components of the ecosystem including hydrology. The program will be used to determine if the project is successful and to facilitate adaptive management of the post-project system.

Objectives

The goals of this chapter are to assess changes in hydrologic characteristics of the Kissimmee River that were associated with channelization, and to establish a baseline for evaluating post-restoration hydrologic responses. Hydrology differs from most other baseline studies of the Kissimmee River because long periods of hydrologic data spanning pre- and post-channelization time periods are available. These data allow changes to be measured directly without making comparisons to reference sites. However, it is important to separate changes caused by channelization from those caused by other confounding factors, especially changes in hydrologic drivers (e.g., rainfall). Thus, it is helpful to outline a conceptual model for hydrology to clarify the relationships between hydrologic drivers and responses to the restoration project. Evaluating hydrologic changes also requires a review of the hydrologic criteria that were proposed for the project (Loftin et al. 1990b, U. S. Army Corps of Engineers 1991). Finally, the baseline assessment must characterize baseline conditions for an expanded hydrologic monitoring network established for Phase I of the restoration project. To meet this goal, the chapter has the following objectives (1) present a conceptual model of surface water hydrology for the Kissimmee River, (2) examine changes in the hydrologic drivers, (3) quantify changes in hydrologic characteristics of the river channel and floodplain, (4) reexamine the hydrologic criteria proposed for the Kissimmee River Restoration Project and develop specific hydrologic restoration expectations, and (5) summarize baseline data for the expanded hydrologic monitoring network.

A Hydrologic Conceptual Model

The Hydrologic Conceptual Model

A conceptual model is a formal representation of the components of a system, their interactions, and the factors that influence those interactions. By representing these relationships, a conceptual model is a useful starting point for predicting how a system is likely to respond to management actions such as channelization and flow regulation. Conceptual models were developed for guiding the evaluation program for the Kissimmee River Restoration Project. Existing conceptual models for the river describe changes associated with channelization for major biological communities including vegetation (Toth et al. 1995), invertebrates (Harris et al. 1995), fish (Trexler 1995), and water birds (Weller 1995), as well as an overall ecosystem model (Dahm et al. 1995). Several important abiotic attributes of the system, including dissolved oxygen, nutrients, geomorphology, and hydrology were not captured in detail in those models. Hydrologic changes are a critical driver in the existing models, but hydrology per se was never treated in detail.

This section develops a highly simplified conceptual model of hydrology for the Kissimmee River (Figure 2-1). The model contains a single compartment that represents the quantity of water present as stage or water elevation. For the purposes of this simplified model, the compartment can be thought of as representing any level in a hierarchy of spatial units for a stream ecosystem (e.g., Frissell et al. 1986) — from a channel cross-section, to an alternating point bar unit, to a reach between tributaries, or to the entire drainage network.

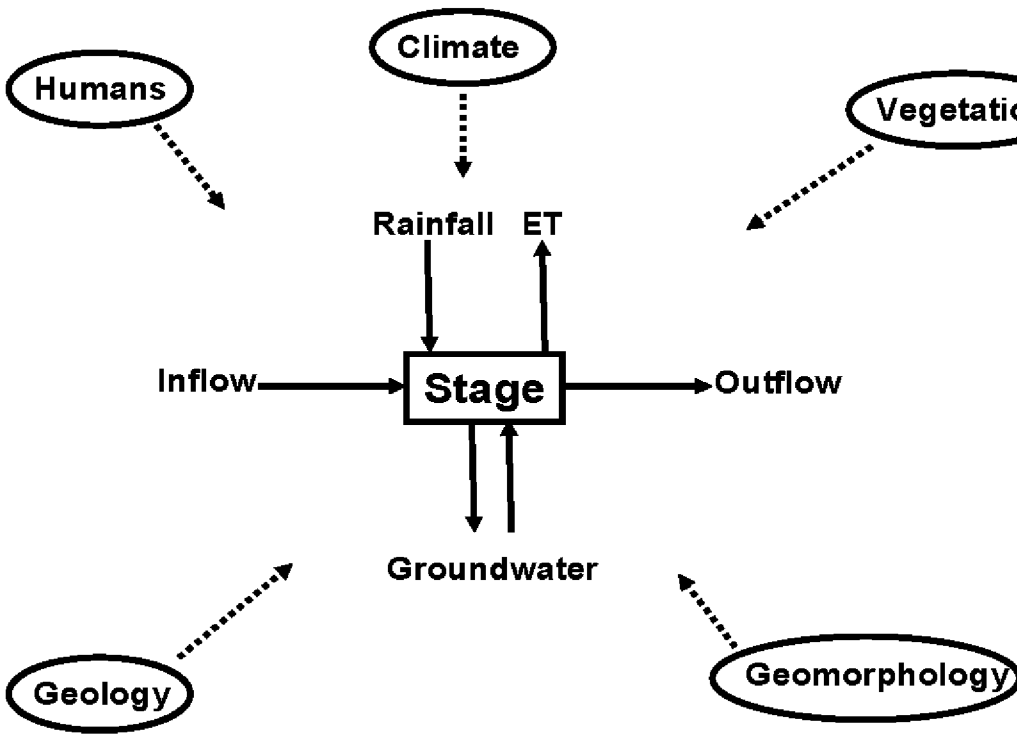


Figure 2-1. Conceptual model for hydrology in the Kissimmee River. The box represents a single compartment for water storage indicated by stage. Solid arrows represent fluxes into and out of the compartment. Ovals and dashed arrows represent system drivers.

The conceptual model can be expanded by adding compartments to capture spatial complexity. Additional compartments could distinguish between the river channel and floodplain in the horizontal dimension, between surface water and groundwater in the vertical dimension, and between upstream and downstream reaches (e.g., pools) in the longitudinal dimension.

Hydrologic Processes

Changes in stage or storage within the compartment depend on differences between fluxes into and out of the compartment (Figure 2-1). These fluxes are determined by a small number of hydrologic processes that are commonly illustrated as steps in the hydrologic cycle in introductory hydrologic texts.

Precipitation. Within the Kissimmee Basin, precipitation takes the form of rainfall and has a highly seasonal distribution.

Inflows and outflows. Inflows are surface water inflows, including those from the upstream reach and tributaries. The outflow is to the downstream reach of the Kissimmee River, which ultimately discharges into Lake Okeechobee. The Kissimmee River lacks distributaries.

Groundwater exchanges. Exchanges between surface water and ground water have not been well-studied within the Kissimmee Basin. In the basin, groundwater can be partitioned into a Surficial Aquifer System (SAS) and a deeper Floridan Aquifer System (FAS), which are separated by an intermediate confining layer of variable thickness (Hawthorne Formation) (Phelps 2002). At certain times of the year, exchanges between the surface water and the SAS may be significant (e.g., as floodplain water levels are receding). Exchanges between the SAS and the FAS occur in the upper basin (Parker et al. 1955), but it is unclear how important connections between the SAS and the FAS are in the lower basin. A recent attempt to use stable isotopes and other chemical parameters to measure the contribution of groundwater from the FAS produced equivocal results (Phelps 2002).

Evapotranspiration. Evapotranspiration includes the return of water vapor to the atmosphere either from soil and water surfaces (evaporation) or from plants (transpiration). Rates of evaporation depend on temperature and relative humidity. The flux also depends on the area of the water surface exposed to the atmosphere. Transpiration strongly depends on the type of vegetation. Evapotranspiration rates (estimated from rainfall-runoff) showed significant decreases between pre-channelization to post-channelization periods for the lower basin, but not the upper basin (Obeysekera and Loftin 1990). Obeysekera and Loftin suggest that more rapid drainage in the channelized system may have reduced the opportunity for evapotranspirative losses.

Drivers

When characterizing differences among stream ecosystems, four factors (relief, lithology, runoff, and vegetation) are generally considered to control stream ecosystem hydrology and geomorphology (Brussock et al. 1985, Montgomery 1999, Winter 2001). These factors can also be used to understand changes over time in the Kissimmee River. A fifth factor, human influences, must also be included.

Climate. Climate affects hydrology in south Florida primarily through the quantity, timing, and distribution of rainfall. The Kissimmee Basin rainfall is highly seasonal with distinct wet (June–November) and dry (December–May) seasons. Climatic shifts associated with El Niño can profoundly influence rainfall patterns over the basin. During El Niño events, the Kissimmee Basin has experienced significantly above normal dry season rainfall (Schmidt et al. 2001). Climatic events can also influence the strength of tropical systems, including tropical storms and hurricanes, which can be significant sources of rainfall during the wet season. Temperature and relative humidity can influence the rates of evapotranspiration.

Vegetation. Vegetation acts in four important ways. First, interception by and stemflow down vegetation can alter the timing and quantity of rainfall reaching the ground. Second, type of vegetation can influence the rate of evapotranspiration. Third, vegetation offers resistance to flow. Resistance varies with the species, growth form, size, and density of the plants. Fourth, vegetation along channel banks can increase bank stability and can affect channel characteristics.

Lithology. Lithology or the composition of the underlying rock has several effects. Porosity influences the amount of storage available for groundwater, the rates of groundwater movement, and the exchange between the SAS and the FAS. The type of underlying material determines to what degree the channel is confined. Chemical composition of the underlying rock can influence the overlying water chemistry and the distribution and composition of vegetation.

Relief. Relief describes the shape of the land surface. It acts primarily by determining the slope of the water surface gradient, which in turn has a major influence on the velocity and discharge of water. Relief was altered by the excavation of the C-38 canal and other drainage projects (agricultural drainage ditches) within the basin, especially on the floodplain.

Humans. Human activity is a super-driver capable of influencing other drivers. In the case of the Kissimmee River, these influences can include very specific impacts associated with the channelization and flow regulation. Human activity can also have indirect effects such as changes in land use with population growth that can alter relationships between rainfall, runoff and inflows into the river. These indirect effects are much more difficult to demonstrate.

Conceptual Model Synthesis

The hydrologic conceptual model raises several issues that are important to consider while examining changes to the hydrology of the river. (1) The Kissimmee River is a spatially complex system with conditions likely to vary with local conditions (e.g., ground elevation) along the length of the channel and across the breadth of the floodplain. Conditions within natural river channels are likely to be quite different from those in the C-38 canal. (2) The channelization project directly and indirectly affected hydrology. Direct changes included altering the relief of the system by excavating the C-38 canal and regulating the flow with water control structures. Indirect effects included changes in rates of evapotranspiration associated with changes in the distribution of water and possibly vegetation. (3) Changes in climatic drivers may be confounded with changes in channelization.

Evolution of the Hydrologic Criteria

Hydrologic needs

A number of scientists, engineers, managers, and concerned stakeholders involved with the Kissimmee River participated in the Kissimmee River Restoration Symposium in 1988. The symposium was a forum to summarize data from studies of the river, especially the Kissimmee River Demonstration Project, and to discuss ecological and engineering concerns related to restoration of the river. The proceedings from this symposium were published by the South Florida Water Management District (Loftin et al. 1990a). One outcome of this symposium was the adoption of ecological integrity as the restoration goal. An ecosystem with ecological integrity is one that is capable of "supporting and maintaining a balanced, integrated, and adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region" (Toth 1990b). The ecological integrity goal emphasized the need to reestablish hydrologic drivers to meet the needs of an ecosystem, rather than focusing on the needs of individual species.

Several presentations at the Symposium referred to specific hydrologic requirements for the Kissimmee River. The following statements paraphrase those requirements:

- To avoid impacting fish communities in river channels, sustained average velocities should not exceed 0.5 m/s (1.5 ft/s) and minimum sustained flows of $\geq$7 m^3/s (247 ft^3/s) are needed to preserve habitat quality (Wullschleger et al. 1990b).
- Florida Game and Fresh Water Fish Commission personnel concluded the following: (1) mean channel velocity up to 2 ft/s would be tolerated by most river species if rest areas (backwaters) were available; (2) if rest areas are not available, velocity of 1.5 ft/s would cause fish migration; (3) if fish migrated from an area, they would not return until velocity was <1 ft/s (Miller 1990).
- Increased bird use of Paradise Run (a portion of the Kissimmee River near Lake Okeechobee), can be attributed to the run's relict pre-channelization river characteristics including some channel flow, periodic water level fluctuation, flooding, plant species, and structural diversity (Toland 1990).
- Stage fluctuations that create flooding during fall and early winter make wet prairies attractive feeding sites especially for dabbling ducks, which feed on seeds. Overwintering ducks leave the area by March. Dewatering wet prairies from spring through early summer allowed annual plants such as wild millet (*Echinochloa walteri*) to germinate and produce seed (John and Turnbull 1990).
- Maintain minimum flows in river channels during summer (June–October) of 250 cfs to maintain dissolved oxygen levels (Wullschleger et al. 1990a).
- Previous studies show that Kissimmee River wetland vegetation communities depend on seasonally fluctuating hydroperiod and that Kissimmee River water levels can fluctuate 3–4 ft annually and flooding can persist for three to five months (Miller et al. 1990).
- Channelization altered flow regimes in the river, which resulted in the loss of (1) inundation of floodplain adjacent to the river channel, which allowed fish movements; (2) fluctuating stages; (3) floodplain recession, which allowed export of animals to the channel to support the river channel food web; and (4) plant growth, sediment deposition, and habitat diversity (Toth 1990a).

These statements indicate that a range of attributes of the river-floodplain ecosystem depend on the hydrologic regime, and that the attributes can depend on different aspects of the flow regime (stage, discharge, velocity). Some statements identify specific thresholds for flows and velocities. Where specific values are given, it was not always clear how they were determined. In some cases, it was not clear if velocity meant mean channel velocity or point measurements.

Papers by Toth (1990b) and Obeysekera and Loftin (1990) discuss changes to hydrologic characteristics of the river associated with channelization. Toth (1990b) in particular, linked the ecological integrity of the river to hydrologic characteristics, especially the flow regime and stage fluctuations. Toth (1990b) summarized the hydrologic determinants of ecological integrity in the following excerpt:

"In summary, pre-channelization hydrologic determinants of ecological integrity of the Kissimmee River ecosystem featured highly variable stage and discharge regimes that included: (1) continuous discharge regimes, with velocities ranging from 0.6 to 1.8 ft/sec when flows were confined within channel banks, (2) a discharge/stage relationship that resulted in frequent overbank flow and long recession intervals, (3) lengthy floodplain hydroperiods with depths typically between 1–2 feet on most of the floodplain, but deeper near the river, and (4) water level fluctuations that led to regular seasonal wet and dry cycles along the periphery of the floodplain, while the remainder of the floodplain was exposed to only intermittent drying periods that varied in timing, duration, and spatial extent."

Hydrologic criteria

The four hydrologic determinants of ecological integrity described above by Toth (1990b) eventually became five hydrologic criteria for the river restoration project. The hydrologic determinants were revisited in a memorandum from L. A. Toth to M. K. Loftin dated 1988. This memorandum describes hydrologic characteristics that were supplied to Dr. H. W. Shen to support hydrologic modeling to evaluate alternative restoration plans for the Kissimmee River. The modeling report appears as an appendix in Loftin et al. (1990b) and includes a copy of this memorandum as an appendix. The four determinants were reworked into five restoration criteria that must all be met simultaneously to achieve the ecological integrity. These criteria were included in Loftin et al. (1990b) for use in evaluating alternative restoration plans, and were also included in the feasibility report for the Kissimmee River Restoration Project (U. S. Army Corps of Engineers 1991). The five criteria are:

(1) Continuous flow with duration and variability characteristics comparable to pre-channelization records. The most important features of this criterion are: (a) reestablishment of continuous flow from July through October, (b) highest annual discharges in September–November and lowest flows in March–May, and (c) a wide range of stochastic discharge variability.
(2) Average flow velocities between 0.8–1.8 feet per second, when flows are contained within channel banks.
(3) A stage-discharge relationship that results in overbank flow along most of the floodplain when discharges exceed 1400 cubic feet per second in the upper reaches of the river and 2000 cubic feet per second in the lower reaches.
(4) Stage recession rates on the floodplain that do not typically exceed 1 foot per month.
(5) Stage hydrographs that result in floodplain inundation frequencies comparable to pre-channelization hydroperiods, including seasonal and long-term variability characteristics.

The feasibility report goes on to describe the relationship of each criterion to ecological integrity. These criteria have also been restated in several publications, including Shen et al. (1994), Toth (1995), and Shen (1996).

The hydrologic criteria describe different aspects of magnitude, frequency, duration, timing, and rates of change of hydrologic events (Table 2-1). These same attributes are captured in the natural flow regime concept defined by Poff et al. (1997). The natural flow regime is based on flow because flow data, unlike stage data, are readily comparable among sites and are available for a network of stream gauging sites established early in the 20th century and maintained by the United States Geological Survey (USGS). Each criterion, except stage-discharge relationships, corresponds to one of the components of the natural flow regime. One challenge for this chapter is to express the hydrologic criteria as quantifiable metrics that can be evaluated as restoration expectations by the evaluation program.

Table 2-1. Hydrologic criteria and the metrics used for evaluation.

Criterion	Notes	Ecological function	Natural Flow characteristic*	Candidate metric
Continuous flow	Continuous flow during June–Oct.	Needed to maintain DO regime	Duration / Magnitude	Days with zero discharge
Seasonality of flow	Peak flow in Sept.–Nov., low flow in March–May		Timing	Mean monthly flow
	Stochastic variability of flow		Timing	
Mean channel velocity	Avg velocity of 0.8–1.8 ft/s when flow is within channel banks	Protect river biota from excessive flows that could disrupt feeding and reproduction, maximizes habitat availability	Magnitude	Mean channel velocity
Stage-discharge relationships	Bankfull Q = 1400–2000 cfs	Defines threshold for floodplain inundation		
Stage recession rates	Rate <1 ft/30 d	Allows prolonged inundation of the floodplain needed for wetland plants	Rate of change	Event recession rate
Stage hydrographs		Requires inundation of the floodplain needed for wetland plants and for the exchange of organisms and materials between the channel and floodplain	Frequency	Maximum stage Minimum stage Mean stage Median stage Change in stage No of days inundation

* Natural flow regime considers five characteristics of flow: magnitude of flow events, frequency, duration, timing, and rate of change (Poff et al. 1997).

METHODS

Study Site

The Kissimmee Basin

The Kissimmee Basin is located in subtropical, south-central Florida (Figure 2-2), where rainfall averages 124 cm/year and falls primarily during a summer-autumn wet season (Warne et al. 2000). The basin is located entirely within the Atlantic Coastal Plain physiographic province in a region of low topographic relief (White 1970). The basin occupies a swale on the Osceola and Okeechobee Plains that is bordered by the Lake Wales Ridge to the west and northwest, and by marine scarps to the east. For purposes of discussion, the basin is frequently divided into an upper basin that extends northward from Lake Kissimmee and a lower basin that contains the Kissimmee River from the outflow of Lake Kissimmee to Lake Okeechobee.

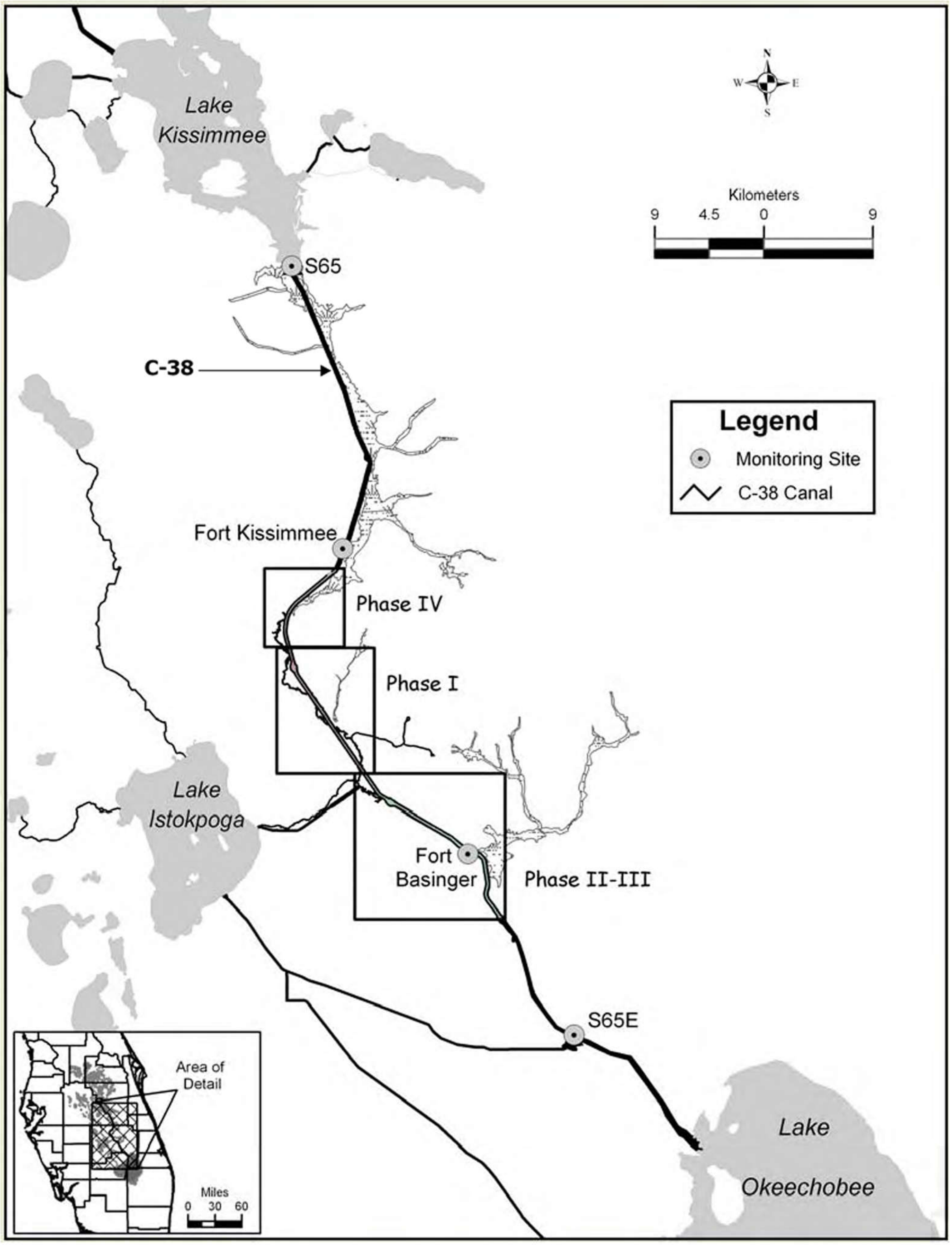


Figure 2-2. Map of the Kissimmee River showing the locations of pre-channelization hydrologic monitoring sites. Rectangles identify the reaches of the river affected by each phase of the restoration project.

The upper basin (4229 km^2) includes several small tributary streams and more than 20 lakes. Collectively, lakes account for 10% of the area of the upper basin. The three major tributary streams are Boggy Creek, which flows into East Lake Tohopekaliga; Shingle Creek, which flows into Lake

Tohopekaliga; and Reedy Creek, which flows into Lake Hatchineha. Excessive rain can raise lake stages causing overflow of the lake margin.

The lower basin (1963 km^2) contains the Kissimmee River and extends from the outlet of Lake Kissimmee to the inflow of Lake Okeechobee. Early studies indicated that mean annual discharge was 1647 cfs in the Kissimmee River near Lake Okeechobee, and that the upper basin contributed approximately 58% of the flow in the river (Parker et al. 1955, p. 307). The remaining 42% came from direct rainfall over the river and from inflow from tributary basins, which range in area from a few square miles, to 143 mi^2 for Chandler Slough (Abtew 1992). The river channel has slopes of only 0.00009 to 0.000057 (Warne et al. 2000). River channels were approximately 115 ft (35 m) wide with a floodplain of 1–3 miles (2–5 km) in width. Early descriptions of the area emphasized that the Kissimmee River flowed through a marshy floodplain with few trees. The floodplains were flanked by dry grassy prairies with some clumps of low shrubs and occasional cabbage palm (*Sabal palmetto*), slash pine (*Pinus caribaea*), and live oak (*Quercus geminata*) savannahs, especially in the Indian Prairie region (Harper 1927).

Hydrologic conditions in the Kissimmee River have been modified by a number of changes to the basin since the 1880s, but the greatest changes were associated with the Central and Southern Florida Flood Control Project (Appendix 2-1A). During the 1960s, the Kissimmee River was channelized along its entire length by excavation of the C-38 canal. This canal is wider, at 90–300 feet (27–91 m), and deeper, at 30 feet (9 m), than the natural river channel. It is 56 mi (90 km) in length, only about half the length of the original meandering channel. The C-38 canal is divided into five pools by water control structures. These structures are operated to maintain stage within a fairly narrow range. The presence of the water control structures and the characteristics of the canal flattened the normal slope of the water surface. These changes permanently inundated the floodplain at the downstream end of the pool and permanently drained the upstream end. Additional information on the basin hydrology can be found in Huber et al. (1976), Loftin et al. (1990a, 1990b), and Warne et al. (2000).

Field Methods

Long Term Changes

Long term changes in hydrologic conditions in the Kissimmee River were examined using stage and flow data from permanent stations that have long periods of record spanning pre- and post-channilization conditions (Table 2-2). These sites were originally established by the USGS as stage and flow monitoring sites. After the Central & Southern Florida (C&SF) flood control project in 1948, the Fort Kissimmee and Fort Basinger stations were deactivated and later reactivated. The S-65 and S-65E stations were originally USGS locations that were replaced by water control structures and taken over by the South Florida Water Management District (SFWMD) (Appendix 2-2A). Stage was monitored with mechanical stage recorders, and converted to discharge using rating curves developed from stage and field measurements of velocity and discharge (Parker et al. 1955). At water control structures, flow was estimated from flow equations using head and tailwater stages, and information about gate openings (Otero 1995, Ansar and Alexis 2003).

Table 2-2. Stations with periods of record before and after channelization.

Station	Data type	Period of record	Min	Max
S-65	Discharge (cfs)	1-Oct-33–30-Apr-99	0	11600
Fort Kissimmee	Stage (ft)	9-Dec-41–30-Apr-99	37.95	50.12
Fort Basinger	Stage (ft)	21-Jun-31–30-Apr-99	23.88	55.84
	Discharge (cfs)	1-Oct-48–30-Sep-64	247	16800
S-65E	Stage (ft)	1-Jan-30–30-Apr-99	13.27	29.31
	Discharge (cfs)	1-Oct-28–30-Apr-99	0	23500

Mean daily stage and flow data were retrieved from the SFWMD hydrologic database DBHYDRO. Stage data are always in reference to the NGVD 1929 datum. In DBHYDRO, each data series is identified with a dbkey, a unique identifier in the database. The dbkeys for the data used in this study and notes on the data series are summarized in Appendix 2-2A. Data were organized by water years lasting from May 1 to April 30, which follows the convention used by the South Florida Environmental Report, so that future analyses can be compared directly with that annual report. Descriptive analyses (time series, frequency distributions, stage duration curves) were used to characterize stage and flow data for a reference period prior to channelization and a baseline period that characterized the channelized system. The length of the reference and baseline periods varied among sites depending upon data availability (Table 2-2). The reference period ended in Water Year (WY) 1962 (April 30, 1962) because this water year preceded completion of any of the water control structures in the upper basin (Guardo 1992). The baseline period began as early as WY 1972 because this period followed the completion of the channelization project. The baseline period ended in WY 1999, just prior to the beginning of Phase I construction in June 1999.

All the pre-channelization stage data were collected at river channel stations. To evaluate floodplain inundation, river channel stage was compared to estimates of the average floodplain elevation given in Obeysekera and Loftin (1990) of 43 feet at Fort Kissimmee, 28.5 feet at Fort Basinger, and 21 feet at S-65E. Similar average floodplain elevations of 43 feet MSL at Fort Kissimmee, 30 feet at Fort Basinger, and 20 feet at S-65E were also given on Plate 5 in U. S. Army Corps of Engineers (1969).

In addition to river data, climatic data were also examined because of the potential for climatic shifts to be confounded with the impacts of channelization. Monthly rainfall for the upper and lower basins was obtained from weighted averages monthly values estimated from Thiessen polygons (Geoff Shaughnessy, SFWMD, unpublished data). Also, information was extracted from Abtew et al. (2004) on drought conditions, as indicated by the Palmer Drought Severity Index, for both the upper and lower Kissimmee Basins. Information on El Niño/La Niña conditions were drawn from Huebner (2000).

Hurricanes and tropical storms passing over the Kissimmee Basin were identified from tracking maps in Williams and Duedall (2002) and Neumann et al. (1999). Hurricane effects, particularly rainfall, occur over a larger area than a line representing the hurricane track. Two hurricanes that made landfall in Florida in September and October 1947 were included even though the hurricane track shows both storms passing to the south of Lake Okeechobee. These storms are commonly considered to have contributed to severe flooding in the Kissimmee Basin (Shen et al. 1994, Koebel 1995).

Evaluation of Hydrologic Criteria

Evaluation of the hydrologic criteria, as stated in the feasibility report (U. S. Army Corps of Engineers 1991), poses two problems. First, all criteria were not expressed as specific metrics. Second, desirable conditions (e.g., stage recessions not to exceed 1 ft /30 d) were identified for only a few of the criteria, and even for these the metrics were not clearly stated. To resolve these issues, one or more candidate metrics were identified for each hydrologic criterion. These metrics were evaluated for pre-channelization and post-channelization periods.

Phase I Baseline Monitoring

Phase I of the restoration project spans most of Pool C and the downstream end of Pool B. The area included in Phase I does not contain any of the long-term hydrologic monitoring stations (Figure 2-2). To evaluate hydrologic changes associated with Phase I, a dense hydrologic monitoring network (Figure 2-3) was established in Pool C between August 22, 1996 and November 24, 1998 to collect hydrologic data prior to the initiation of restoration. Four stations were located in remnant river channels: PC11R, PC33, KRBNS (PC43), and KRDRS (PC54). The remaining 17 stations were located on the floodplain. The 21 stations were arranged as five transects running east to west across the floodplain. This design was adopted so that these monitoring sites could provide information about changes in stage, which could be used to calibrate hydraulic simulation models (U. S. Army Corps of Engineers 1991).

At each station, surface water elevation (stage in feet NGVD 1929) was monitored using a float-encoder. Stage measurements were made to the nearest 0.01 feet and the calibration of the instrument was maintained within 0.02 feet. Surface water wells were positioned so that water levels could be monitored even if water levels dropped below the ground's surface. Additionally, an ultrasonic velocity meter (UVM) was installed at PC33 to allow continuous discharge measurements. Flow measurements are described in detail in Appendix 2-5A.

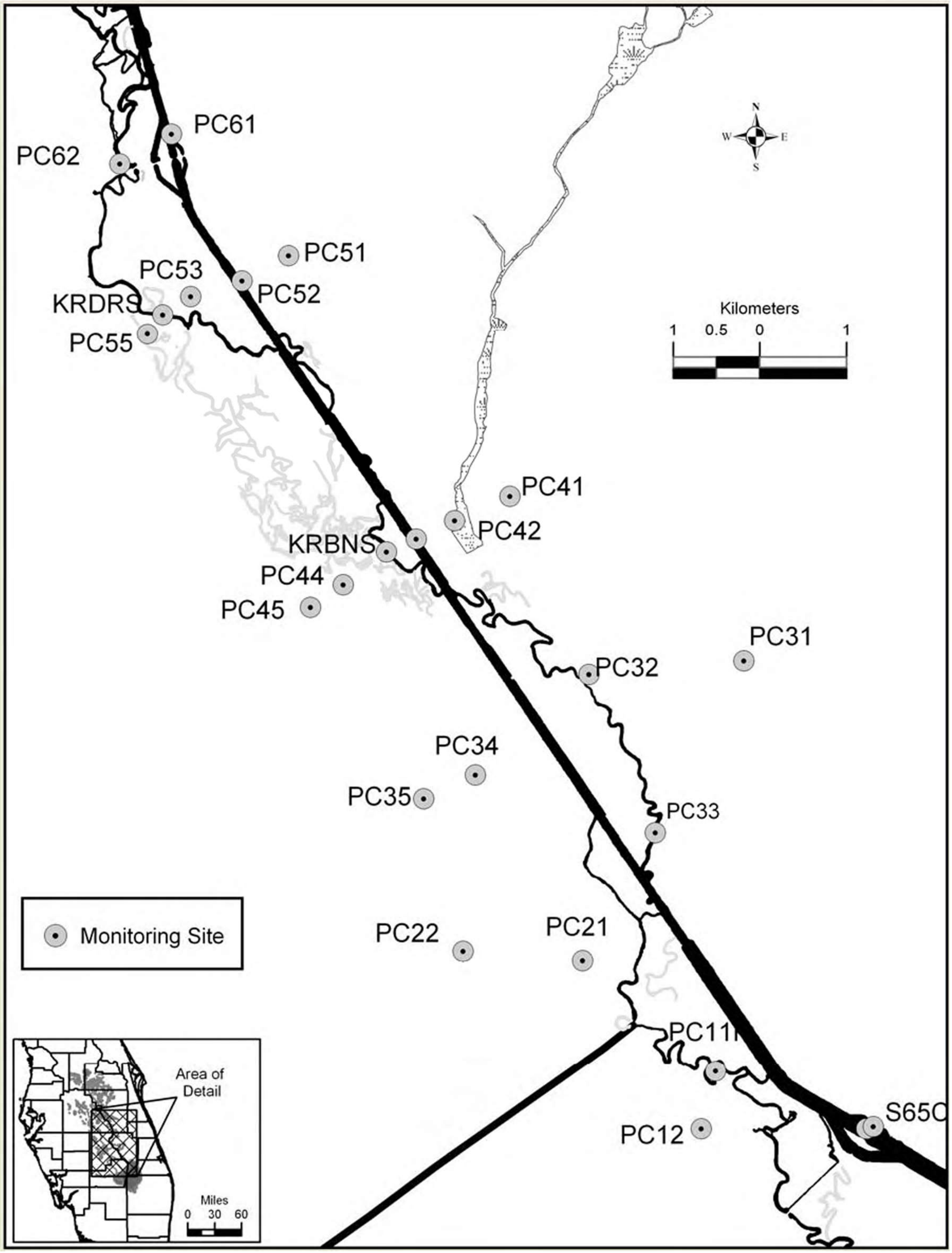


Figure 2-3. Locations of the baseline hydrologic monitoring sites in the area of Phase I of the Kissimmee River Restoration Project.

RESULTS

Long Term Trends

Rainfall

Annual rainfall ranged from 27 to 84 inches in the Kissimmee Basins for 1915–1999 (Figure 2-4A). For most water years, annual rainfall in the upper and lower basins increased or decreased in the same direction but not always by the same amount. For nearly 50 years preceding channelization (WY 1915–1961), annual rainfall in the upper basin ranged from 31.22 inches to 84.05 inches and averaged 51.00 (SE = 1.42) inches. Over the same period, annual rainfall in the lower basin had a slightly narrower range (35.20 inches to 66.48 inches), but a nearly identical average of 51.04 inches (SE 1.24). In the 28 years following channelization but prior to the initiation of Phase I construction (WY 1972–1999), annual rainfall in the upper basin ranged from 33.24 inches to 68.40 inches and averaged 48.08 inches (SE = 1.45). Over the same channelized period, annual rainfall in the lower basin ranged from 27.10 inches to 64.78 inches and averaged 45.27 inches (SE = 1.51). Similar spread of the box plots and their symmetry about the median suggested that variances were homogenous between the upper and lower basins during the reference and baseline periods (Figure 2-4B).

The reference and baseline periods had similar values for mean monthly rainfall, which suggested that only small changes had occurred in the seasonality of rainfall (Figure 2-5). The distribution of mean monthly rainfall suggested distinct dry (November–May) and wet seasons (June–October). For dry season months, mean monthly rainfall for the reference period tended to be within 1 SE of the mean for the baseline period in both basins. For wet season months, mean monthly rainfall for reference period was usually within 1 SE of the baseline values for the upper basin. However, in the lower basin, mean monthly rainfall for the reference period tended to be higher than for the baseline period.

Flood events in the Kissimmee Basin have been linked to hurricane and tropical storm activity. The frequency of tropical storms passing over the Kissimmee Basin was 0.13 storms/year for both the baseline and reference periods (Appendix 2-3A and Appendix 2-4A). The frequency of hurricanes passing over the basin after channelization was 0.08 hurricanes per year, which was approximately half the frequency of 0.17 for the reference period. Not all hurricanes maintained hurricane strength as they passed over the basin.

River Channel Stage

Mean daily stage was available at four stations along the river and was plotted through May 31, 1999, which marked the beginning of Phase I construction (Figure 2-6). From upstream to downstream, these stations were S-65 at the outflow from Lake Kissimmee, Fort Kissimmee in upper Pool B, Fort Basinger in upper Pool D, and S-65E near Lake Okeechobee. Stages decreased with location along the river channel in relation to the sloping ground elevation. All four stations exhibited a narrowing of the range of stage fluctuation after stage regulation began in the 1960s.

Mean daily stage data for Fort Kissimmee was available for a 20 year reference period (WY 1943–1962) representing pre-channelization, and a 14 year baseline period (WY 1985–1999) representing the channelized river (Appendix 2-2A). During the reference period, data were missing for 579 d, which included one year of missing values (October 1, 1953–September 30, 1954). During the baseline period, data were missing for 77 d. During the reference period, mean daily stage ranged from 38.47 feet to 50.12 feet and averaged 44.20 feet (Figure 2-7A). A frequency distribution of mean daily stages approximated a normal distribution. During the baseline period, mean daily stage had a narrower range of only 38.02 feet to 46.76 feet and averaged 41.61 feet (Figure 2-7B), which was 2.59 feet less than the average stage for the reference period. Stage conditions at this site are not representative of the rest of the river during the baseline period because a fluctuating pool stage regulation schedule was implemented on October 28, 1985 for Pool B (Toth 1991). When the S-65B structure was installed, it was operated to maintain a constant stage of 40 feet in Pool B. The fluctuating stage schedule was implemented as part of the Kissimmee River Restoration Demonstration Project. The schedule allowed stage to vary from 39 feet to 42 feet with a drawdown to 38 feet every three–five years. Stage fluctuation within a water year at Fort Kissimmee ranged from 0.72 feet to 9.02 feet during the reference period and 3.0 feet to 8.3 feet during the baseline period. Stage duration curves for Fort Kissimmee showed that a greater percentage of the reference period exhibited higher stages than during the baseline period (Figure 2-7C). The stage duration curve for the reference period shows that 76% of the measurements exceeded 43 feet, which is the average floodplain elevation (Obeysekera and Loftin 1990).

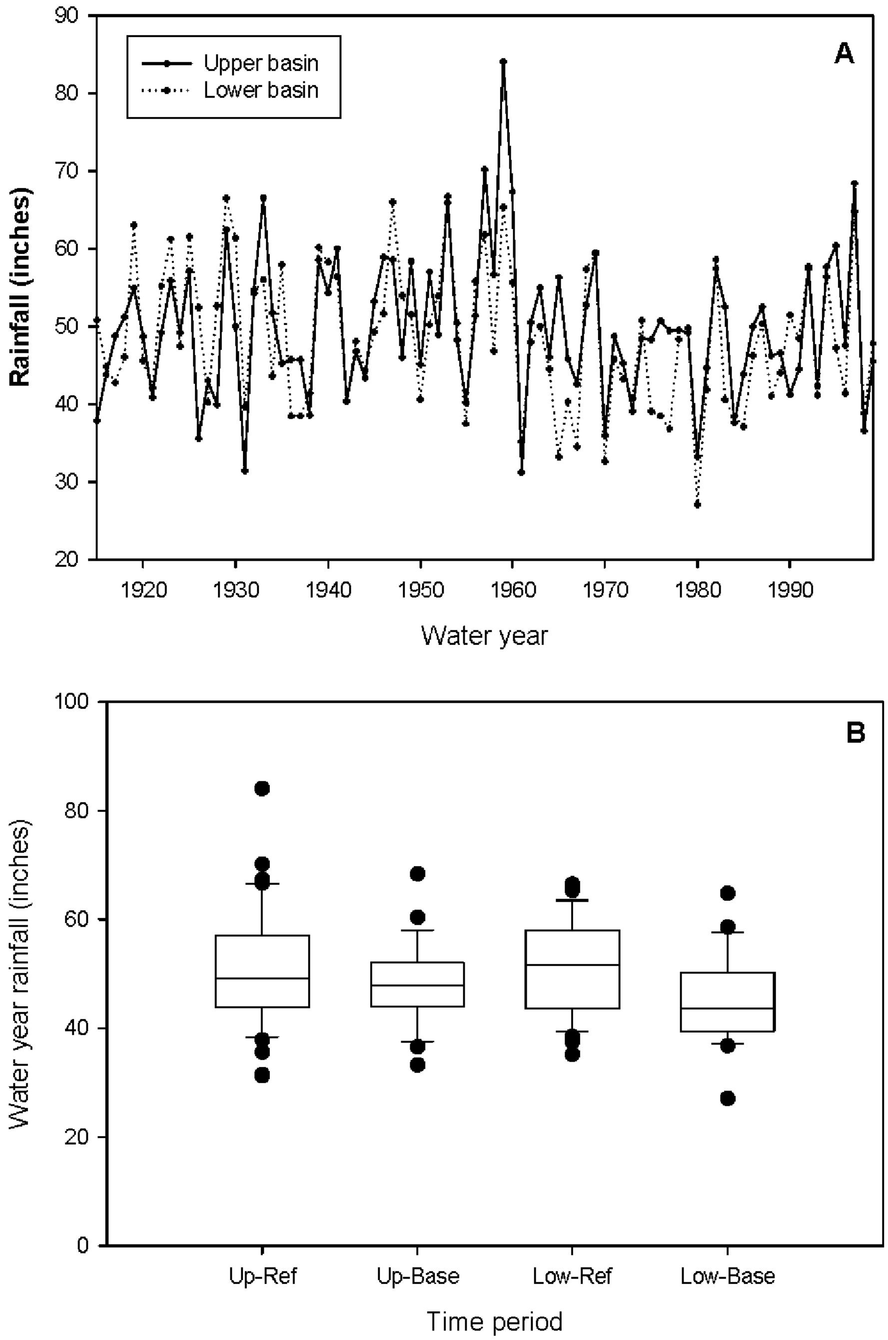


Figure 2-4. A. Annual rainfall by water year (May 1–April 30) for the upper and lower Kissimmee Basins. B. Box plots of annual rainfall for the upper basin reference and baseline periods, and for the lower baseline reference and baseline periods. The reference period was WY 1915–1961, and the baseline period was WY 1972–1999. In the box plots, the horizontal line is the median, the ends of the box represent the 25th and 75th percentiles and the error bars represent the 10th and 90th percentiles. Filled circles are values <10th or >90th percentile.

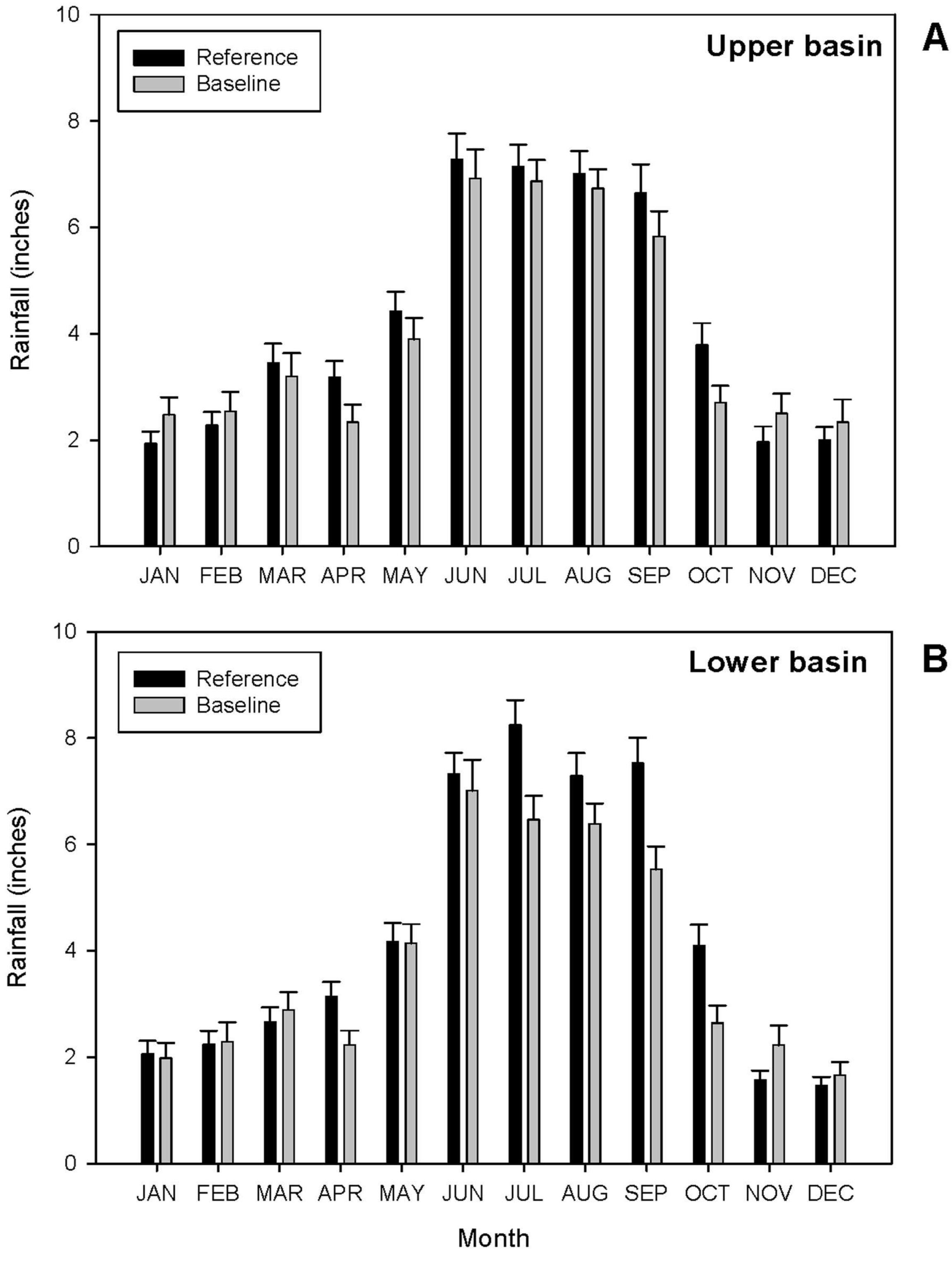


Figure 2-5. Mean monthly rainfall for the reference (1915–1961) and baseline (1972–1999) periods in the upper basin (A) and lower basin (B). Error bars are 1 SE.

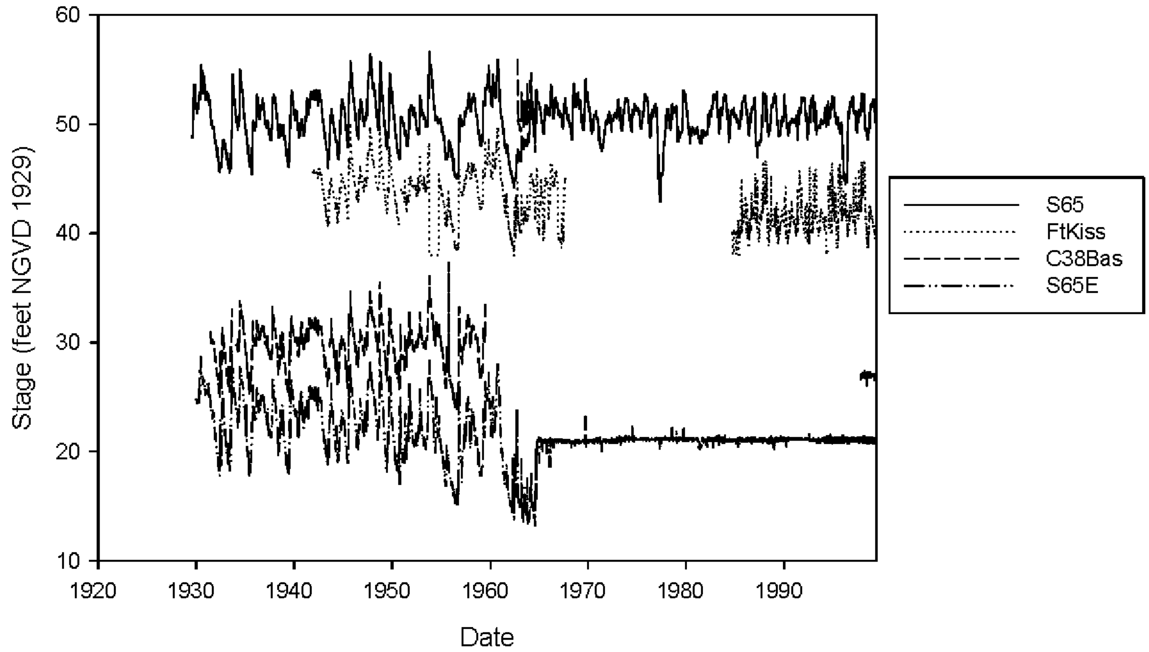


Figure 2-6. Mean daily stage at S-65, Fort Kissimmee, C38Bas, and S-65E. Period of record shown was truncated on June 1, 1999, for the end of the baseline period. Note that the Fort Kissimmee stage period of record resumes after a fluctuating regulation schedule was implemented for Pool B as part of the Kissimmee River Demonstration Project.

At Fort Basinger, stage data were available for 20 years (WY 1933–1962) during the reference period, but only one year (WY 1999) was available during the baseline period. Data were missing for 247 d during the reference period, but none was missing for the baseline period. During the reference period, mean daily stage ranged from 23.88 feet to 37.34 feet and averaged 29.69 feet (Figure 2-8A). For the single year of baseline data, stages varied over a much narrower range of 26.03 feet and 27.82 feet, and the average stage of 26.94 feet for the baseline period was 2.75 feet less than the average for the reference period (Figure 2-8B). Stage duration curves indicate that the mean floodplain elevation of 28.5 ft at this location was exceeded on approximately 75% of days during the reference period (Obeysekera and Loftin 1990), while floodplain elevation never exceeded the mean during the baseline period (Figure 2-8C).

At S-65E, mean daily stage data were available for 32 years (WY 1931–1962) during the reference period, and data were missing for only three days. Mean daily stage ranged from 14.5 feet to 29.31 feet and averaged 22.30 feet (Figure 2-9A). During the baseline period, mean daily stage data were available for 29 years (WY 1972–1999), and data were missing for 33 days. During the baseline period, mean daily stage varied over a much smaller range of 20.00 feet to 22.23 feet and averaged only 21.07 feet (Figure 2-9B). Stage duration curves showed that mean daily stage exceeded the mean floodplain elevation of 21 feet at this location (Obeysekera and Loftin 1990) for 69% of the reference period and for 97% of the baseline period (Figure 2-9C). The nearly continuous flooding at this location during the baseline period reflected ponding at the lower end of the pools that resulted from the flat water surface within pools.

Discharge

Changes in discharge associated with channelization were examined using a continuous record (i.e., no missing values except during construction) of mean daily discharge for the outflow from Lake Kissimmee, which is the location of S-65, and in the lower Kissimmee River at the present location of S-65E. Discharge data were not included for Fort Basinger or Fort Kissimmee because data at Fort Basinger were available for only three years (October 1, 1948–September 30, 1951) before channelization, and none were available for Fort Kissimmee.

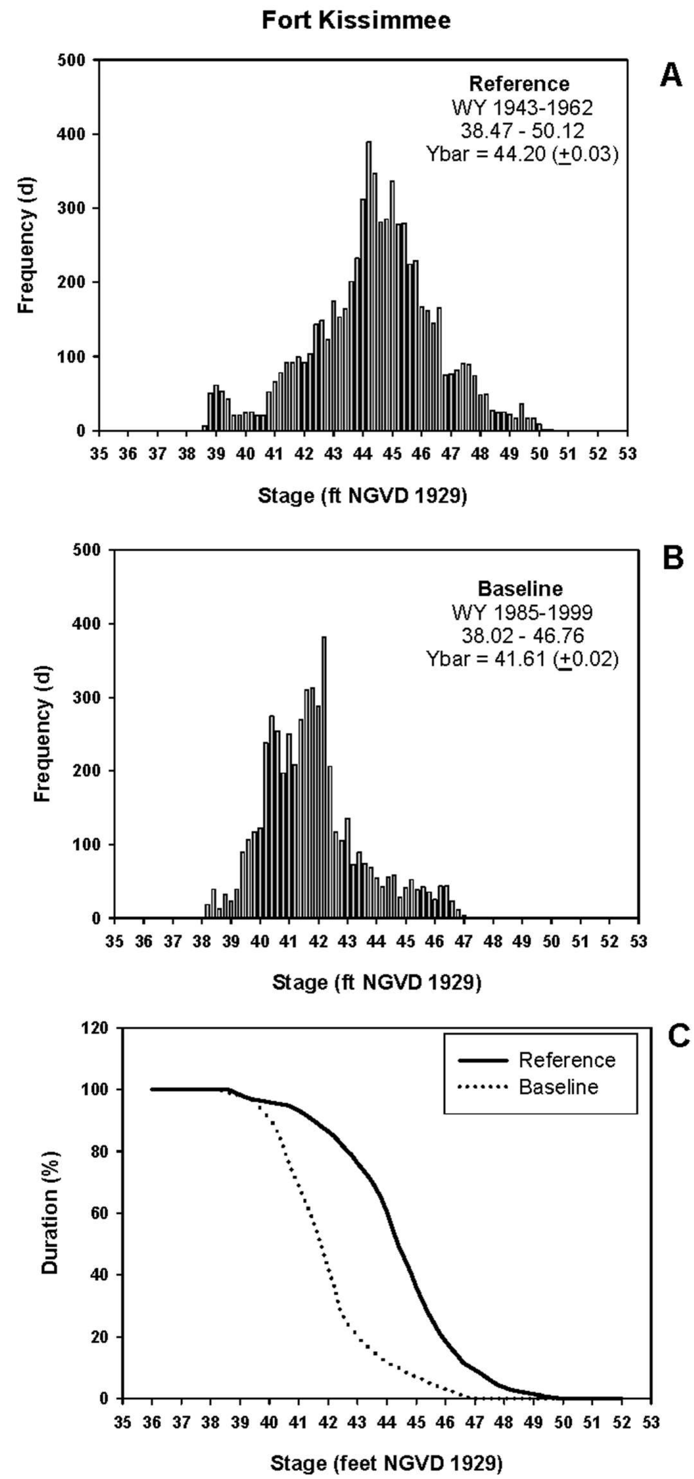


Figure 2-7. A. Frequency of mean daily stage at Fort Kissimmee for the reference period. B. Frequency of mean daily stage during the baseline period. C. Stage duration for the reference and baseline periods. Mean floodplain elevation at this station is 43 feet (Obeysekera and Loftin 1990).

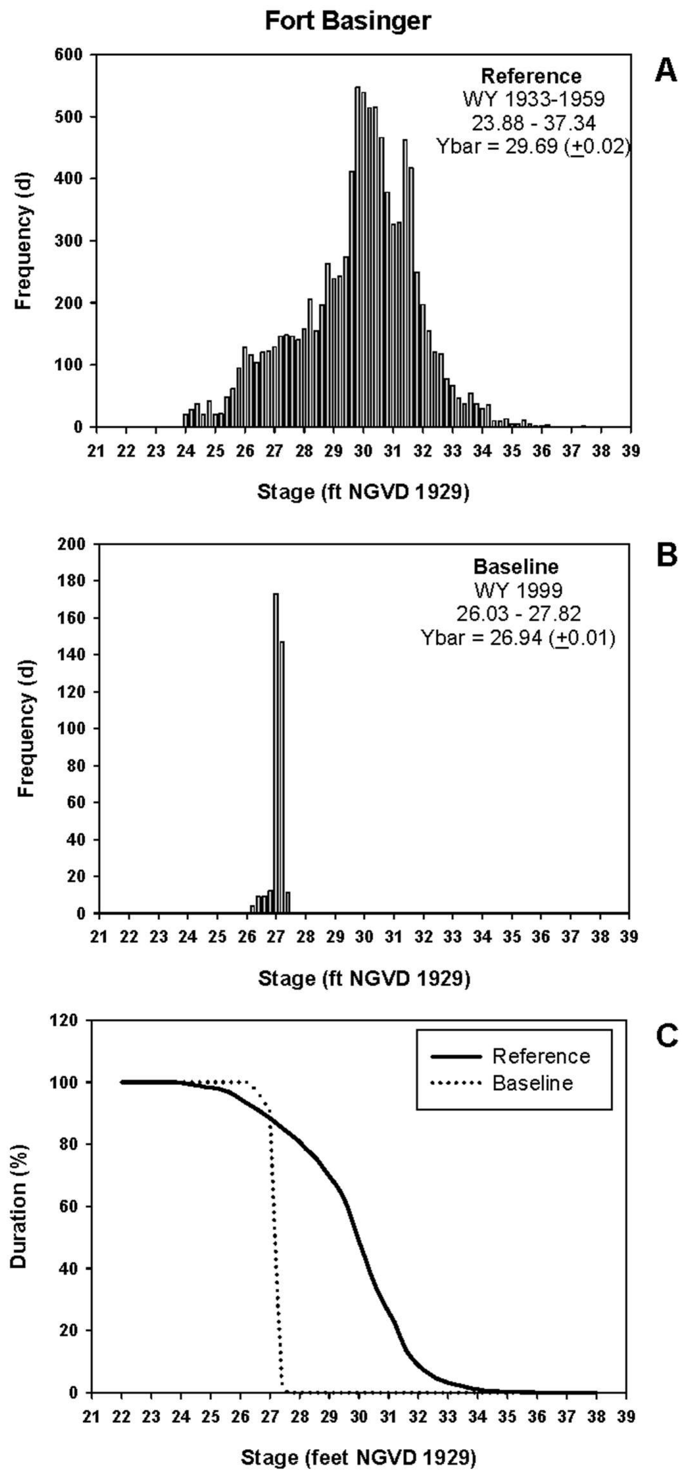


Figure 2-8. A. Frequency of mean daily stage at Fort Basinger for the reference period. B. Frequency of mean daily stage during the baseline period. C. Stage duration for the reference and baseline periods. Mean floodplain elevation at this station is 28.5 feet (Obeysekera and Loftin 1990).

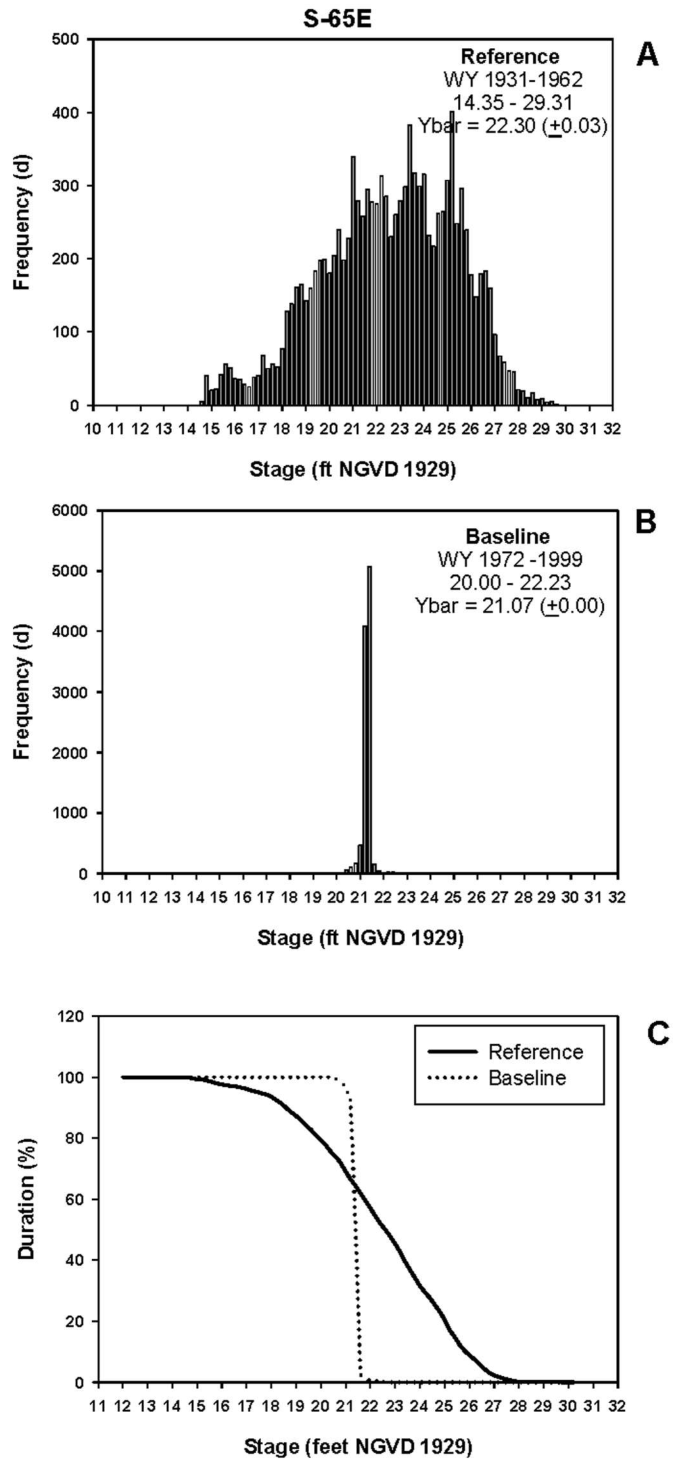


Figure 2-9. A. Frequency of mean daily stage at S-65E for the reference period. B. Frequency of mean daily stage during the baseline period. C. Stage duration for the reference and baseline periods. Mean floodplain elevation at this station is 21 feet (Obeysekera and Loftin 1990).

At both S-65 and S-65E, mean daily discharge tended to change more gradually during the pre-channelization period than during the post-channelization period (Figure 2-10). Extreme highs and lows and rapid changes were much more frequent in the post-channelization period. For example, prior to channelization, discharge was 0 cfs for only six days at S-65 and always >0 cfs at S-65E, but during the baseline period, the number of days with discharge of 0 cfs increased to 3108 d at S-65 and 787 d at S-65E. The smaller number of days without discharge at S-65E than at S-65 probably reflected its downstream position in the drainage network and the contribution of watershed inflows from tributaries and runoff.

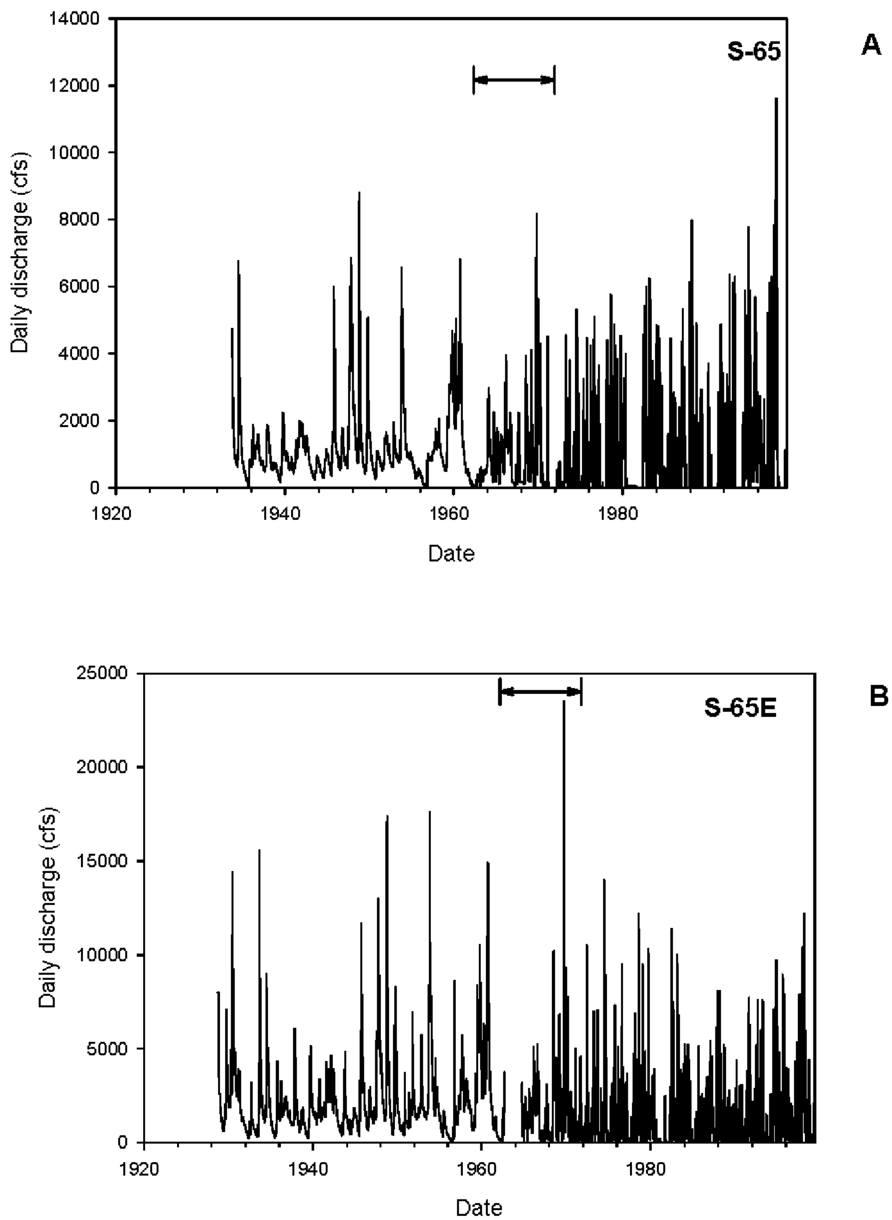


Figure 2-10. Mean daily discharge at S-65 and S-65E. Double-headed arrows indicate the time interval when channelization occurred.

A frequency distribution of mean daily discharge at S-65 approximated the shape of a log normal distribution during the reference period (WY 1935–1962) (Figure 2-11A). For this period, mean daily discharge varied between 0 and 8800 cfs and averaged 1233 cfs. After channelization, mean daily discharge varied over a larger range of between 0 and 11600 cfs than during the reference period (Figure 2-11B), but the average of 909 cfs for the baseline period (WY 1972–1999) was lower than for the reference period. At S-65, the reference period contained a larger percentage of mean daily discharges between 0 cfs and 1400 cfs, but the baseline period contained a larger percentage of discharges of 0 cfs or >1400 cfs (Figure 2-11C).

At S-65E, mean daily discharge during the reference period (WY 1930–1962) ranged from 68 cfs to 17600 cfs and averaged 2191 cfs (Figure 2-12A). During the baseline period (WY 1972–1999), mean daily discharge ranged from 0 to 14,000 cfs (Figure 2-12B). A larger percentage of time during the reference period than the baseline period exhibited mean daily discharges >0 cfs and $\leq$3000 cfs when the stage duration curves converge (Figure 2-12C).

Mean annual discharge at S-65 (mean daily discharge averaged for a water year) did not show clear trends between the reference and baseline period (Figure 2-13A). During the reference period (WY 1935–1962), mean annual discharge ranged from 331 cfs to 3042 cfs and averaged 1233 (SE = 145) cfs. During the baseline period (WY 1972–1999), mean annual discharge had a slightly lower range of 24–3005 cfs and a slightly lower average of 909 (SE = 139) cfs. At S-65E, mean annual discharge exhibited a similar small decrease from the reference (WY 1929–1962) to the baseline (WY 1972–1999) period (Figure 2-13B). During the reference period, mean annual discharge ranged from 568 cfs to 5287 cfs and averaged 2191 (SE = 226) cfs, and during the baseline period, it ranged from 158 cfs to 3802 cfs and averaged 1327 (SE = 177) cfs.

Hydrologic Criteria

Continuous Flow

The continuous flow criterion was quantified with the metric - number of days in a water year when the mean daily discharge was 0 cfs. Days of zero discharge were rare during the reference period at S-65 (Figure 2-14A) and S-65E (Figure 2-14B), but were much more common after channelization and flow regulation in the 1960s. During the reference period at S-65, only six days of zero discharge were recorded and these occurred in October 1956 (WY 1957). These days, which were recorded as zero discharge in DBHYDRO, were actually a period of reverse flow into Lake Kissimmee (J. Chamberlain, unpublished data). Reverse flow occurred because heavy rainfall (16 in. in two days) followed severe drought conditions, and because constructed levees along the river reduced the floodplain width to 400 ft in some downstream areas. During the baseline period, days of zero discharge were much more common at S-65 especially after 1980 (Figure 2-14A), which may have resulted from changes in the regulation schedule. Days of zero discharge ranged from 0 to 312 d per water year and averaged 111 d (Table 2-3). At S-65E, zero discharge did not occur during the reference period. During the baseline period, it ranged from 1 d to 125 d per water year and averaged 28 d per water year (Table 2-3). Data for PC33 were available only for one year (WY 1999) in the baseline period, during which PC33 had 346 d of zero discharge.

Flow Variability

The second part of the continuous flow criterion emphasizes the seasonality and natural variability of discharge. Flow variability was assessed using the distribution of mean monthly discharge (average of mean daily discharge for a given month) and the coefficient of interannual mean monthly discharge

<u>Reference Conditions.</u> Reference conditions were derived from daily discharge data at historic river channel gages at the outlet of Lake Kissimmee (near existing location of S-65) and near Lake Okeechobee (near existing location of S-65E) from 1933 to 1960. Pre-channelization discharge data were estimated for S-65C to provide reference conditions for Phase I of the restoration project. These data were estimated using pre-channelization daily discharge at the outlet of the Kissimmee River basin (S-65E) and the ratios of drainage basin areas associated with these locations.

Pre-channelization mean monthly flows were higher during September through November and lower from January through June (Figure 2-15A). Interannual variation of historic monthly flows (Figure 2-15B) indicates minimal differences between months, with the largest variation occurring in June at the downstream gauge near Lake Okeechobee.

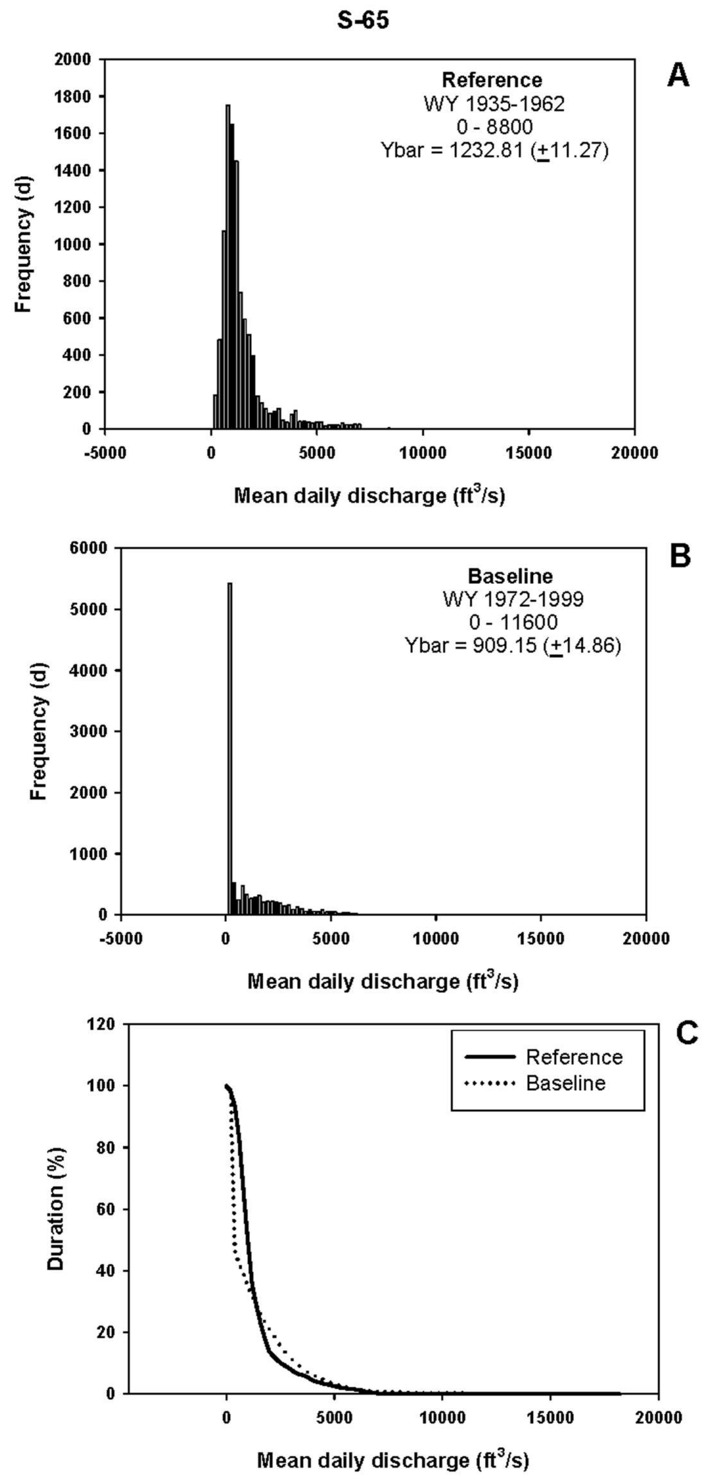


Figure 2-11. A. Frequency distribution of mean daily discharge for the reference period at S-65. B. Frequency distribution of mean daily discharge for the baseline period. C. Duration of discharges during the time interval for the reference period and the baseline period.

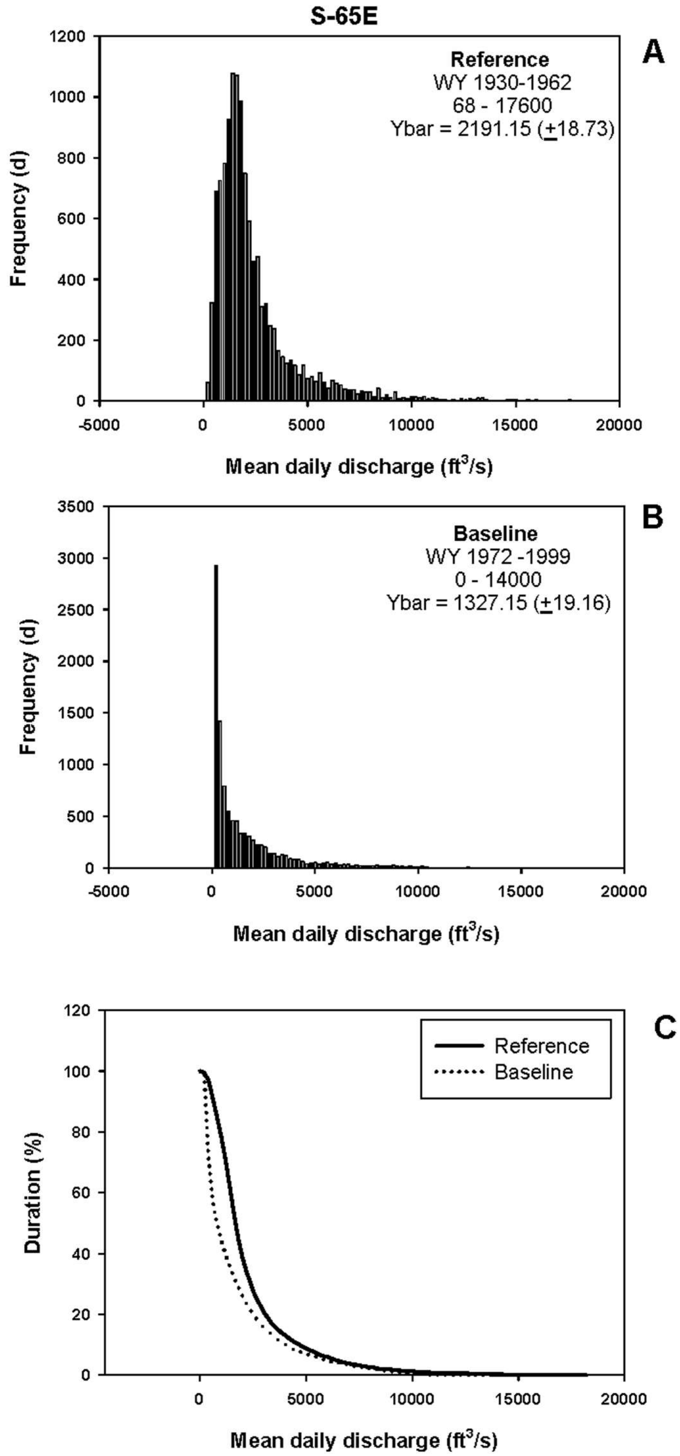
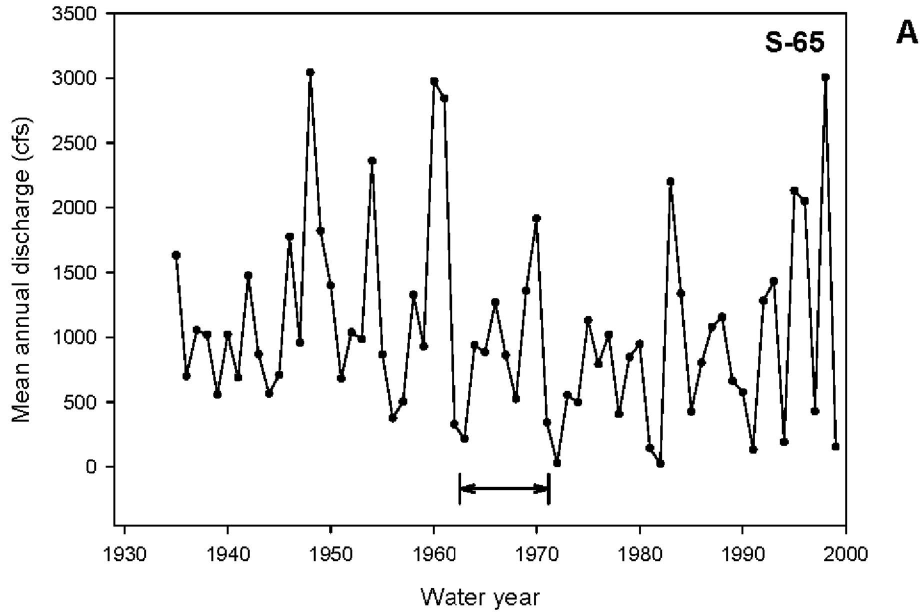


Figure 2-12. A. Frequency distribution of mean daily discharge for the reference period at S-65E. B. Frequency distribution of mean daily discharge for the baseline period. C. Duration of discharges during the time interval for the reference period and the baseline period.



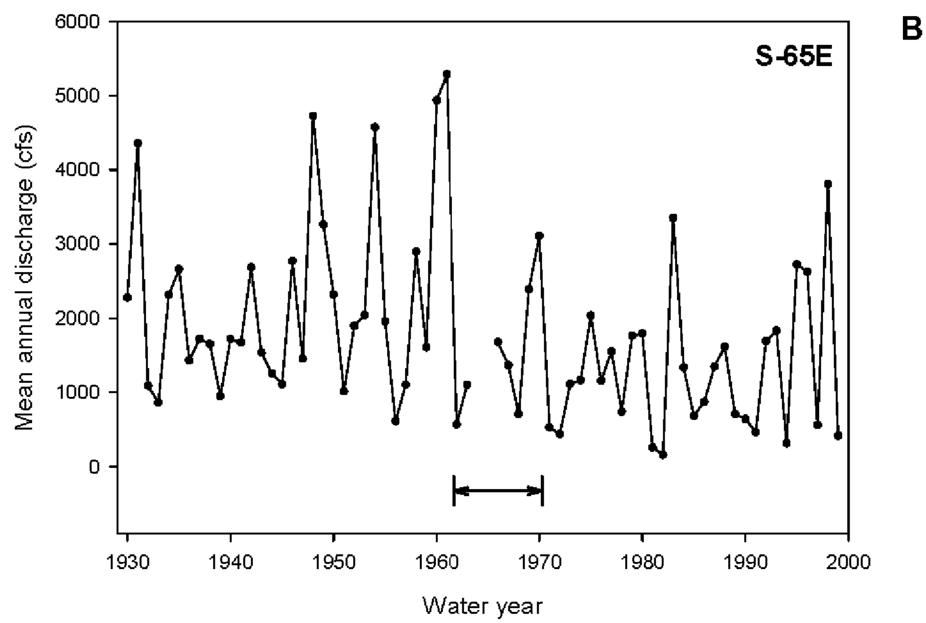


Figure 2-13. Mean annual daily discharge at S-65 (A) and S-65E (B). Double-headed arrows indicate the time interval when channelization occurred.

Baseline Conditions. Baseline conditions were derived from daily discharge at S-65, S-65C, and S-65E from 1971 to 1998 and daily discharge at PC33 on Micco Bluff Run, a remnant river channel in Pool C. S-65 is located at the outlet of the Upper Kissimmee Basin and contributes approximately 60% of the flows through the channelized Kissimmee River. S-65C is located near the middle of the area to be restored. The S-65E structure is located at the outlet of the Kissimmee River basin, approximately seven miles downstream of the restoration project limits. Data collected from November 1997 to May 1999 at PC33 are representative of baseline conditions in sections of river channel that will be affected by the first phase of restoration.

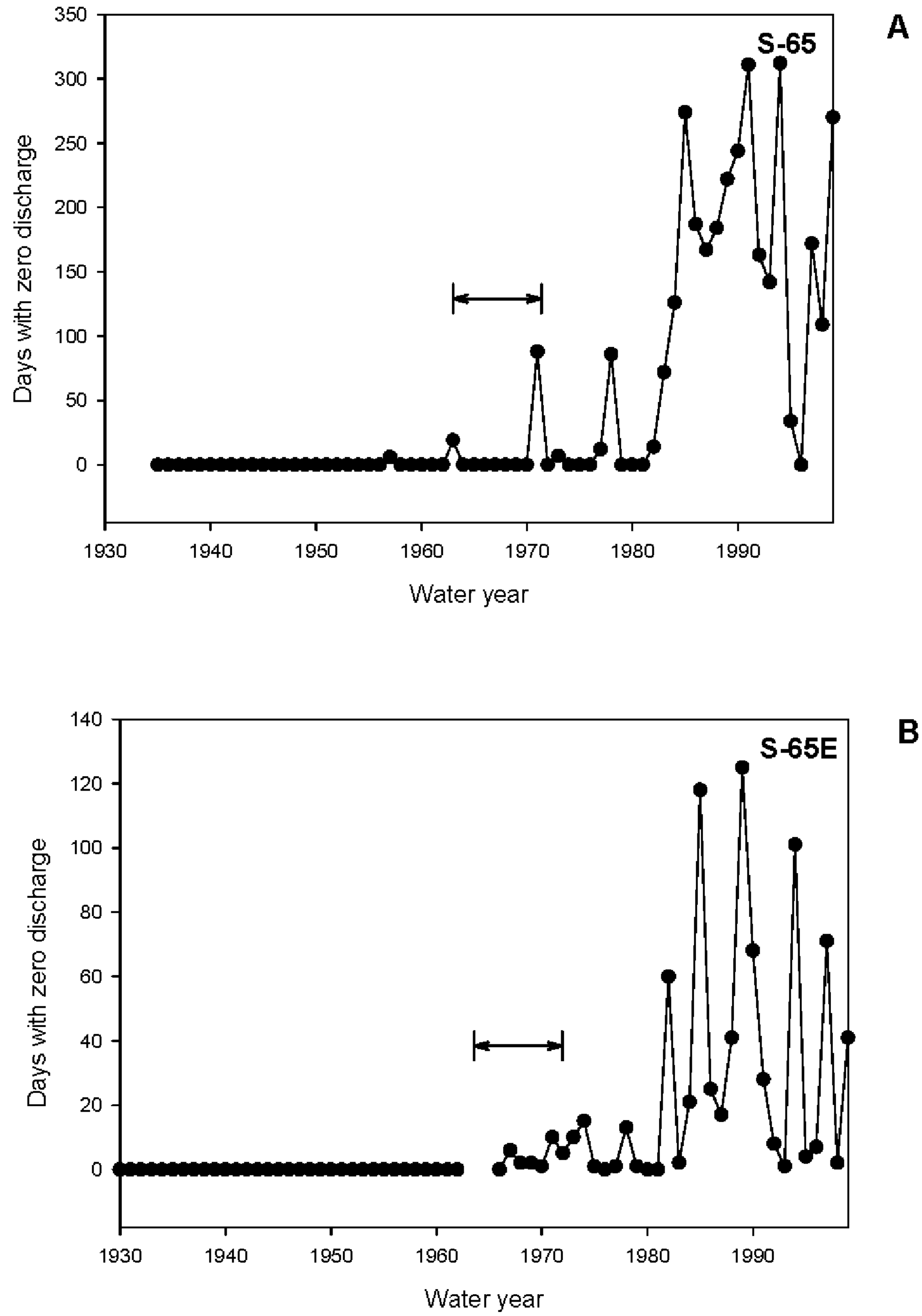


Figure 2-14. Number of days each water year that mean daily discharge was 0 cfs at S-65 (A) and S-65E (B). Double-headed arrows indicate the time interval when channelization occurred.

At S-65, S-65C, and S-65E, the highest flows occurred from January through April and in August and September, while low flows occurred in June, November, and December (Figure 2-16A). During wet season months from June through October, flows increased along the channelized river due to lower basin tributary inflows. During the dry season, flows were primarily a function of headwater discharges with little difference between upstream and downstream locations.

Discharges at the S-65 structures represent flows in the C-38 canal and are different from flow conditions in remnant river channels. Monthly mean discharges at PC33 lacked a seasonal pattern. Discharges were zero

75% of the time from November 1997 to May 1999. Daily river flows (PC33) were less than 5% of C-38 discharge 83% of the period when PC33 flows were >0 ft^3/s.

Table 2-3. For the metric number of days per water year with discharge = 0, the number of water years of observations (N), the range, mean (Ybar) and median values for reference and baseline periods at three sites.

	Reference Period				Baseline Period			
Site	N	Range	Ybar(SE)	Median	N	Range	Ybar(SE)	Median
S-65	28	0–6	0.21(0.21)	0	28	0–312	111(20.60)	97.5
PC33*					1	346		
S-65E	33	0	0 (0.01)	0	28	1–125	28.07(7.01)	11.5

* PC33 only had data for WY 1999.

Interannual variation of monthly mean flows (Figure 2-16B), as described by the coefficient of variation (standard deviation/mean), was high (relative to the historic system) during most months. S-65 had the highest variability, which occurred during months with high frequencies of zero flow (June, July, October, November, and December). Baseline intraannual and interannual distributions of monthly mean flows resulted from the current operation schedule at S-65, which is designed to lower stages in the headwater lakes between February and June in preparation for wet season rainfall. Lakes are allowed to fill to their maximum flood control elevation from June to November through February. Flood control operations have produced a seasonal shift of high and low flows and extended periods of no flow.

Mean Channel Velocity

Reference Conditions. Reference conditions were derived from the USGS historic stream gauging data at Kissimmee River below Lake Kissimmee (USGS site 2269000) and Kissimmee River near Cornwell/Bassinger (USGS site 2272500). A total of 342 measurements were collected between 1931 and 1959 (309 below Lake Kissimmee and 33 near Cornwell/Bassinger). Of these measurements, 179 were rated fair to excellent by the USGS and were used to derive mean velocities in the main river channel, which ranged between 0.8 to 1.8 ft/s (0.2 to 0.6 m/s) during 93% of these sampling events (Figure 2-17). Main channel discharges associated with velocities between 0.8 to 1.8 ft/s (0.2 to 0.6 m/s) ranged from approximately 100 to 2100 ft^3/s (3 to 59 m^3/s), with flows exceeding 500 ft^3/s (15 m^3/s) during 88% of the sampling events.

Baseline Conditions. Baseline conditions were derived from daily discharge at site PC33 on Micco Bluff Run, a remnant river channel in Pool C. Data from this site are representative of baseline conditions (November 1997–May 1999) within remnant river channels that will be affected by the first phase of restoration. Daily discharge at PC33 ranged from 0 to 1170 ft^3/s (33 m^3/s) but flows greater than 100 ft^3/s (2.8 m^3/s) occurred only 5% of the time. Mean channel velocities were calculated by dividing discharge by the cross sectional area of the river channel and ranged from 0.0 to 1.61 ft/s (0.49 m/s). However, because remnant river channels rarely conveyed discharge, mean channel velocities were less than 0.8 ft/s (0.2 m/s) 99% of the baseline period (Figure 2-17).

Stage Recession Rates

Reference Conditions. Reference conditions were derived from daily stage data at Fort Kissimmee (Figure 4) and Fort Basinger (Figure 5) from 1942 to 1959. Based on these data, peak stages typically occurred in September or October and slowly receded until May or June. Slow stage recession rates provided connectivity between the river and floodplain that contributed to habitat diversity and functionality, and allowed for transfer of available food resources between the river and floodplain.

Thirty-day recession rates were calculated by the difference between maximum and minimum stages for each recession event divided by the total number of days water levels receded, and multiplied by 30 days (Table 2-4, Table 2-5). Small increases in stage were ignored during prolonged recession events. However, a stage increase >1.5 ft (45 cm) was considered a new recession event.

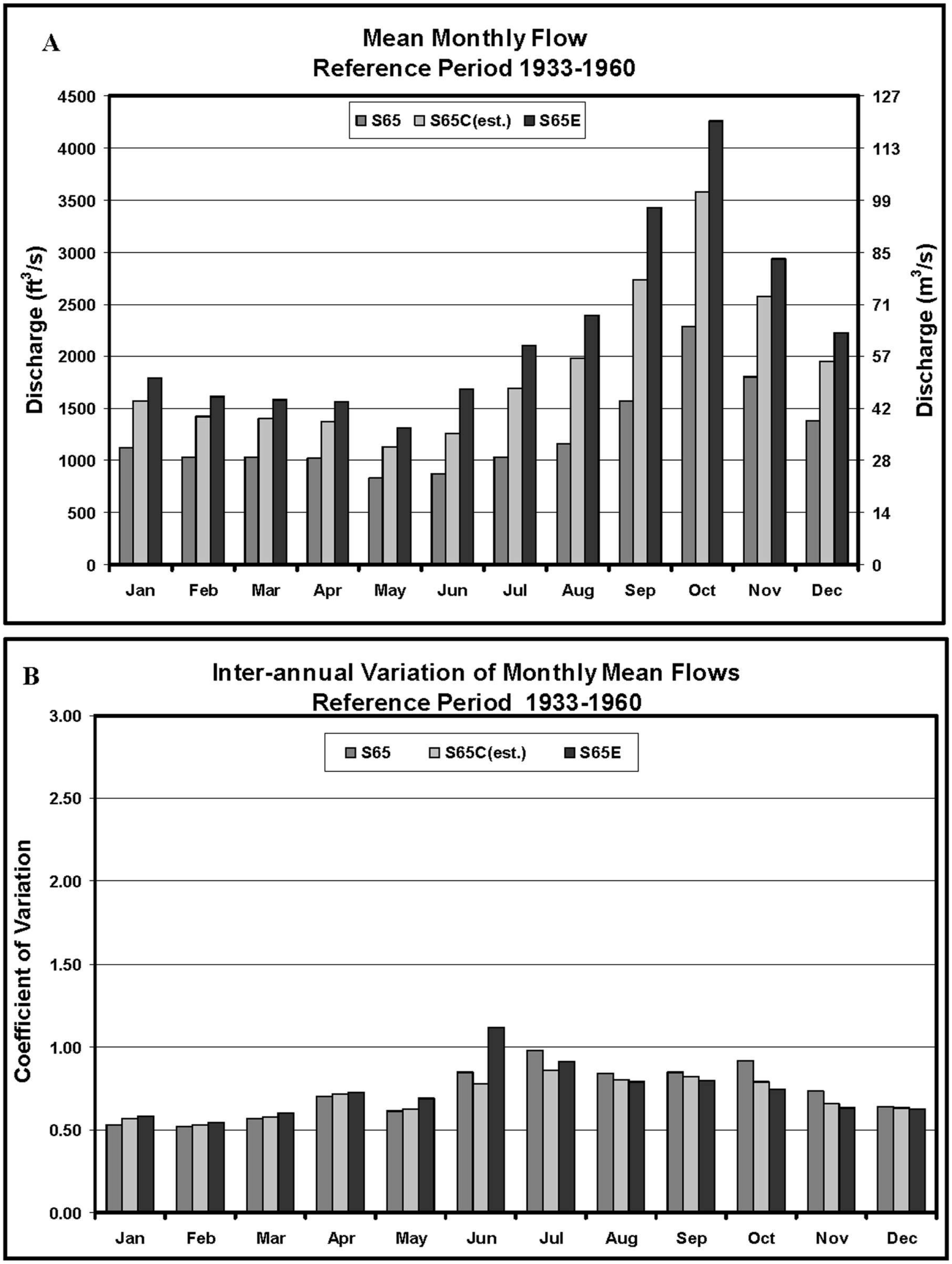


Figure 2-15. A. Mean monthly flow and B. coefficient of variation of mean monthly flow for S-65, estimated S-65C, and S-65E for the reference period (1933–1960).

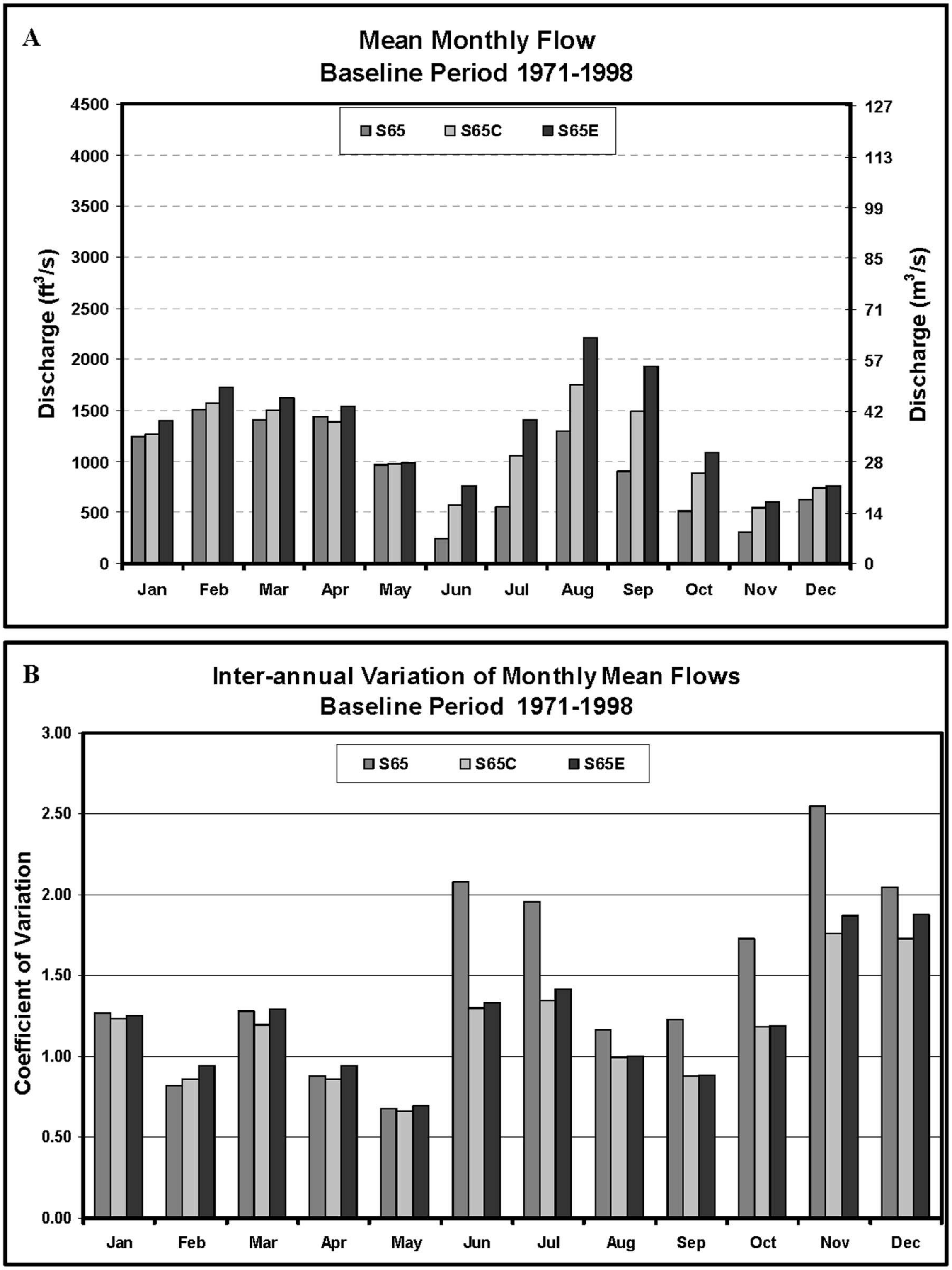


Figure 2-16. A. Mean monthly flow and B. coefficient of variation of mean monthly flow for S-65, S-65C, and S-65E for the baseline period (1971–1998).

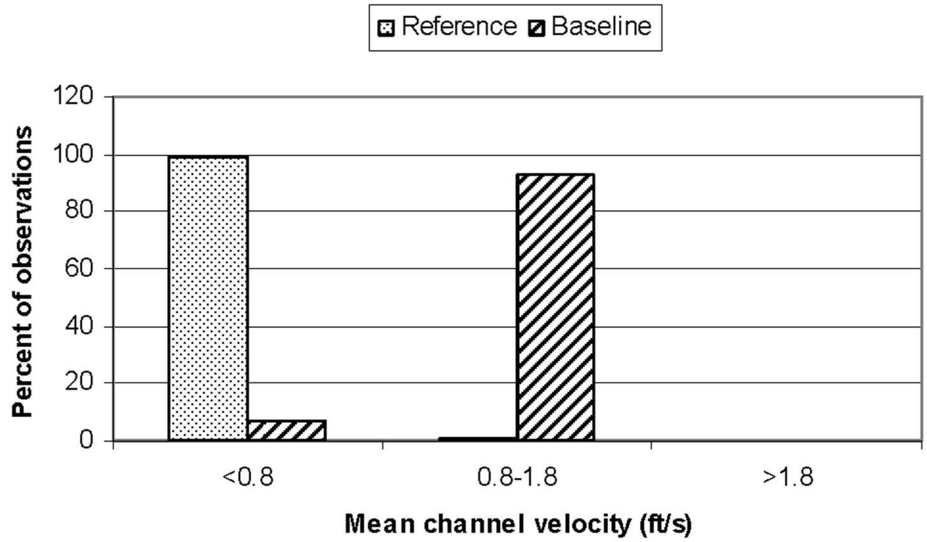


Figure 2-17. Percent of observations of mean channel velocity that were <0.8 ft/s, 0.8–1.8 ft/s, and >1.8 ft/s during the reference and baseline periods.

Table 2-4. Historic stage recession rates at Fort Kissimmee. Recession events exceeding 1 ft/30d are in bold.

Year	Start Date	End Date	Start Stage (ft)	End Stage (ft)	Change in Stage (ft)	Duration (days)	Rate (ft/day)	Rate (ft/30days)	# of Events per Year
1942-43	3-Sep-42	12-May-43	45.7	40.9	4.8	251	0.02	0.58	1
1943-44	11-Oct-43	5-Jun-44	45.2	40.9	4.3	238	0.02	0.55	1
1944-45	26-Oct-44	20-Jun-45	45.5	41.1	4.4	237	0.02	0.55	1
1945-46	18-Sep-45	13-May-46	50.1	43.3	6.8	237	0.03	0.87	1
1946-47	22-Sep-46	12-Feb-47	46.3	43.9	2.4	143	0.02	0.5	1
1947-48	23-Sep-47	1-Jul-48	49.8	43.9	6	282	0.02	0.63	1
1948-49	4-Oct-48	31-May-49	49.7	41.9	7.9	239	0.03	0.99	1
1949-50	1-Oct-49	28-Aug-50	48.1	40.7	7.4	331	0.02	0.67	1
1950-51	31-Oct-50	30-Mar-51	44.4	42.8	1.5	150	0.01	0.31	2
	22-Apr-51	**27-Jun-51**	**44.8**	**41.8**	**3.1**	**66**	**0.05**	**1.39**	
1951-52	20-Nov-51	30-Jun-52	45.2	43.3	2	223	0.01	0.26	1
1952-53	23-Oct-52	3-Jun-53	47	43.6	3.4	223	0.02	0.46	1
1953-54	No Data	---	---	---	---	---	---	---	---
1954-55	1-Oct-54	19-Jun-55	45.4	40.6	4.9	261	0.02	0.56	1
1955-56	13-Sep-55	27-May-56	44.1	38.6	5.5	257	0.02	0.64	1
1956-57	17-Oct-56	20-Feb-57	47.3	43.5	3.8	126	0.03	0.9	1
1957-58	5-Oct-57	22-Dec-57	46.5	44.8	1.7	78	0.02	0.66	1
1958-59	28-Jan-58	22-Jan-59	46.4	42.1	4.4	359	0.01	0.36	1

The duration of recession events at Fort Kissimmee (Table 2-4) ranged from 66 to 359 days and averaged 218 days. Stage recession rates ranged from 0.26 to 1.39 ft (8 to 42 cm) per 30 days. Only one of the 17 recession events exceeded 1.0 ft (30 cm) per 30 days. In April 1951, a dry season rainfall event caused stages to rise briefly before receding to a seasonal low in June. This recession event lasted 66 days, with water levels receding at a rate of 1.39 ft (42 cm) per 30 days.

At Fort Basinger, 22 recession events were identified (Table 2-5). These events lasted from 16 days to 355 days and averaged 173 days. Stages recession rates ranged from 0.27 to 1.93 ft (8 to 59 cm) per 30 days. For seven recession events, the recession rates exceeded 1.0 ft (30 cm) per 30 days and were associated with unusual weather conditions. Three events (April 1944, 1951 and October 1957) resulted from aberrant dry season rainfall, which caused stages to rise briefly before receding to a seasonal low in June. During the recession event of 1948–1949, stage decreased by 8.9 ft (271 cm) and followed two extremely wet years resulting from hurricanes in the Kissimmee basin. In 1955–1956, two of three recession events had short durations (< 20 days) and occurred early in the wet season prior to the normal seasonal stage recession period from September to May. The October 1956 to February 1957 event lasted 121 days and occurred during a severe drought, which was followed by rainfall that caused stages to increase until October 1957.

Table 2-5. Historic stage recession rates at Fort Basinger. Recession events exceeding 1 ft/30d are in bold.

Year	Start Date	End Date	Start Stage (ft)	End Stage (ft)	Change in Stage (ft)	Duration (days)	Rate (ft/day)	Rate (ft/30days)	# of Events per Year
1942-43	3-Oct-42	21-May-43	31.4	26	5.4	230	0.02	0.71	1
1943-44	5-Oct-43	29-Mar-44	32.2	27.4	4.8	176	0.03	0.82	2
	19-Apr-44	**6-Jun-44**	**29.2**	**26.2**	**3.1**	**48**	**0.06**	**1.93**	
1944-45	5-Nov-44	21-Jun-45	30.7	25.8	4.9	228	0.02	0.64	1
1945-46	22-Sep-45	14-May-46	34.6	28	6.6	234	0.03	0.85	1
1946-47	17-Sep-46	11-Feb-47	31.2	28.8	2.3	147	0.02	0.48	1
1947-48	24-Sep-47	3-Jul-48	34.9	29.2	5.7	283	0.02	0.6	1
1948-49	**6-Oct-48**	**1-Jun-49**	**35.5**	**26.6**	**8.9**	**238**	**0.04**	**1.12**	1
1949-50	5-Oct-49	4-Jun-50	33.2	27	6.2	242	0.03	0.77	1
1950-51	26-Oct-50	6-Apr-51	31.6	27.5	4.1	162	0.03	0.76	2
	24-Apr-51	**27-Jun-51**	**30.7**	**27.8**	**2.9**	**64**	**0.05**	**1.37**	
1951-52	3-Oct-51	25-Mar-52	32.8	29.2	3.6	174	0.02	0.62	1
1952-53	28-Oct-52	4-Jun-53	32.7	29.4	3.3	219	0.02	0.46	1
1953-54	13-Oct-53	25-May-54	36.1	29.5	6.6	224	0.03	0.88	1
1954-55	20-Jun-54	10-Jun-55	32	25.9	6.1	355	0.02	0.52	1
	4-Jul-55	**20-Jul-55**	**29.5**	**27.4**	**2.1**	**16**	**0.13**	—	
1955-56	**11-Aug-55**	**30-Aug-55**	**29.4**	**27.4**	**2**	**19**	**0.11**	—	3
	19-Sep-55	29-May-56	28.9	24	4.9	253	0.02	0.58	
1956-57	**21-Oct-56**	**19-Feb-57**	**33.2**	**28.1**	**5.2**	**121**	**0.04**	**1.28**	1
1957-58	**7-Oct-57**	**22-Dec-57**	**32.5**	**29.6**	**2.9**	**76**	**0.04**	**1.13**	2
	3-Feb-58	20-Jun-58	31.6	30.4	1.2	137	0.01	0.27	
1958-59	20-Jul-58	25-Dec-58	30.8	26.8	4	158	0.03	0.76	1

Baseline Conditions. Baseline conditions were derived from daily average headwater stage at S-65C and S-65D from 1971 to 1998. During the baseline period, stages in Pools C and D were a function of operational schedules for water control structures S-65C and S-65D. Stages typically fluctuated within 0.5 ft (15 cm) of control elevations (Figure 2-6). The lack of water level fluctuations produced no stage recession events during the baseline period.

Stage Hydrographs

The intent of the stage hydrograph criterion was to recreate floodplain inundation frequencies that would result in pre-channelization hydroperiods on the floodplain. Pre-channelization stage data were unavailable for floodplain sites. However, river channel stage data were available and the differences in frequency distributions between the reference and baseline periods were described above for Fort Kissimmee (Figure 2-7), Fort Basinger (Figure 2-8), and S-65E (Figure 2-9). These comparisons showed that changes occurred between the reference and baseline periods in the range of stages and number of days that water levels occurred at a given stage. Six metrics were evaluated that could be used to characterize stage hydrographs for a water year (Table 2-1). The first four metrics describe the stages for a water year: (1) maximum stage during a water year, (2) minimum stage during a water year, (3) average stage during a water year, and (4) median stage during a water year. The fifth metric was the change in stage during a water year, which was the difference between the maximum and minimum stage. The sixth metric considered stage relative to floodplain elevation as an indication of the actual inundation of the floodplain: number of days with stage above the average ground elevation. These were compared to mean floodplain elevation to determine when the floodplain was inundated. This comparison underestimated the actual floodplain inundation because it uses an average rather than a minimum floodplain elevation. All of the metrics were evaluated for the reference and baseline periods using stage data from Fort Kissimmee, Fort Basinger, and S-65E (Table 2-6).

Table 2-6. Characteristics of metrics used to describe the stage hydrograph criterion at Fort Kissimmee, Fort Basinger, and S-65E.

Site	Period	N	Range	Avg (SE)	CV	Median
Maximum stage						
Fort Kisssimmee[1]	Reference	20	43.72–50.12	46.87 (0.45)	0.04	46.39
	Baseline	15	41.02–46.76	44.80 (0.46)	0.04	45.27
Fort Basinger[2]	Reference	27	30.66–37.34	32.83 (0.31)	0.05	32.39
	Baseline	1	27.28	27.28		27.28
S-656E[3]	Reference	32	19.40–19.31	26.00 (0.38)	0.08	26.13
	Baseline	28	21.17–22.23	21.46 (0.05)	0.01	21.36
Average stage						
Fort Kissimmee	Reference	20	41.06–46.90	44.18 (0.36)	0.04	44.35
	Baseline	15	39.39–46.62	41.47 (0.36)	0.02	41.33
Fort Basinger	Reference	27	26.60–32.10	29.67 (0.26)	0.04	29.87
	Baseline	1	26.94	26.94		26.94
S-656E	Reference	32	16.40–26.19	22.30 (0.38)	0.10	22.57
	Baseline	28	20.86–21.19	21.07 (0.01)	0.00	21.08
Median stage						
Fort Kissimmee	Reference	20	41.06–46.95	44.30 (0.33)	0.03	44.25
	Baseline	15	39.87–42.72	41.40 (0.20)	0.02	41.32
Fort Basinger	Reference	27	26.33–32.25	29.85 (0.25)	0.04	30.05
	Baseline	1	26.97	26.97		26.97
S-656E	Reference	32	16.32–26.20	22.40 (0.40)	0.10	22.94
	Baseline	28	20.96–21.16	21.09 (0.01)	0.00	21.09
Change in stage						
Fort Kissimmee	Reference	20	2.59–9.02	4.96 (0.39)	0.35	4.60
	Baseline	15	2.98–8.29	5.89 (0.40)	0.26	6.16
Fort Basinger	Reference	27	2.10–12.74	5.76 (0.45)	0.41	5.52
	Baseline	1	1.25	1.25		1.25
S-656E	Reference	32	2.77–11.22	6.82 (0.38)	0.32	6.76
	Baseline	28	0.18–1.49	0.82 (0.06)	0.41	0.81
Inundation						
Fort Kissimmee	Reference	20	17–366	252 (23.64)	0.42	268
	Baseline	15	0–173	57 (15.07)	1.02	27
Fort Basinger	Reference	27	29–366	270 (17.81)	0.34	299
	Baseline	1	0	0		0
S-656E	Reference	32	0–366	248 (19.64)	0.45	274
	Baseline	28	89–366	288 (13.40)	0.25	298

[1] For Fort Kissimmee, the reference period was WY 1943–1962 and the baseline period was WY 1985–1999.
[2] For Fort Basinger, the reference period was WY 1933–1959 and the baseline period was WY 1999.
[3] For S-65E, the reference period was WY 1931–1962 and the baseline period was WY 1972–1999.

Box plots were used to screen the six metrics as potential indicators of change in stage characteristics, by comparing the pre-channelization reference period for each site with the channelized baseline period (Figure 2-18). The range of values for maximum, minimum, mean, and median stage was strongly influenced by location of the monitoring site because of changes in ground elevation along the river (i.e., ground elevation decreases downstream). For Fort Kissimmee, there was no clear difference between the reference and baseline period the maximum stage because of overlapping interquartile ranges (25^{th}–75^{th} percentiles) represented by the boxes. For minimum, maximum, and median stages, the interquartile ranges did not overlap between the reference and baseline periods at Fort Kissimmee. Box plots were not constructed for Fort Basinger during the baseline period because the single water year of data was insufficient to assess the variability among water years. Pre-channelization reference and channelized baseline vales at S-65E were similar.

Change in stage during a water year and inundation (number of days that river channel stage exceeded average ground elevation) appeared independent of ground elevation effects because the interquartile ranges (boxes) of the reference periods broadly overlapped regardless of the location along the river channel. The 25^{th} percentile for change in stage was at least 3.75 feet at all three sites for the reference period (Figure 2-18), suggesting that a reasonable expectation would be a fluctuation in stage of at least 3.75 feet in most years. For inundation, the 25^{th} percentile was at least 180 d so a reasonable expectation might be for river channel stage to exceed the average ground elevation by at least 180 d in most years. Before channelization began in 1962, the water years that the inundation metric had values less than 180 d were usually associated with drought periods (1955–1957, 1961–1963, Abtew et al. 2004) at Fort Kissimmee (Figure 2-19), Fort Basinger (Figure 2-20) and S-65E (Figure 2-21).

Baseline for Phase I

For the expanded hydrologic monitoring network for Phase I, the baseline period began on August 26, 1996, with activation of the first of the new sites (PC21), and continued through January 12, 1999. Most sites were established in late summer–early fall of 1998, which allowed less than a full year of data to be collected before Phase I backfilling began. The climatic conditions during this period were unremarkable, except for an El Niño event that lasted from November 1, 1997 through March 31, 1998. Rainfall for the upper Kissimmee Basin was 16.53 inches above average for this period and 21.38 inches above average for the lower basin (Huebner 2000). This El Niño event occurred during WY1998, and the annual rainfall for that year was 36.59 inches in the upper basin and 38.81 for the lower basin (Figure 2-4A). Both upper and lower basin values were below the long term average for both the baseline and reference periods. No hurricanes or tropical storms passed over the basin during this time period (Appendix 2-3A and 2-4A).

The movement of water through the Pool C reach of the C-38 canal was regulated by the S-65B structure at the upstream end of the pool, and by S-65C at the outflow. These structures were operated to maintain the stage in Pool C at 34 ft. For most of the baseline period, stages in the C-38 canal fluctuated within a 0.5 foot range. The tailwater stage at S-65B indicated the stage at the upstream end of the C-38 canal in Pool C and varied between 33.8 and 34.4 ft for most of the period (Figure 2-22). The S-65C headwater stage indicated that the stage at the downstream end of Pool C varied between 33.4 ft and 34 ft. These differences reflect the relatively flat water surface. Using 8.5 miles as the length of Pool C (Abtew 1992), the slope of the water surface in Pool C during the Phase I baseline period was frequently 5×10^{-6} and never exceeded 2.3×10^{-5}. These values are much lower than the range of slopes for the natural channel bed of 5.7×10^{-5} to 9×10^{-5} reported by Warne et al. (2000).

In remnant river channels, stages also approximated 34 ft (Figure 2-22) because these channels were directly connected to the C-38 canal. On the floodplain, position from upstream to downstream along Pool C had relatively little influence on stage. However, stage varied with location across the width of the floodplain because of changes in ground elevation. Sites located closer to the edge of the floodplain, such as PC51 and PC55, tended to have higher and more variable stages. This probably was the result of rainfall, especially the heavy rainfall associated with the November 1997–March 1998 El Niño event. Because of the relatively flat water surface in Pool C, sites located on the floodplain in the northern half of the pool had stages that tended to be below ground level (Figure 2-23). The number of days that water exceeded ground level tended to increase downstream until PC12, which was inundated throughout the baseline period.

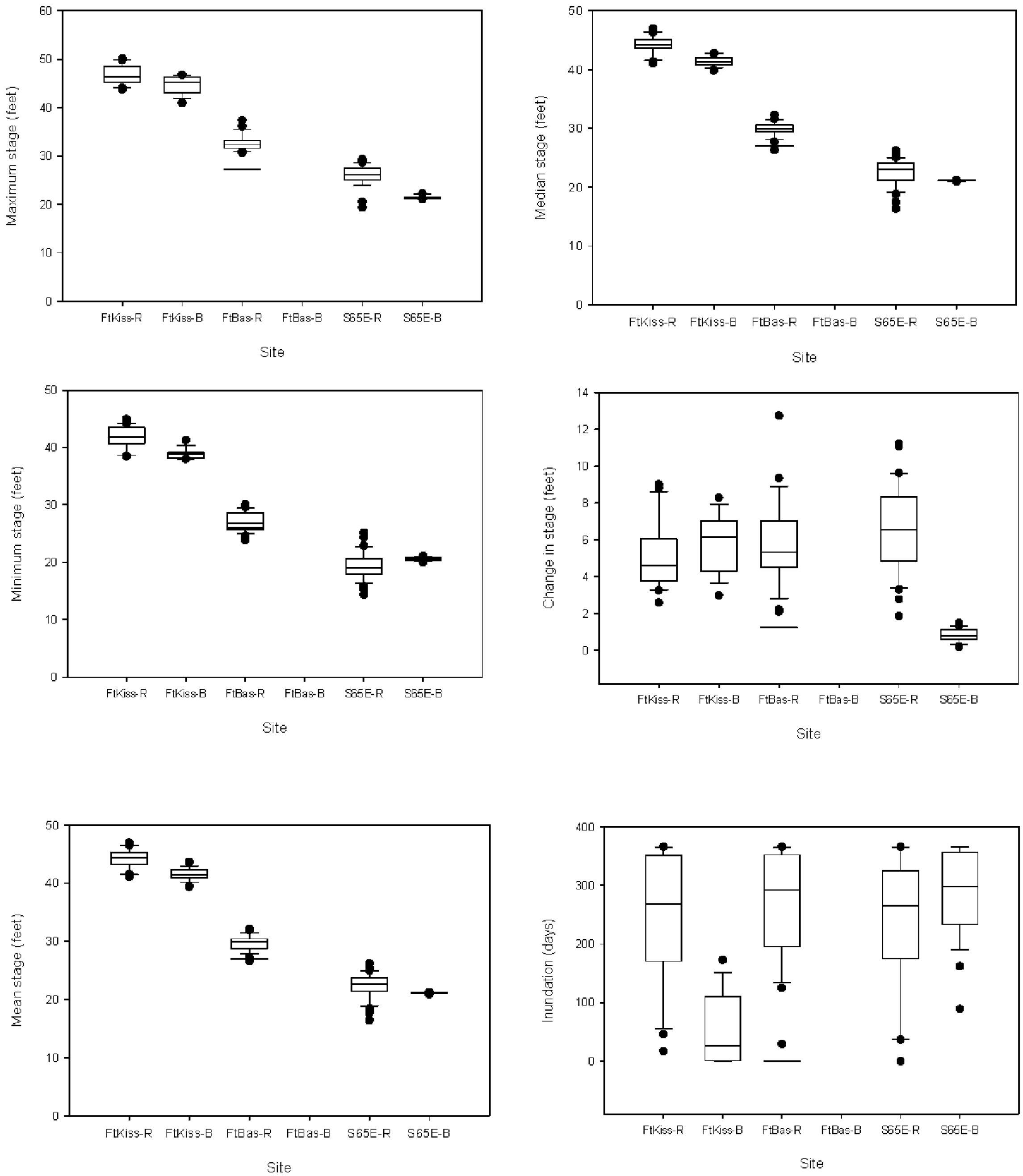


Figure 2-18. Box plots for six metrics describing stage by years for different sites. Sites were Fort Kissimmee during the reference period (FtKiss-R) and baseline (FtKiss-B) periods, Fort Basinger during reference (FtBas-R) and baseline (FtBas-B) periods, and S-65E during reference (S65E-R) and baseline (S65E-B) periods. A box plot was not constructed for Fort Basinger during the baseline period because the single water year of data during the baseline period was not sufficient. In the box plots, the horizontal line is the median, the ends of the box represent the 25th and 75th percentiles and the error bars represent the 10th and 90th percentiles. Filled circles are values <10th or >90th percentile.

Mean daily discharges at S-65, PC33, S-65C, and S-65E were parallel during the baseline period for Phase I (Figure 2-24). Discharge at PC33 was much lower than the other sites because PC33 measures discharge only through the remnant river channel, while the canal sites measure all of the water except for small volumes passing through auxiliary structures in the tieback levee. The similarity of flow among the canal sites indicated the importance of the outflow from Lake Kissimmee at S-65 in determining discharge through the system. While PC33 was located in a remnant channel, discharges as high as 800 cfs were measured at that site during the baseline period. Most of these high values occurred during the El Niño period (November 1997–March 1998) when the lower basin in particular was receiving greatly elevated rainfall. These flows may have been heavily influenced by floodplain runoff or inflows from Oak Creek, a tributary to Micco Bluff Run just upstream of PC33.

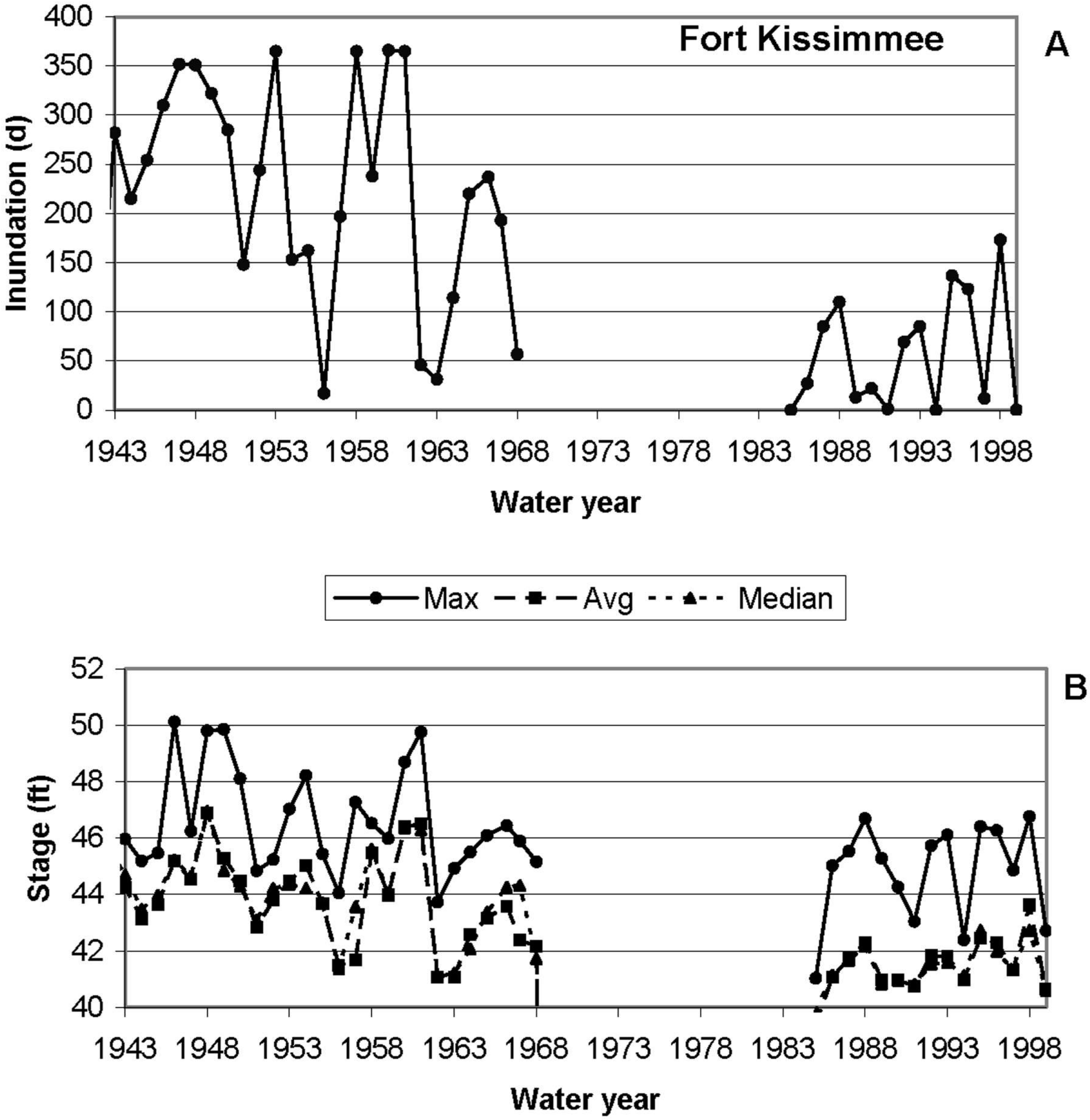


Figure 2-19. A. Number of days that the floodplain was inundated by stages exceeding the mean floodplain elevation of 43 feet (Obeysekera and Loftin 1990) and B. the maximum, average, and median stage for a water year at Fort Kissimmee.

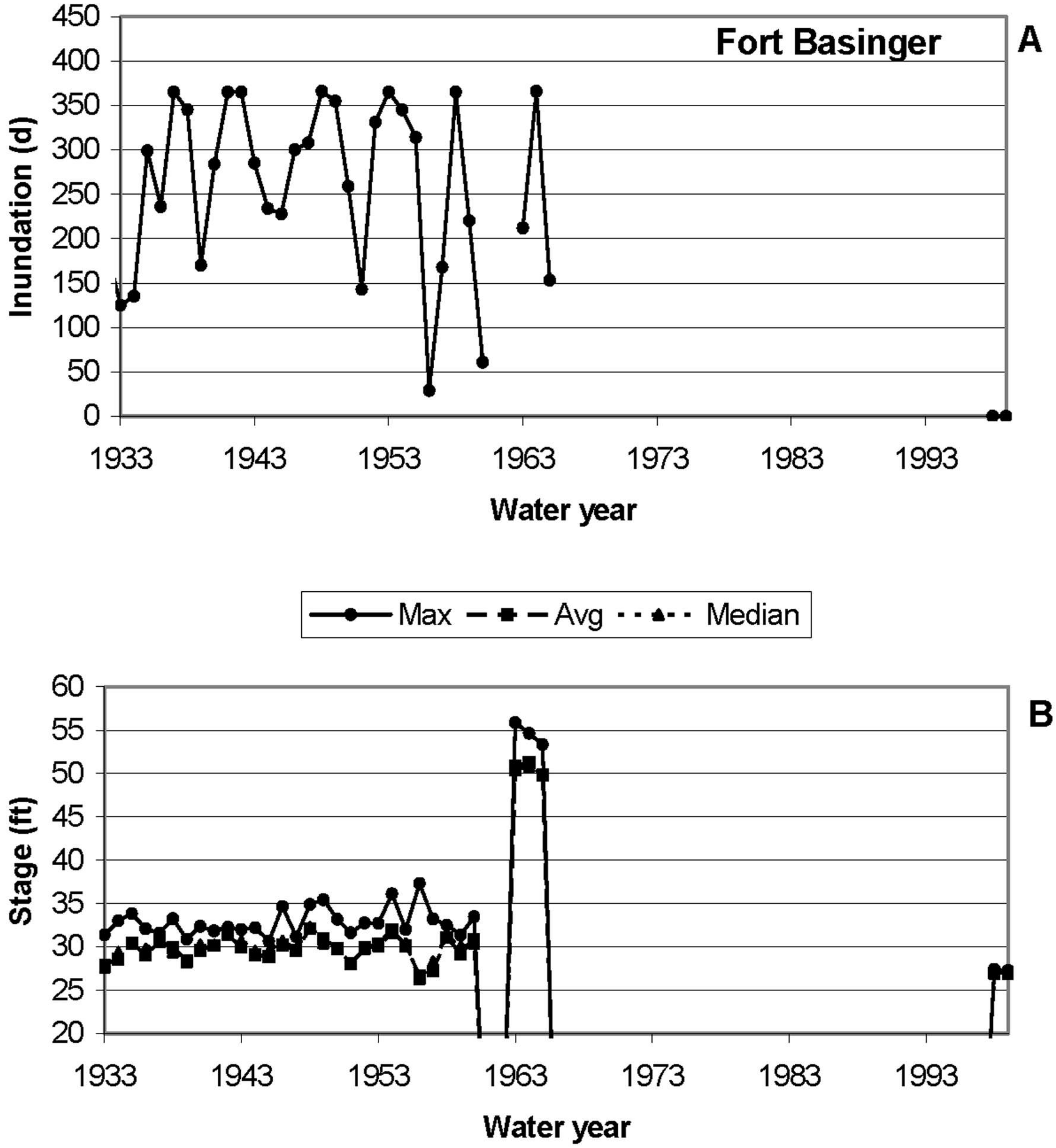


Figure 2-20. A. Number of days that the floodplain was inundated by stages exceeding the mean floodplain elevation of 28.5 feet (Obeysekera and Loftin 1990) and B. the maximum, average, and median stage for a water year at Fort Basinger.

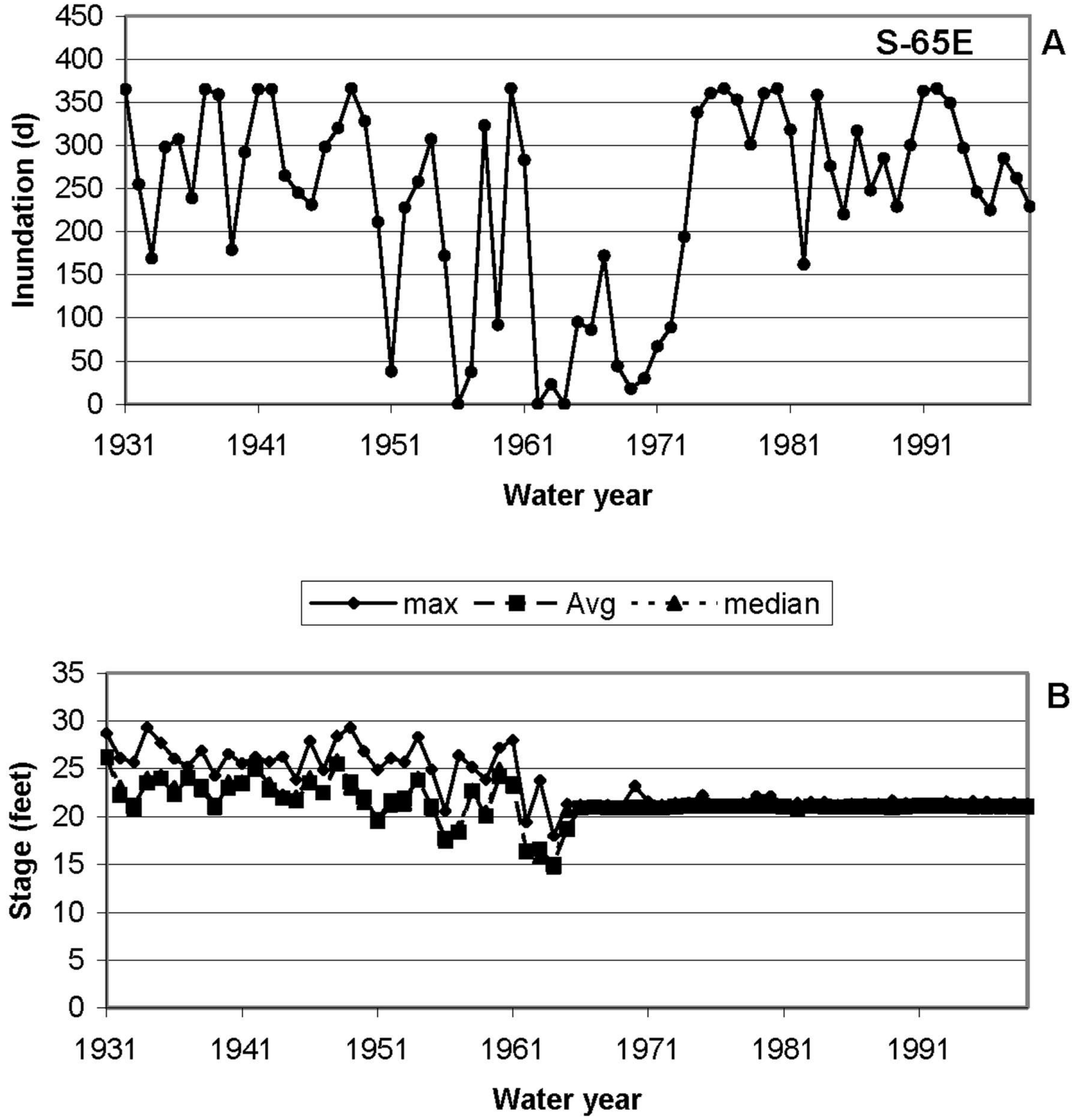


Figure 2-21. A. Number of days that the floodplain was inundated by stages exceeding the mean floodplain elevation of 21 feet (Obeysekera and Loftin 1990) and B. the maximum, average, and median stage for a water year at S-65E.

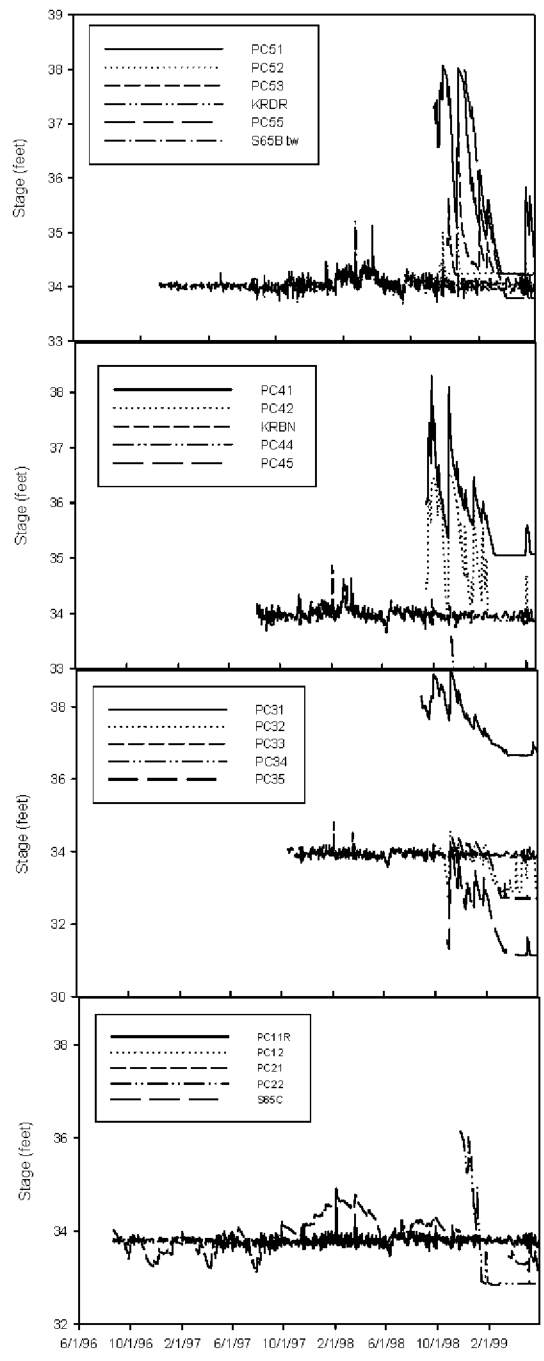


Figure 2-22. Mean daily stage data from the Phase I baseline monitoring network. Data presentation was truncated at the end of the baseline period on May 31, 1999. All stations were located on the floodplain except PC11R, PC33, KRBN, and KRDR, which were located in remnant river channels, and the tailwater of S-65B and the headwater of S-65C, which were located in the C-38 canal.

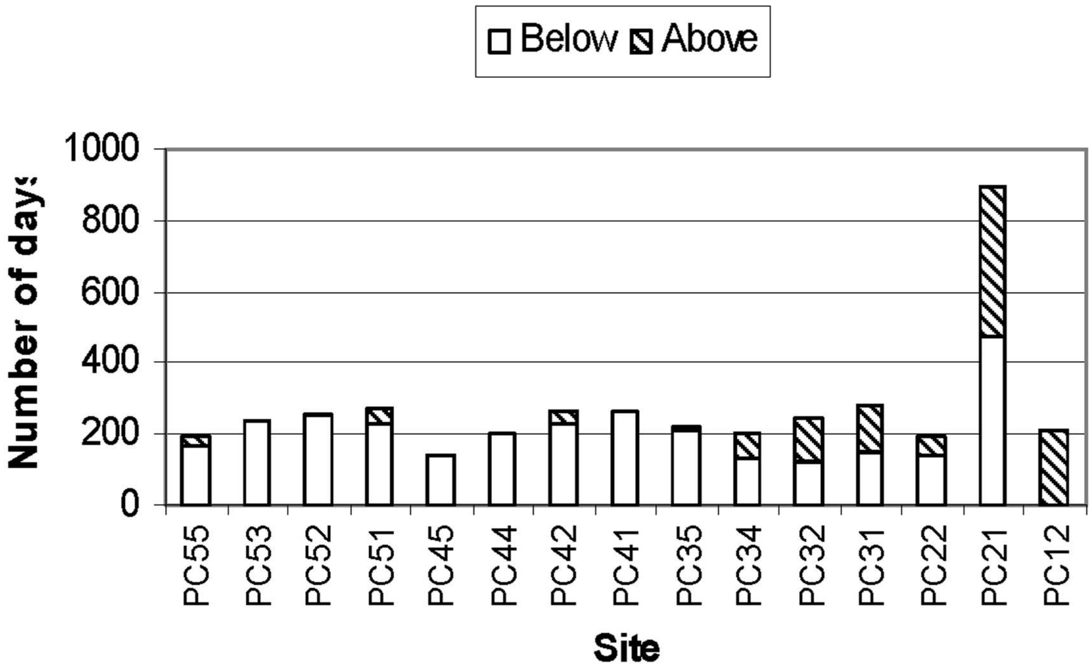


Figure 2-23. Number of days that stage was above or below ground level for each floodplain site during the baseline period (August 22, 1996–May 31, 1999).

DISCUSSION

Long-term changes

Long Term Data Issues

The ability to make inferences about changes in a long-term data series depend on the availability of data, continuity of the time series, compatibility of the data over time, and representation of the periods of time of interest. These issues are discussed in order below.

Availability. Retrospective analyses are constrained by the existence of data collected in the past, usually for some other purpose. Hydrologic monitoring stations along the Kissimmee River were established originally by the U. S. Geological Survey and later assumed by the South Florida Water Management District. None of the pre-channelization stations occur within the area of Phase I of the restoration project and only the Fort Basinger site occurs within the restoration project domain. However, the similarities between the upstream S-65 and downstream S-656E sites should bracket the flows into and out of the project area. All the available data came from river channel sites and not from the floodplain, making it difficult to characterize floodplain hydroperiods. Fairly long data sets are available for the pre-channelization period at each site.

Compatibility. The data used for the long-term comparisons should be appropriate for this type of analysis. Both the U. S. Geological Survey and the South Florida Water Management District followed similar protocols for collecting stage and flow data. Both agencies use the same standard for calibrating stage monitoring equipment to within 0.02 feet (Rantz and others 1982). Changes observed in this study were much greater than the error that might be associated with calibration. Instrumentation has been upgraded as new technology has become available. For stage measurement, these changes have mainly involved instrumentation for recording and transmitting data and not the actual stage measurement. Another compatibility issue involves stations that were discontinued and then reactivated, such as Fort Kissimmee and Fort Basinger. When Fort Kissimmee was reactivated, it was on the same site. The Fort Basinger site was discontinued after the river channel at its location was destroyed by construction of the C-38 canal. When the site was reactivated, it was located in a nearby remnant channel within 1000 ft of the original location.

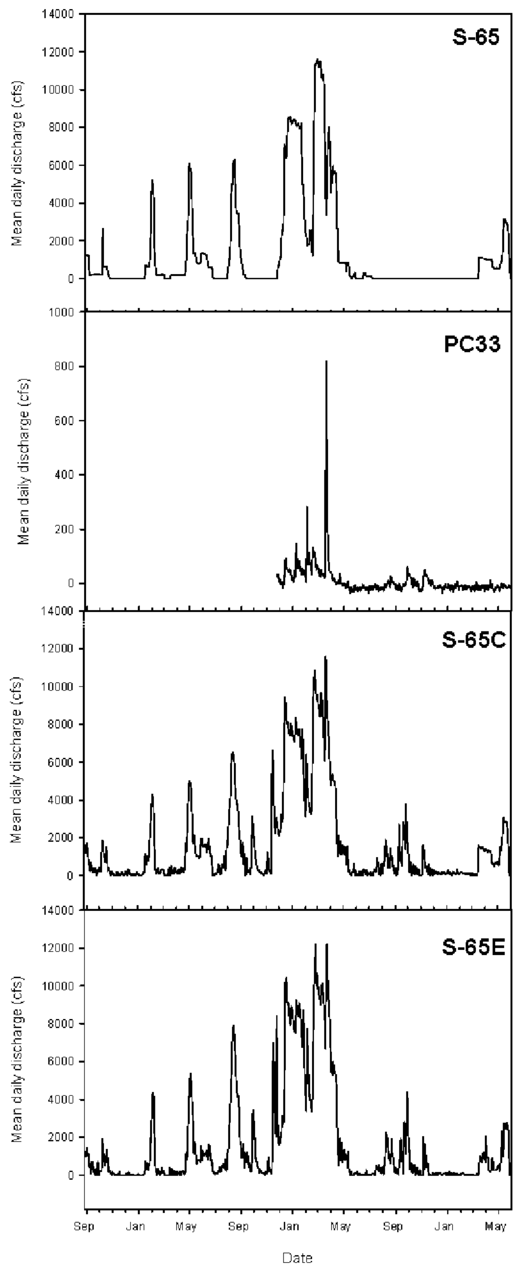


Figure 2-24. Mean daily discharge during the baseline period (August 26, 1996–May 31, 1999) at S-65, PC33, S-65C, and S-65C. Discharge measurements at S-65, S-65C, and S-65E were made at the structure on the C-38 canal and represent the water moving through the system. PC33 was located in a remnant river channel, and discharge measurements at this site are only for the remnant river channel and did not include the floodplain or the C-38 canal. The x-axis begins with September 1996 and continues through May 1999.

Continuity. Issues involving continuity include missing data because of equipment problems or discontinued stations. Fairly complete records of stage and discharge were available for S-65 and S-656E. Fort Kissimmee and Fort Basinger both had long gaps because the stations were discontinued. Fort Kissimmee was reactivated only after a fluctuating stage regulation schedule was implemented for Pool B; the baseline period at this station was atypical of the baseline period from 1972–1985 at this site and any other site. Fort Basinger was reactivated only just before the beginning of Phase I of construction, so the baseline period consisted of only one water year.

Representation. The ability to characterize hydrologic conditions during the pre-channelization reference and channelized baseline periods depends on having a time series of data that is representative of those time periods. The period of record should be long enough to capture natural variability. Fairly long periods of record were available for S-65 and S-65E and for the pre-channelization reference period for Fort Kissimmee and Fort Basinger. However, if monitoring had begun one year earlier at S-65E or six years earlier at S-65, a flow event of magnitude that has never been measured at S-65 and only once at S-65E (in the late 1960s) would have been captured. This event involved two tropical storms that passed over the basin during August 7–14, 1928 and dumped 16.21 inches of rainfall in the St. Cloud area. This intense rainfall produced a stage of 29 feet and an estimated peak flow of 20,000 cfs at the Highway 70 bridge across the Kissimmee River, just upstream of the present location of S-65E (U. S. Army Corps of Engineers 1969).

Confounding Factors

The hydrologic conceptual model identified climate as an important driver that could change over time and be confounded with channelization. Mean annual rainfall to both the upper and lower Kissimmee Basins decreased by approximately 10% between the reference and baseline periods (Obeysekera and Loftin 1990). Consequently, severe droughts in the Kissimmee Basin appear to have occurred less frequently during the reference period than the baseline period. Abtew et al. (2004) identified three severe drought events (1932, 1955–1957, 1961–1963) during the reference period and seven severe events (1971–1972, 1973–1974, 1980–1982, 1985, 1988–1989, 1990) during the baseline period. Other climatic changes involve El Niño and La Niña events that can affect the seasonality of rainfall over the Kissimmee Basin (Huebner 2000, Schmidt et al. 2001).

Another potential confounding factor involves changes in land use in the watershed, which can alter rainfall–runoff relationships. Following channelization, a large portion of the lower basin was converted from unimproved pastures to improved pastures with a roughly three-fold increase in drainage density network (Obeysekera and Loftin 1990). Additionally, the human population of the counties that contribute to the basin has continued to grow. This increase in population size is likely to be accompanied by changes in land use, which could ultimately affect hydrologic characteristics. These relationships between land use changes and hydrology have not been quantified.

Changes in operation of the water control structures may also be a confounding factor. The regulation schedules for operating the water control schedules have changed several times during the baseline period (Appendix 2-1A). There were also short-term deviations to regulation schedules for some of the upper basin lakes to allow drawdowns for various lake management activities. Probably the most critical change to operating rules, already discussed, involved the implementation of a fluctuating stage regulation schedule for Pool B. Knowledge of this change was critical for interpreting stage data at Fort Kissimmee during the baseline period.

Impacts of channelization

This study identified several changes in hydrology that resulted from channelization and flow regulation. These changes include a narrowed range of stage fluctuation; more erratic discharge patterns, especially increasing the number of days with no flow; and shifts in the seasonality of flow. Channelization and excavation of the C-38 canal caused flow to be carried by the canal instead of by the natural river channel. Other studies determined that the lack of flow in remnant channels caused a proliferation of aquatic plants in the remnant channels (Bousquin 2005) and contributed to an accumulation of organic deposits in the river channel (Anderson et al. 2005). Loss of flow also reduced rates of reaeration, resulting in near constant hypoxic conditions (Colangelo and Jones 2005). These changes in flow greatly altered habitat characteristics and affected community structure for many plants (Bousquin 2005), invertebrates (Koebel et al. 2005), and fish (Glenn 2005). Floodplain inundation patterns were altered, and the effects vary along the length of a pool. At the downstream end of a pool, the floodplain is inundated nearly continuously, while at the upstream end, the floodplain is almost permanently dry. The effect of permanent inundation at the downstream end of the pool on wetland plant and animal communities is not well understood.

Developing Hydrologic Restoration Expectations

Evaluation of the hydrologic criteria

An early section of this chapter described the development of the five hydrologic criteria for the Kissimmee River Restoration Project (Loftin et al. 1990b, U. S. Army Corps of Engineers 1991). These are treated as six criteria in this section by separating the seasonal variability component from the continuous flow criterion. Metrics were evaluated for each criterion except stage-discharge relationships. This criterion was the only one that could not be related to one of the characteristics of the natural flow regime. While the criterion is described as stage-discharge relationship, the emphasis is on having bankfull discharge in the range of 1400–2000 cfs. These values are based on changes in the shape of a plot of mean channel velocity against discharge (Figure 7.7 in Huber et al. 1976) at different locations along the river (Warne et al. 2000). The lower end of the range (1400 cfs) applies to the upper reaches of the Kissimmee River and the upper end (2000 cfs) applies to the S-65E location. For the remaining five criteria, at least one metric was identified that showed a difference between the reference and baseline periods, suggesting that it was affected by channelization and may respond to restoration.

Developing Restoration Expectations

Five expectations are proposed for evaluating the restoration of hydrology:

(1) The number of days that discharge equals 0 cfs in a water year will be zero for restored channels of the Kissimmee River (Anderson and Chamberlain 2005a).
(2) Intra-annual monthly mean flows will reflect historic seasonal patterns and have inter-annual variability (coefficient of variation) <1.0 (Chamberlain and Anderson 2005).
(3) River channel stage will exceed the average ground elevation for 180 d per water year and stages will fluctuate by 3.75 feet (Anderson and Chamberlain 2005b).
(4) An annual prolonged recession event will be reestablished with an average duration ≥ 173 days and with peak stages in the wet season receding to a low stage in the dry season at a rate not to exceed 1.0 ft (30 cm) per 30 days (Chamberlain 2005a).
(5) Mean velocities within the main river channel will range from 0.8 to 1.8 ft/s (0.2 to 0.6 m/s) for a minimum of 85% of the year (Chamberlain 2005b).

One of the requirements for developing restoration expectations is to identify external constraints, which may necessitate adjusting the expected values. Most of the hydrologic expectations have the potential to be affected by regulation schedules and operational rules. Climatic patterns are another constraint. An extreme shift in rainfall conditions such as a severe drought may make it difficult to achieve expected hydrologic conditions.

Another requirement for developing expectations is to specify a mechanism that links the restoration project to the anticipated changes. Each hydrologic expectation will be achieved through similar mechanisms. The steps in the mechanism involve backfilling the C-38 canal, carving new channels to reconnect the remnant river channels, removing the S-65B and S-65C water control structures, and implementing the Kissimmee River Headwaters Revitalization stage regulation schedule for S-65.

A final requirement for developing an expectation is to specify a trajectory for achieving the expectation.

Hydrology should respond almost instantly to the restoration project, and most expectations should start to show responses after backfilling of the canal and reconnection of the river channels. Seeing the full measure of the expected response may not be possible until the headwaters stage regulation schedule is implemented.

Restoration of Hydrology

When the construction phases of the restoration project are completed, the Kissimmee River will still be a managed system. Reestablishing pre-channelization hydrology to the river will be constrained by limitations of the physical system (canals and water control structures) and the rules for operating these structures. Modeling completed for the feasibility report (U. S. Army Corps of Engineers 1991, 1996) resulted in a regulation schedule for S-65 called the headwaters revitalization schedule that should result in a more natural seasonal distribution of flows, with less frequent periods of no flow, and stage frequencies that are more comparable to the pre-channelization river (U. S. Army Corps of Engineers 1996). Also, climatic factors can constrain the restoration of hydrology. Changing patterns in rainfall may make it more difficult to achieve expectations based on the pre-channelization condition. It may be useful or even necessary to link some hydrologic expectations to climatic conditions. Future studies should include examining rainfall – discharge relationships.

The restoration of ecological integrity in the Kissimmee River is tied to reestablishing pre-channelization hydrology. However, pre-channelization hydrology was variable and there is not *one hydrology* that can be recreated. Different attributes of the river-floodplain ecosystem depend on flow events of varying magnitude, durations, and frequencies (Whiting 2002). Continuous flow may be required to maintain dissolved oxygen concentrations, but the higher discharges required for channel maintenance may be needed less often than once a year. Flows of different magnitude and frequency are necessary to maintain integrity on the floodplain.

Restoration of hydrology may be much more complicated for the Kissimmee River than for some other river restoration projects. For example, a number of recent restoration projects have involved the removal of a dam so that flow regulation is no longer an issue. An extreme case is the destruction of dams in the Euphrates and Tigris Rivers to restore flow to the river delta wetlands. Restoration of hydrology in the Kissimmee River will have a strong management component that will require adaptive thinking.

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CHAPTER 3

RIVER CHANNEL GEOMORPHOLOGY OF THE CHANNELIZED KISSIMMEE RIVER, FLORIDA

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ABSTRACT: We established a baseline for evaluating river channel geomorphic responses in the first reach of the Kissimmee River scheduled for restoration (Impact area) and in an upstream Control area. Examination of aerial photography from 1994 revealed that none of the 53 meanders in the Control area or the 82 meanders in the Impact area had active point bars and that relict point bars were overgrown with vegetation. We also characterized channel geomorphology from depth measurements and core samples collected on transects across remnant river channels in Control and Impact areas. Remnant river channels, which lacked flow since channelization, had accumulated organic deposits on the natural channel bed substrate. These substrate-overlying deposits ranged from 1 to 98 cm in thickness and were composed primarily of organic matter. Cross-sectional profiles constructed from the depth to the channel bed substrate, which was predominately sand, appeared to retain the shape of the historic river channel. In the Impact area, substrate-overlying organic deposits reduced the average channel depth by 8%, increased the width/depth ratio by 13%, and reduced channel cross-sectional area by 8%. In the Impact area, mean thickness of substrate-overlying deposits averaged 14 cm, percent of samples without substrate-overlying deposits averaged 3%, and thickness of substrate-overlying deposits at the thalweg averaged 21 cm. The Control area contained more deposition above the substrate layer, with a 57% increase for mean thickness of substrate-overlying deposits and an 81% increase in the thickness of substrate-overlying deposits at the thalweg over the Impact area. Relative to reference values for an area with partially restored flow, values for the Impact area represent a two-fold increase in mean thickness of deposits overlying the natural channel bed substrate, a 95% reduction in the percent of samples without substrate-overlying deposits, and a two-fold increase in the thickness of substrate-overlying deposits at the thalweg.

INTRODUCTION

The structure and function of stream ecosystems are closely coupled to the morphology of the river channel, and morphology is strongly influenced by climate, basin physiography, and geology (Brussock et al. 1985, Knighton 1988). In the Kissimmee River basin, these factors include highly seasonal rainfall (average of 124 cm/yr), a narrow basin of low relief (channel slopes of 0.00006–0.00009 m/m), and a geology composed predominately of unconsolidated fine and medium grained sands (Warne et al. 2000). The river channel that developed in this setting was shaped by the interaction of discharge and unconsolidated sediments, which gave rise to a meandering and sometimes anastomosing river channel, with typical features such as point bars on the inside of meander bends. Historical analyses indicate that the Kissimmee River had high rates of channel migration, cutoff, and avulsion, which were related to high frequency (return interval <1 yr) of bankfull discharge (Warne et al. 2000). Other characteristics relevant to the geomorphology of the river are summarized in Table 3-1.

Table 3-1. Geomorphic characteristics of the Kissimmee River system based on Warne et al. (2000).

Characteristics	Values
Basin Area	7766km^2 (3000 miles^2)
Basin Length	152 km (95 miles)
Maximum basin width	62 km (39 miles)
Basin relief	94 m (309 ft)
Relief ratio[1]	0.6 m/km
Stream order (main channel)[2]	4th and 5th
Bank-full channel width	15 to 35 m (50 to 115 ft)
Bank-full channel depth at thalweg	15 to 35 m (50 to 115 ft)
Meander wavelength	125 m (410 ft), range 90 to 400 m (295 to 1300 ft)
Sinuosity[3]	1.67 to 2.1
Channel slope[4]	
in the north	0.09 m/km (0.0009 m/m)
in the south	0.057 m/km (0.00057 m/m)
Entrenchment ratio[5]	>20
Width to depth ratio[6]	7 to 9
Drainage density[7]	0.73 to 1.60 km/km^2 (1.17 to 2.58 mile/mile^2)
Floodplain width	1.5 to 3 km (0.9 to 1.8 mile)
Meander width ratio[8]	10 to 28
Bankfull discharge	
upper reaches	40 m^3/s (1400 cfs)
lower reaches	57 m^3/s (2000 cfs)
Mean annual discharge - lower reach	62 m^3/s (2166 ft^3/s)

[1] Ratio between basin relief and length of basin at its longest axis.
[2] Strahler (1957).
[3] Ratio of stream length per unit valley length.
[4] Koebel (1995).
[5] Width of the flood prone area (floodplain area inundated at twice the bankfull stage) to bankfull surface width.
[6] Bankfull channel width versus bankfull channel depth.
[7] Stream length per unit area of watershed.
[8] Ratio of meander wavelengths to overall width of flood prone area.

Channelization of the Kissimmee River through the excavation of the C-38 canal parallel to the length of the river valley, resulted in the destruction of river channel where the canal intersects the meandering channel, and diversion of flow from channel remnants to the deeper and wider canal (Figure 3-1). As the canal became the primary conduit for moving water, remnant channels no longer received the flow of water (Anderson and Chamberlain 2005) that creates the dynamic relationship between river velocities and sediment size, deposition, and transport, which characterizes a functioning system capable of transporting sediments (Leopold 1994). Anecdotal observations of the channelized river suggest that the absence of flow has allowed floating aquatic and littoral vegetation beds to encroach on mid-channel areas and contribute to a layer of organic deposition on the channel bottom (Vannote 1973, Milleson et al. 1980, Perrin et al. 1982). Also, herbicidal control of freely-floating aquatic vegetation has influenced the rate of organic deposition (Grimshaw 2002). The change in the composition of the channel bed contributed to depressed concentrations of dissolved oxygen, which decreased habitat quality for river fish communities (Perrin et al. 1982, Toth 1993). Toth (1991, 1993) determined that the substrate-overlying deposits, which were >1 m thick in some locations, contained organic matter and marl and filled in the channel and decreased its cross-sectional area. Reestablishment of flow, especially bankfull discharge, is likely to reverse these impacts of channelization. The Kissimmee River Demonstration Project provides some evidence that such a reversal is possible. When weirs were placed across the C-38 canal to divert water through three remnant river channels, these channels received flows $\geq$26 m^3/s, which approached bankfull discharge, for 233–307 d of a three and one-half year period. This increase in flow reduced the thickness and extent of substrate-overlying organic deposits and exposed the natural sand substrate (Toth 1991, 1993).

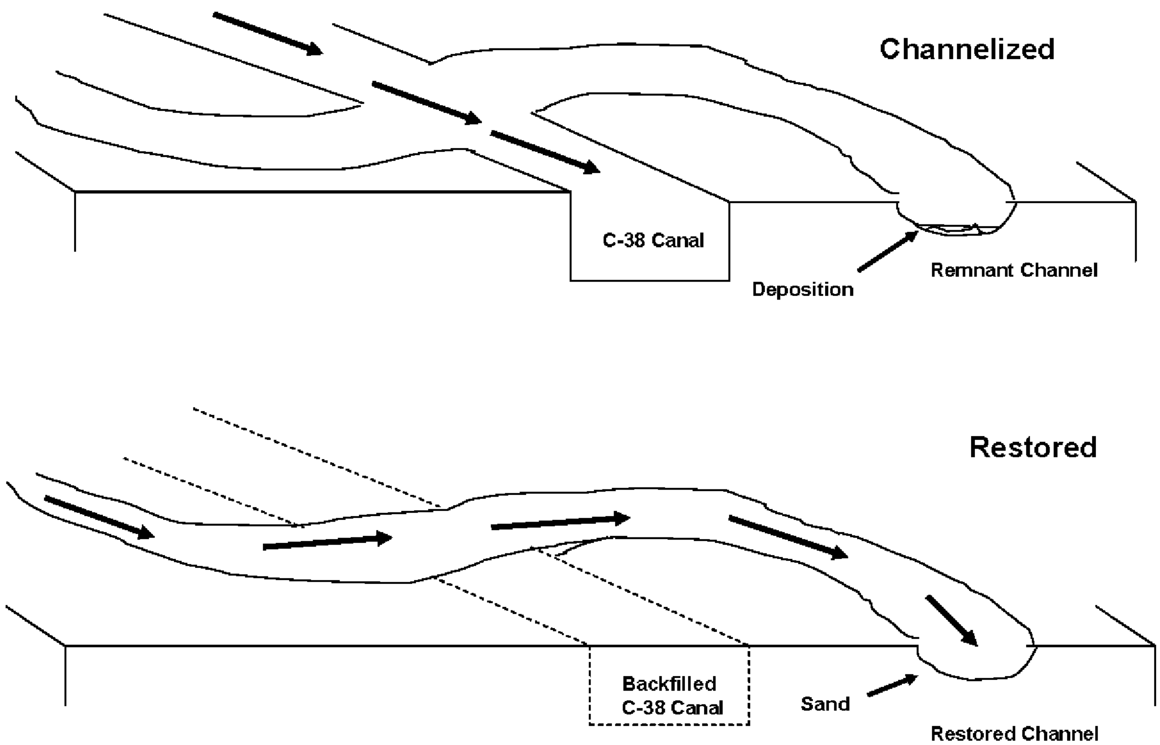


Figure 3-1. Schematic representation of a cross section through the Kissimmee River and its floodplain illustrating changes in geomorphology between the channelized system (top) and the restored system (bottom). Arrows indicate the direction of water flow through the C-38 canal in the channelized system and through the reconnected river channel in the restored system.

Objectives

This chapter characterizes aspects of fluvial geomorphology within the Kissimmee River that have been altered by channelization, and thus establishes a baseline for evaluating responses to restoration. Restoration will involve backfilling the canal, reconnecting remnant river channels, and reestablishing flows that approximate pre-channelization frequency and magnitude through the reconstructed system. To reconnect remnant river channels, new channels will be carved through the floodplain (new or recarved channels) and across the backfilled canal (connector channels). We focused baseline geomorphic studies on aspects of channel structure that were impacted by the virtual elimination of flow, especially the loss of bankfull discharge, which is essential for the transportation and deposition of sediments. While new channels and connector channels do not exhibit the effects of channelization, they were included in this chapter because data that characterizes their condition at the time of construction provides a baseline for evaluating their adjustments to flow following the restoration project. The specific objectives of this study included quantifying:

(1) The presence of point bars.
(2) Channel morphology.
(3) Characteristics of channel bed deposits.
(4) Riparian soils and vegetation.
(5) Reference conditions that could be used to develop restoration expectations.

METHODS

Study Sites

The Kissimmee River is located in south-central Florida and is channelized along its entire length by the C-38 canal, which is 9 m-deep and 30–100 m-wide. Where the canal intersects the meandering river channel, the larger canal obliterates the natural channel. The resulting remnants of the natural river channel remain connected to the canal and hold water but carry essentially no flow. During channelization, a series of water control structures were installed that divided the canal into a series of pools. Our study involved the longest remnant river runs in Pool A and Pool C and a short run in Pool B (Map Appendix 1A). We designated runs in Pool A as the Control area because they are upstream of the restoration project, and restoration activities (i.e., canal backfilling) should not reestablish flow to these runs. We designated runs in Pool C and one run (UBX) in Pool B as the Impact area because these runs will be reconnected during Phase I of the restoration project and will then carry flow.

Backfilling began in June 1999 and ended in February 2001. During this period, three new channels (Fulford, Strayer, and Loftin) were carved across the floodplain, and four connector channels (Loftin-Micco Bluff connector, Oxbow 13-Micco Bluff connector, Strayer-Fulford connector, and Montsdeoca-Fulford connector) were carved across the backfilled canal in the Impact area. The dimensions of these new and connector channels were determined by criteria specified in the construction plans for the restoration project (U. S. Army Corps of Engineers 1998). New channels were designed to match the characteristics of natural channels and were adjusted for channel pattern (i.e., curved or straight). Straight and slightly curving channels were carved with a bottom width of 64 feet (19.5 m) and sides with a 2:1 slope (2 horizontal feet for every vertical foot). Transitional channels were carved with a bottom width of 42 feet (12.8 m), a 2:1 slope on the exterior side of the curve, and a 4:1 slope on the interior side. Curved channels were given a bottom width of 30 feet (9.1 m), a side slope of 2:1 on the exterior of the curve, and a side slope of 7:1 on the interior of the curve. The width of connector channels was constructed to match the bottom width of the channels it joined on either side of the backfilled canal. The northern side of the connector channel had a 16:1 slope, and the southern side had a 4:1 slope.

Point Bar Presence and Absence

We examined aerial photographs of the channelized river with ArcView (Environmental Systems Research Institute) to determine if point bars were present on meander bends. These photographs (1:6000 scale) were taken at an altitude of 914 m between February 26, and March 15, 1994, when the river stage was 33.75–33.82 ft NGVD (10.29–10.32 m NGVD). The area examined covered meanders from all remnant river runs in Pool A, Pool C, and UBX run in Pool B.

Core Sampling

We characterized river channel sediments and riparian soils with core samples collected between November 1997 and February 1999 for remnant river channels, and in 2000 for new river channels and connector channels. Core samples were collected in remnant river channels on fixed transects that were established in 1988 (Montsdeoca, MacArthur, UBX) and in 1997 (Micco Bluff, Oxbow 13, Ice Cream

Slough, Rattlesnake Hammock, and Persimmon Mound). Twenty-one transects were located along three remnant river channels in the Control area, and 86 were located along five remnant channels in the Impact area (Appendix 3-1A). In the Control area, six transects were established in Ice Cream Slough Run (Map Appendix 1A), five in Rattlesnake Hammock Run (Map Appendix 1A) and ten transects in Persimmon Mound Run (Map Appendix 3A). In the Impact area, five transects were located in UBX Run (Map Appendix 5A), 18 in Montsdeoca Run (Map Appendix 8A), 11 in Oxbow 13 Run (Map Appendix 8A), 28 in Micco Bluff Run (Map Appendix 7A), and 24 in MacArthur Run (Map Appendix 7A). Eight transects at the lower end of MacArthur Run will not be affected by restoration until Phase II/III, and another eight transects were destroyed during Phase I construction (Appendix 3-1A). The Impact area also contained 17 transects located on three new channels and five transects located in four connector channels. These same transects were also used for river channel vegetation studies (Bousquin 2005).

Transects were located to include both straight and curved sections of river channel. We classified transects by channel pattern (i.e., straight or curved sections of channel) based primarily on the shape of the channel in plan view (Appendix 3-1A, Map Appendices 1A, 3A, 4A, 5A, 7A, and 8A). We also used the shape of the channel profile (see Digital Appendix on attached CD) especially the position of the thalweg (i.e., the deepest portion of the channel profile). Transect profiles that tended to be symmetrical with a broad, shallow thalweg (Figure 3-2A) were considered to be from straight sections of channel. Profiles with deep thalwegs that were shifted to one side (Figure 3-2B) were considered curved. For all profiles classified as curved, the thalweg was shifted toward the side of the channel, which corresponded to the outside of the meander bend in plan view. Profiles were also plotted for new channels and connector channels, and these profiles indicated the shape of the channel as constructed. Some transects were located in curved and straight sections of all four channel types except connector channels, which lacked curved sections because of their short length and design.

The location of each transect was permanently marked on both banks of the river with galvanized pipe (3.81 cm diameter) and referenced with differential Global Positioning System. To maintain position during sampling, a cable was stretched between the transect markers and pulled tight with a winch. The marker on the left side of the channel facing downstream was designated the "0" position on the transect. Core samples and channel depth measurements were taken at 1.5 m intervals along each transect from the 0 point.

Submerged core samples were taken waterward of each bank with a coring device (Davis and Steinman 1998) made of polyvinyl chloride (PVC). The core consisted of a 1 m long tube of clear PVC (3.81-cm diameter) attached to a check valve. The valve was kept open during the descent of the core but closed when the core was extracted, creating a vacuum that helped retain the sample. Threaded extensions of PVC pipe were attached to the valve to allow sampling at different depths. These extensions were perforated to allow water above the valve to drain. A t-shaped handle was attached to the top of the core to aid penetration and extraction. The coring device was marked at 10-cm intervals to facilitate depth measurements.

To collect a sample, the core was lowered into the water until it made contact with solid substratum, and this depth to substratum was recorded to the nearest cm using a meter stick and markings on the coring device. Then the core was pushed into the substratum to a depth of 1 m or until resistance prevented further penetration. The core was then extracted and characterized in the field.

Each core sample was divided into substrate-overlying deposits and natural channel bed substrate. We assume that the first appreciable layer (>10 cm thickness) composed of ≥50% sand or peat represents the bed of the pre-channelization river channel and that material above this substratum layer represents relatively recent (i.e., post-channelization) deposits on the channel substratum. The substrate layer and the depositional layer overlying it were divided into sublayers based on appearance (e.g., color, texture, consistency). We also assumed that the depth of the channel substratum corresponded to the depth at which the corer encountered resistance (i.e., the depth to substratum).

For each sublayer, we recorded the following physical characteristics: structure, thickness (to the nearest cm) and, composition. Sublayer structure was characterized as uniform, mixed, or laminated. Uniform sublayers contained >95% by volume of one sediment type (e.g., sand). Mixed sublayers contained more than one sediment type, which were not arranged in layers. Laminated sublayers contained narrow (<1 cm thick) bands of secondary sediment types within a primary sediment type. Sediment type of each sublayer was identified based on relative amounts of mineral and organic material (Table 3-2). When a sublayer contained two or three sediment types, the type accounting for the highest proportion of the sample was assigned a 1, the next highest a 2, and the third highest a 3 (e.g., 1 sand, 2 mucky sand). To simplify the presentation of results, these types were converted to the types described in Table 3-2. Color (hue, value, and chroma) was determined for each sublayer by comparison to Munsell's Soil Color Charts. Hue indicates the color of the light, value indicates the amount of light, and chroma indicates the purity of the dominant wavelength.

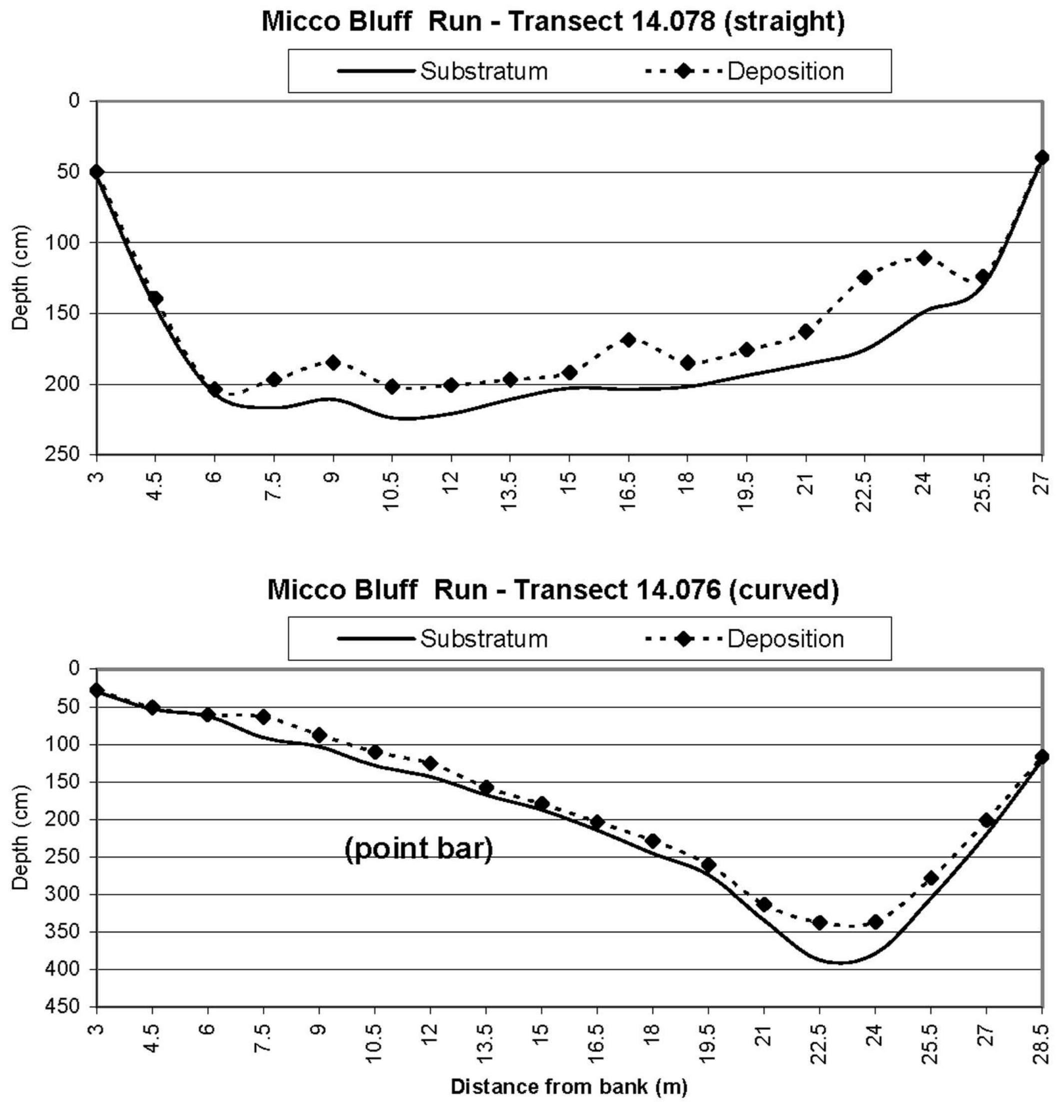


Figure 3-2. Representative channel profiles for transects located in straight (A) and curved (B) sections of channel. These profiles show the surface of the substrate-overlying depositional layer (dashed line) and the surface of the historic channel substrate (solid line). Note that the shallow slope of the bank on the left side of bottom Figure represents a point bar.

Riparian soil core samples were taken at 1.5 m intervals from the water's edge to 4.5 m upland on each bank if the bank was dry and exposed soil was present. Soil cores were taken with a stainless steel coring tube (Oakfield soil sampler, model DB3, diameter = 2.1 cm). Riparian soil cores were collected to a depth of 70-cm, if attainable. For each sublayer in a core, we identified the sediment type (Table 3-2) and the color using Munsell's Soil Color Charts. We documented riparian vegetation at all transects by recording the most common vegetation species occurring within 0.5 m of either side of the riparian core samples.

Chemical Analysis

Additional physical and chemical analyses were undertaken to help ascertain the origin of the marl sediment type. Eighty-two core samples were collected for the major soil types: sand, marl, muck detritus, and muck. An additional ten samples were collected from material excavated during channelization and deposited as spoil mounds along the canal bank. These core samples were collected at sites known to contain specific soil types. Soil samples were analyzed by the University of Florida's Soil and Water Science Department. Particle size distribution and texture were performed on sand and marl samples. Percent organic carbon was measured for all samples, and percent total carbon was measured for samples of marl, muck, and muck detritus. Calcium carbonate was measured for sand, marl, and spoil samples. Mineralogy of spoil and marl samples was examined with x- ray powder diffraction.

Channel Morphology

We used the depth to substratum to develop a cross-sectional profile of the river channel at each transect (see Digital Appendix on attached CD). Depth to substratum was standardized to the operational stage (elevation) of the pool (34 ft NGVD in Pool C, 40 ft NGVD in Pool B, and 46 ft NGVD in Pool A) so that differences in water level among sample dates did not influence measurements of cross-sectional area. We also estimated a depth to organic deposits on the natural river bed substrate by subtracting the thickness of substrate-overlying organic deposits from the corrected depth to substrate and plotting the resulting depth on the channel cross-section. For each profile of a remnant channel, we estimated the cross-sectional area of the channel with and without deposition on the substrate layer using AutoCAD Land Development software.

We characterized the shape of channel cross-sections with the width/depth ratio (w/d) for transects of fixed length described by Olson-Rutz and Marlow (1992) where w is the transect length and d is the average depth. We estimated w/d_{sub} by dividing the transect length (width in Appendix 3-1A) by the average of the corrected depth to substratum measurements for that transect. We also estimated w/d_{dep} using the average of the depth to the substrate-overlying deposits measurements for the baseline period.

Quantification of Substrate-Overlying Deposits

We characterized the substrate-overlying deposits on each transect with three metrics. Mean thickness (cm) of substrate-overlying deposits for each transect, including littoral macrophyte beds was calculated as the average thickness of substrate-overlying deposits for all cores from a transect. We expect values of this metric in restored reaches to decrease. However, substrate-overlying deposits are not expected to disappear entirely because of the influence of littoral macrophyte beds. We defined percent of samples without substrate-overlying deposits as the percent of samples on a transect without such deposits. This metric reflects the areal coverage of substrate-overlying deposits and indicates the availability of habitat suitable for channel dwelling organisms, such as benthic invertebrates, which require or prefer a sand substratum free of substantial organic deposits. We determined the thickness of substrate-overlying deposits at the thalweg (cm) as the thickness of these deposits in the core taken at the deepest point on a transect. The thickness of substrate-overlying deposits at the thalweg should be free of the influence of littoral macrophyte beds and should show a strong initial response to flow.

Reference Conditions

Reference conditions representing point bar formation and substrate-overlying deposits in the pre-channelization river channel were identified. Differences between the reference condition and the baseline condition should indicate the impacts of channelization. Quantified impacts of channelization can be used to guide the development of expectations for the restored system.

The reference condition for the formation of point bars on meander bends in the Kissimmee River was based on aerial photographs of the pre-channelization river (Frei et al. 2005). These photographs were taken during extreme low water levels (38.64 NGVD at Fort Kissimmee) in June 1956 and covered the area of Pool C. Point bars were recognized in these photographs by color, shape, and location on meander bends. Meander bends were distinguished from minor curvature of the channel by an arc angle of 70° (Rosgen 1996).

The reference condition for organic deposition in the river channel was based on data collected by Toth (1991, 1993) in three remnant river channels to which flow had been reestablished during the Kissimmee River Demonstration Project. These data included measurements of the thickness of substrate-overlying organic deposits using core samples collected at 1.5 m intervals on 25 transects across three remnant river channels. We only used data for 24 transects because one transect (Upper Run Transect 1) did not receive appreciable flow during the Demonstration Project. While these transects were sampled for several years after reestablishing flow, we limited our analyses to data from 1988, which was three years after flow was reestablished and the final year of data collection. These samples should resemble the pre-channelization river channel. We used the raw data (L. A. Toth, unpublished data) to calculate mean thickness of substrate-overlying deposits, percent of samples without substrate-overlying deposits, and thickness of substrate-overlying deposits at the thalweg for each transect (Appendix 3-2A). Because we expected much less deposition in the reference condition, we predicted that reference conditions for mean thickness and thalweg thickness would be less than the baseline values and that the reference condition for percent of samples without substrate-overlying deposits would be higher than the baseline condition for remnant channels.

Statistical Analyses

We used ANOVA to test for differences in the average values for mean thickness of substrate-overlying deposits, percent of samples without substrate-overlying deposits, and thickness of substrate-overlying deposits at the thalweg among the four types of channel (Control, Impact, new, and connector). This analysis involved a nested ANOVA design with three factors: AREA (four channel types), RUN(AREA) (runs nested within channel types), and PATTERN (curved or straight channels). Runs were treated as a nested factor within channel type because runs were not independent of channel type (i.e., each run did not occur in each channel type). This design should distinguish between the effects of channel type and river run. Channel pattern was included as an independent factor because it was a potential source of variation. Effect sizes were estimated as variance components using formulas from Quinn and Keough (2002); variance components are approximate because of unequal sample sizes.

Use of ANOVA assumes that the observations are independent and that the groups are normally distributed with homogenous variances (Underwood 1997). Because observations are values for transects rather than individual cores from a transect, we suggest that the observations (i.e., transects) are independent. We used box plots to evaluate the assumptions of normality and homogeneity of variances prior to performing the ANOVA (Quinn and Keough 2002), and data were transformed as necessary to satisfy the assumptions of ANOVA. We also verified that the assumptions were not violated during the ANOVA with a graphical analysis of residuals after the ANOVA (Quinn and Keough 2002). If the effect of AREA (channel type) was significant ($p < 0.01$), we identified channel types that were different by making all pair-wise comparisons with Tukey's HSD, which kept the experiment-wise error rate at 0.05. Statistical analyses were performed in SYSTAT version 7 (SYSTAT Software, Inc., Chicago, Illinois).

We used a statistical power analysis to estimate the magnitude of change that could be determined for mean thickness of substrate-overlying deposits, percent of samples without substrate-overlying deposits, and thickness of substrate-overlying deposits at the thalweg using the method for a t-test described in Zar (1984). For this analysis, we held α, the probability of making a type I error (rejecting the null hypothesis when it is true), at 0.05, and allowed β, the probability of making a type II error (accepting the null hypothesis when it is false), to be 0.25, 0.1, and 0.05. Also, we assumed that the variance for the Impact area for the baseline period would be representative of the pooled variance (S^2_p). Finally, we solved for the minimum detectable difference (δ) at sample sizes (n) ranging from 1 to 50 with the following equation:

$$\delta = \sqrt{(2S^2_p/n)} * (t_{\alpha(1),v} + t_{\beta(1),v})$$

where $t_{\alpha(1),v}$ was the critical value for a one-tailed t-test with α set at 0.05 and v degrees of freedom, and $t_{\beta(1),v}$ was the critical value for a one-tailed t-test substituting β for α with v degrees of freedom.

RESULTS

Point Bar Formation

Aerial photographs from 1994 showed that remnant channels in the Impact area contained 82 meanders and that those in the Control area contained 53 meanders. None of these meanders exhibited signs of active point bar formation (i.e., exposed sand). All relic point bars were colonized by plants, and most were densely vegetated.

Table 3-2. Sediment types and composition from river bottom samples.

Detritus[1]:	Organic debris composed entirely of recognizable plant material such as leaves and stems.
Marl:	Fine silty, clay-like deposit.
Muck:	Fine organic material irrespective of mineral matter content.
Muck Detritus:	Mix muck and detritus. Detritus is 10-90% of sample volume.
Mucky Peat[1]:	Muck mixed in a layer of peat. Peat is 50-90% of sample volume.
Mucky Sand:	Muck mixed in a layer of sand. Sand is 51-90% of sample volume.
Peat:	Consolidated coarse organic sediments.
Sand:	Granular, inorganic sediments.
Sandy muck:	Sand mixed in a layer of muck. Muck is 51-90% of sample volume.
Sandy Peat[1]:	Sand mixed in a layer of peat. Peat is 50-90% of sample volume.
Sandy Marl[2]:	Sand mixed in a layer of marl. Marl is 50-90% of sample volume.
Marly Sand[2]:	Marl mixed in a layer if sand, sand is 50-90% of sample volume.

[1]Class used in the original classification but not observed in core samples and dropped from the revised classification.
[2]Class was added in the revised composition classes.

Channel Characteristics

Cross-sectional profiles of transects in remnant channels were well-defined (see Digital Appendix on attached CD), and we concluded that profiles of the substrate layer retained the shape of the channel prior to channelization.

Remnant channels in the Impact area ranged in width from 12 to 62 m (Appendix 3-1A) with an average of 35 m (Table 3-3), while those in the Control area ranged from 31 to 47 m with an average width of 38 m. New channels tended to be wider than remnant channels, with an average width of 45 m, but the range of width (38–59 m) overlapped the upper range for remnant channels. Connector channels were widest, with a range of 54 to 88 m and an average of 70 m. The greater width of new and connector channels reflects design criteria for the slope of the sides, especially in connector channels.

Width/depth ratios using depth to substrate (w/d_{sub}) ranged from 10 to 46 for remnant channels in the Control and Impacts areas and in the new channels. Mean values for these three channel types were similar, ranging from 22 to 25 (Table 3-3). The similarity of new channels to remnants of the natural channel reflects their design. Connector runs were designed to be wider, which resulted in larger values for w/d_{sub}. Values of w/d_{sub} for connector runs ranged from 22 to 70 and averaged 47 which was almost twice the average for the remnant channels. Because the substrate-overlying deposits were thin relative to channel width, the presence of deposition caused w/d_{dep} to be slightly larger than w/d_{sub} (Table 3-3). The difference between w/d_{dep} and w/d_{sub} were greater in the remnant channels, where there was more deposition, than in the new and connector channels. Width/depth ratios were slightly larger in straight channel sections than curved ones (Figure 3-3), reflecting a tendency toward greater mean depths in curved sections.

Most remnant channels in the Impact area were larger in cross-sectional area, based on corrected depth to substratum, than those in the Control area (Figure 3-4). Cross-sectional area averaged 64 m^2 in the Control area and 35 m^2 in the Impact area, which is approximately half of the average for the Control area (Table 3-4). Throughout the Control and Impact areas, curved channels in most runs had slightly larger areas than straight sections, but these differences were usually within 1 SE (Figure 3-4). The presence of deposition reduced cross-sectional area by an average of 14% in the Control area and 8% in the Impact area (Table 3-4). In most Impact area runs, deposition reduced the cross-sectional area of the channel by similar amounts in straight and curved channels (Figure 3-4). In UBX run and all Control area runs, deposition reduced channel cross-sectional area 5–11% more in straight channels than curved ones, and UBX run had the largest difference with 3 transects in straight channels having three-times the reduction for 2 transects in curved channels.

Table 3-3. Mean (SE) values for channel width (m), depth to substratum (Z_{sub}, cm), width/depth ratio using the depth to substrate (w/d_{sub}), depth to substrate-overlying deposit (Z_{dep}, cm), and width/depth ratio using the depth to the substrate-overlying depositional layer (w/d_{dep}).

Area	Run	n	Width	Z_{sub}	w/d_{sub}	Z_{dep}	w/d_{dep}
Control	Ice Cream Slough Run	6	34 (1.01)	149	24 (2.84)	121	31 (4.41)
	Rattlesnake Hammock Run	5	37 (1.39)	144	28 (3.86)	121	33 (4.60)
	Persimmon Mound Run	10	41 (1.46)	176	25 (2.10)	157	28 (2.63)
	Control Total	21	38 (1.05)	160	25 (1.52)	138	30 (2.02)
Impact	UBX Run	5	39 (0.64)	186	22 (2.95)	174	24 (3.90)
	Montsdeoca Run	18	30 (1.58)	122	27 (2.43)	112	30 (2.82)
	Oxbow13 Run	11	40 (2.34)	156	26 (1.30)	131	31 (1.90)
	Micco Bluff Run	28	35 (1.64)	172	22 (1.28)	158	23 (1.42)
	MacArthur Run	24	36 (1.07)	208	19 (1.52)	197	20 (1.67)
	Impact Total	86	35 (0.82)	170	22 (0.86)	157	25 (1.02)
New	Fulford Run	3	49 (3.23)	120	42 (4.26)	120	42 (4.29)
	Strayer Run	6	40 (1.16)	232	17 (0.33)	225	18 (0.35)
	Oxbow13 (recarved)	3	50 (4.68)	179	32 (6.30)	178	32 (6.50)
	Loftin Run	5	45 (1.12)	227	20 (1.37)	225	20 (1.37)
	New Total	17	45 (1.37)	202	25 (2.62)	198	25 (2.60)
Connector	Montsdeoca-Fulford	1	88	125	70	124	71
	Strayer-Fulford	1	54	242	22	241	22
	Oxbow13-Micco	1	69	117	59	112	61
	Loftin-Micco	2	71 (17.00)	164	43 (7.31)	158	44 (7.88)
	Connector Total	5	70 (7.61)	163	47 (8.45)	159	49 (8.69)

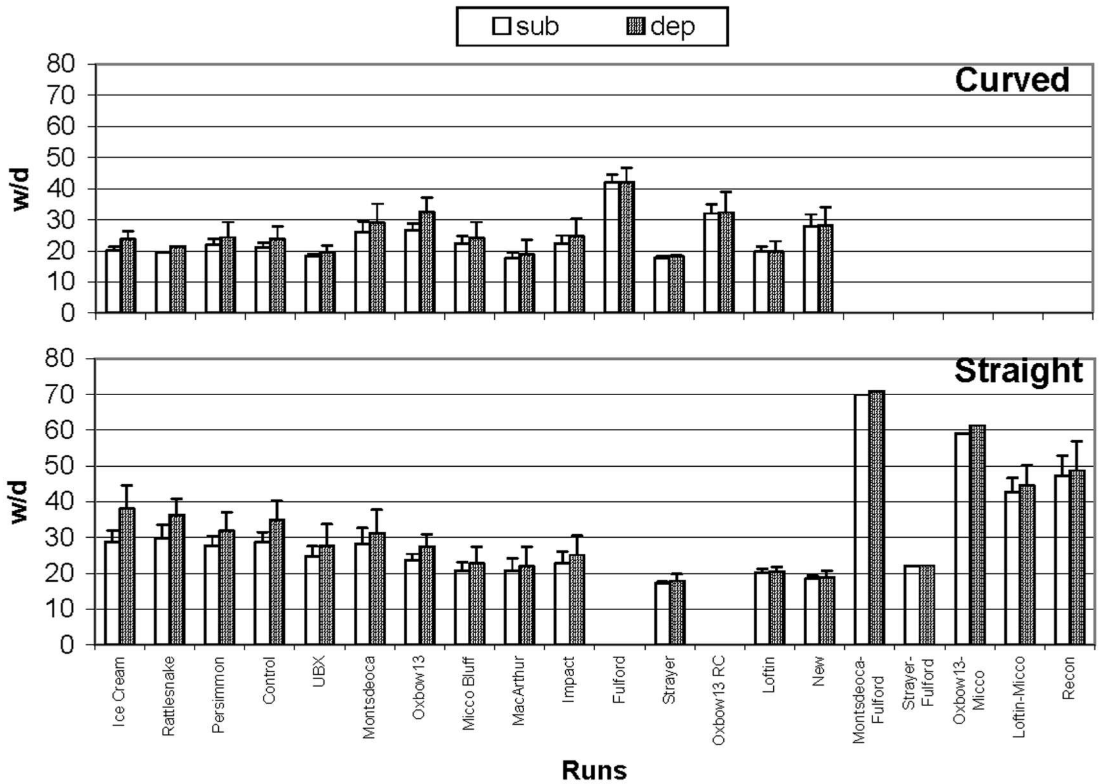


Figure 3-3. Mean width–depth ratio (+1SE) by river runs for transects in curved and straight sections of channel. Width–depth ratios were calculated using the depth to the natural sand substratum (sub) and to the deposits overlying the substratum (dep). River runs are arranged from upstream to downstream within the groupings of Control, Impact, and Recarved.

Sediment Composition

Channel sediments were characterized by collecting 7-56 samples per transect for a total of >3,000 samples. We defined 12 sediment types, but three (detritus, sandy peat, and mucky peat) were not observed in core samples (Table 3-2). The remaining nine sediment types occurred in the natural channel bed substratum, and all but two types (marly sand and peat) occurred in the substrate-overlying deposits (Figure 3-5). Almost 90% of the substrate-overlying sublayers were characterized as muck (herein defined as fine particles from highly decomposed plant fragments, irrespective of mineral matter content) or muck detritus (fine muck and plant fragments). Muck as used in this study included both material commonly recognized as muck and more flocculent material. Another 11% were marl (calcite mud). The substrate was more diverse with mucky sand and sand accounting for 76% of the sublayers.

The interpretation of the composition of these sediment types is supported by physical and chemical measurements. Munsell soil colors reflect the relative amounts of organic and inorganic material in sediment layers. Muck and muck detritus, which were mostly organic, typically had dark brown colors (e.g., 10yr 3/1 and 10yr 3/2). Sand was typically white (2.5yr 8/1), and marl was gray (e.g., 10yr 5/1). Intermediate categories reflected the mixing of inorganic and organic material. Mucky sand, which is predominately sand, was gray (e.g., 10yr 6/1), while sandy muck, which is mostly organic, was brown (e.g., 10yr 3/2).

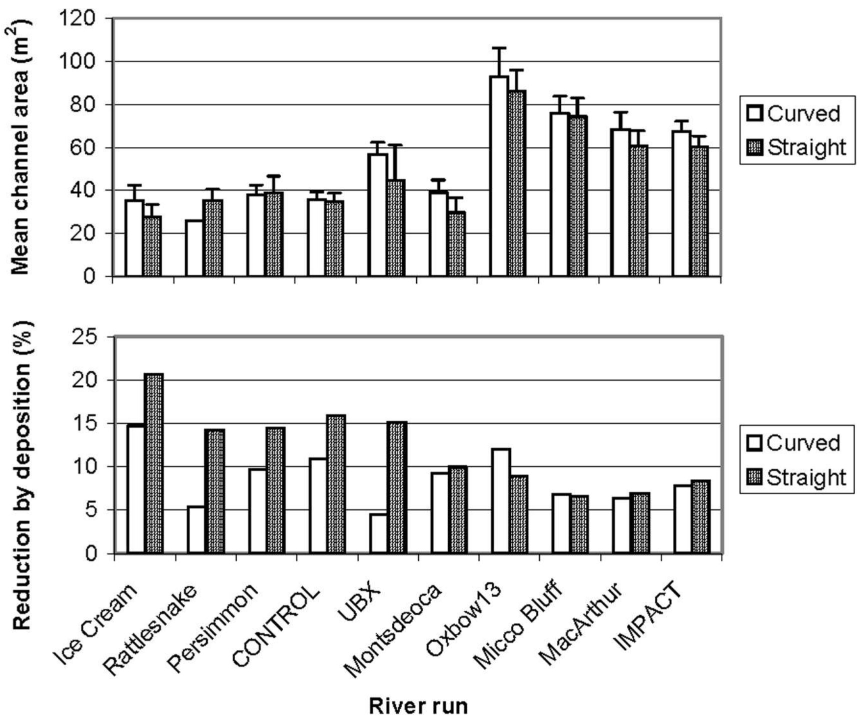


Figure 3-4. Mean (SE) cross-sectional area based on the depth to substratum and percent reduction by deposition on top of the substrate for curved and straight sections of remnant channels in the Control and Impact areas.

Table 3-4. Mean (SE) cross-sectional area (m^2) of channel above the river substrate-overlying deposits and above the river channel bed substratum. Reduction is the percent reduction of the channel area by the substrate-overlying depositional layer.

Area	Run	Deposition on top of substrate	Substrate	Reduction
Control	Ice Cream Slough Run	26 (3.72)	31 (4.39)	18 (2.06)
	Rattlesnake Hammock Run	29 (4.57)	33 (4.55)	12 (3.20)
	Persimmon Mound Run	34 (4.22)	38 (4.36)	12 (1.63)
	Control Total	31 (2.52)	35 (2.62)	14 (1.29)
Impact	UBX Run	47 (10.05)	49 (9.58)	11 (6.78)
	Montsdeoca Run	32 (4.33)	35 (4.43)	10 (1.45)
	Oxbow13 Run	81 (8.09)	90 (8.89)	11 (1.40)
	Micco Bluff Run	70 (5.67)	75 (5.79)	7 (0.62)
	MacArthur Run	61 (5.32)	65 (5.48)	7 (1.07)
	Impact Total	60 (3.26)	64 (3.42)	8 (0.64)

Sand and marl samples contained particles with different size distributions. Sand samples were composed of sand-sized particles (0.05–2.0 mm), while marl samples contained nearly equal quantities of clay and silt-sized particles (<0.05 mm) and averaged <10% sand (Figure 3-6A). Sand samples contained mostly fine-sized sand particles with some medium-sized particles (Figure 3-6B), while the sand fraction of marl was more evenly divided between very fine, fine, and medium-sized sand particles. The total carbon content (as a percentage) of marl was less than half that of muck detritus but only slightly less than that of muck (Table 3-5). Approximately half the carbon in marl was calcium carbonate, and the other half was organic. The percentage of calcium carbonate in marl was more than twice that in spoil, which was >10 times the content of sand (Table 3-5). X-ray diffraction showed that all marl and spoil samples contained small quantities of quartz and varying amounts of calcite, but marl contained more calcite than spoil did. Marl samples also contained larger amounts of the carbonates aragonite and dolomite and the clays smectite and kaolin.

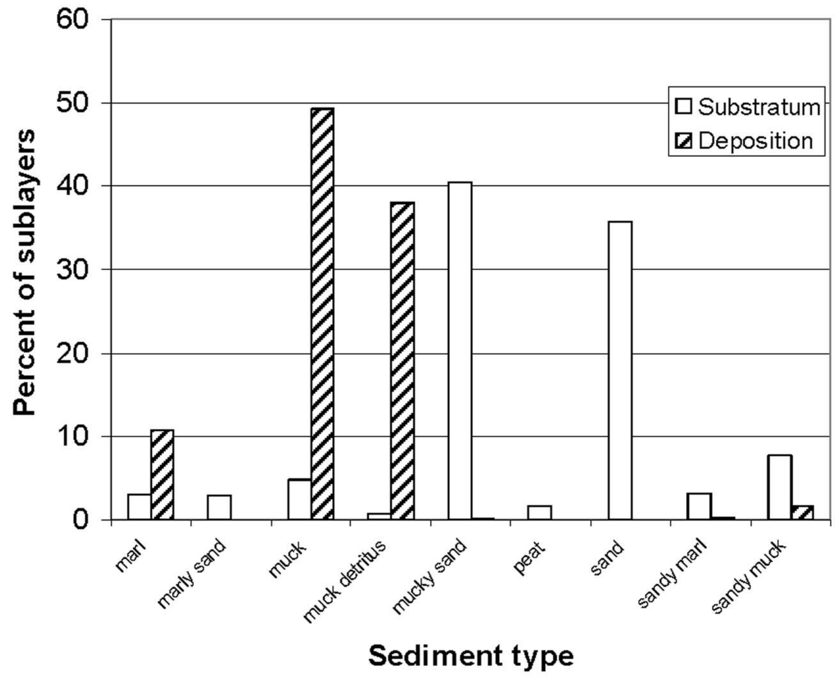
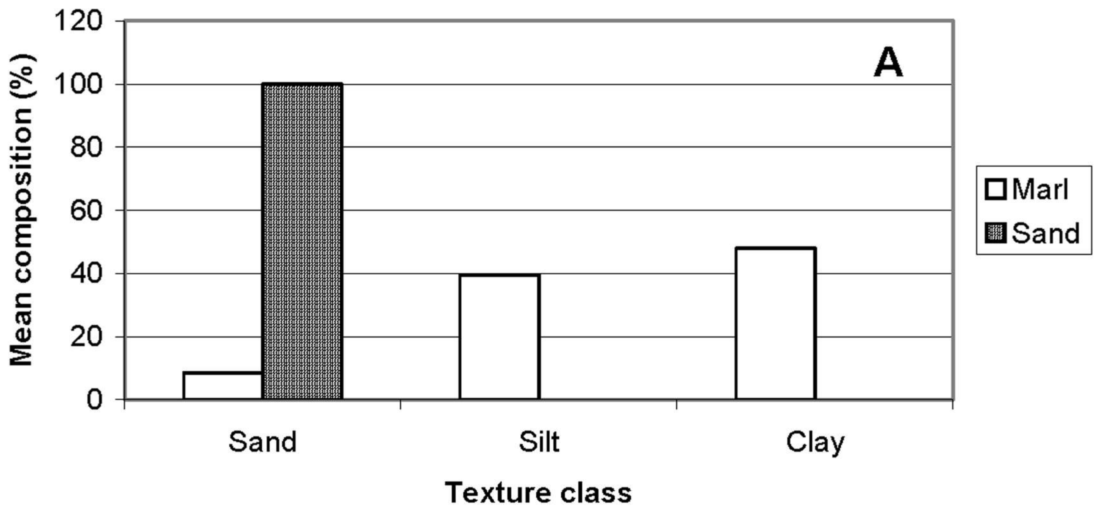


Figure 3-5. Percent of identified sublayers belonging to different sediment types. Sediment types are defined in Table 3-2. Sediment types were identified for 4487 sublayers in the above-substrate depositional layer and for 3351 sublayers in substratum layer.

Substrate-Overlying Deposits

Nearly all core samples from remnant river channels and most of those from new and connector channels contained substrate-overlying deposits. Substrate-overlying deposits were present in 99% of the samples collected in remnant river channels of the Control area and in 97% of those from the Impact area. In new river channels, 80% of the core samples contained a depositional layer above the substrate unit, and in connector runs, 77% did. While most core samples from new and connector river channels contained a depositional layer above the substrate unit, these layers were thinner than those found in remnant channels.

Mean thickness of substrate-overlying deposits was the only one of three metrics used to quantify organic deposits in the river channel that was calculated as an average for each transect. We evaluated how representative the average value was for each transect by calculating precision of the mean (standard error expressed as a percentage of the mean), so that higher values indicate less precise estimates. In remnant channels, precision was frequently less than 20%, which suggests that the mean layer thickness was a good estimate of the mean for the transect. Only 11 transects had precision >30%; none of these transects were in the Control area, and only five were in the Impact area (fewer than 4% of the Impact area transects) (Figure 3-7). Mean thickness tended to be less precise in new channels, where five transects (29% of the new channel transects) had precision >30%, and in the connector channels, where two transects (40% of the connector channel transects) had precision >30%.



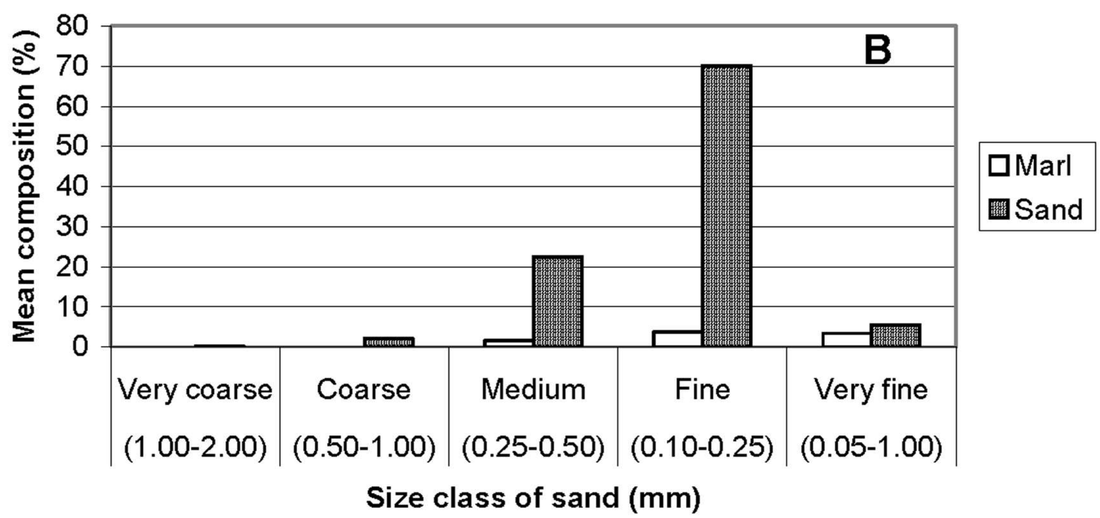


Figure 3-6. Mean composition of different texture classes (A) and different size classes of sand for ten samples of marl sublayers and 21 samples of sand sublayers.

Table 3-5. Mean (SE) percentage of total carbon, calcium carbonate, and organic carbon in different sediment classes.

	Muck/Detritus	Marl	Muck	Sand	Spoil
N	21	10	20	4	10
Total carbon	20.5 (1.5)	6.6 (0.2)	8.5 (0.7)	-------------	-----------
$CaCO_3$	---------------	23* (2.1)	--------------	0.6 (0.2)	10(2.4)
Organic carbon	---------------	3.8 (0.3)	--------------	-------------	-----------

* $CaCO_3$ is 12% carbon, so 3% of C in marl is in the form of $CaCO_3$.

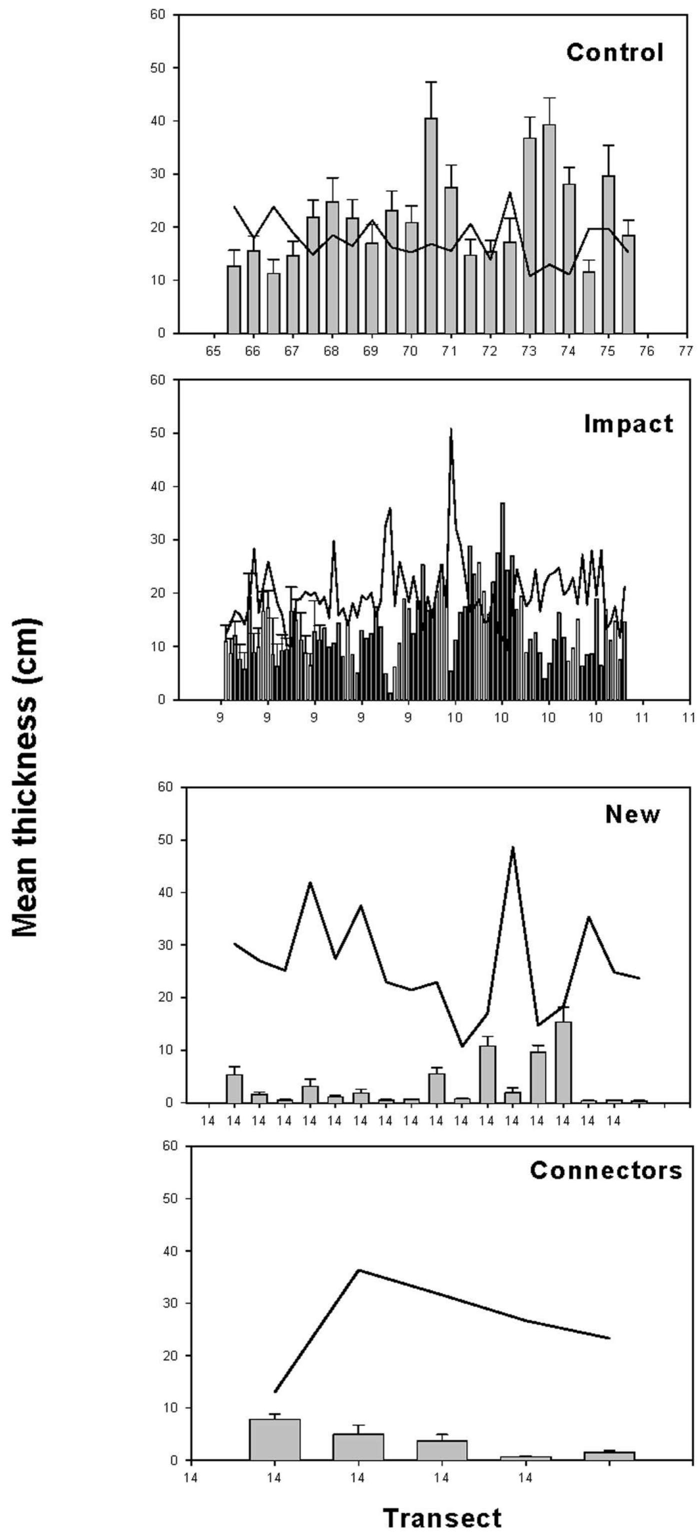


Figure 3-7. Mean thickness (SE) of substrate-overlying deposits (bars) and sampling precision (line) for each transect in the control, impact, new and connector channel types. Sampling precision was the standard error divided by the mean.

Mean thickness of the substrate-overlying deposits averaged 22 cm for 21 transects in the Control area and 14 cm for 86 transects in the Impact area (Table 3-6). Mean thickness of substrate-overlying deposits averaged only 4 cm in both new and connector channels. Box plots for mean thickness of substrate-overlying deposits tended to be symmetrical about the median, which suggested normality, and the overlapping ranges of the whiskers suggested homogeneity of variances among the channel types (Figure 3-8A). Mean thickness of substrate-overlying deposits was significantly different among channel types (AREA) and runs nested within channel type (RUN(AREA)), but it was not different among channel patterns (Table 3-7). The nested model explained 62% of the variance in mean thickness. The variance component for RUN(AREA) was 2.4 times that for AREA (Table 3-7), which shows that RUN(AREA) accounted for a much larger fraction of the variance in mean thickness of substrate-overlying deposits. Tukey's HSD showed that mean thickness of substrate-overlying deposits differed between the Control and Impact areas and that remnant channels differed from both new and connector channels (Figure 3-9).

In remnant river channels, a small percentage of samples on a transect lacked substrate-overlying deposits. In the Control area, percent of samples without substrate-overlying deposits averaged 1%, and in the Impact area it averaged 3% (Table 3-6). Remnant channels averaged a lower percent of samples without substrate-overlying deposits than new and connector channels, which averaged 19% and 25%, respectively. For samples without substrate-overlying deposits, box plots were much less symmetrical than mean thickness of substrate-overlying deposits, which suggests some departure from normality. Remnant channels and new and connector runs had different ranges, suggesting that variances were not homogeneous (Figure 3-8B). We transformed the percent of samples without substrate-overlying deposits by taking the arc sin of the square root of the percent of samples without substrate-overlying deposits after converting percentages to proportions (Quinn and Keough 2002). Using the transformed data, percentage of samples without substrate-overlying deposits was significantly different for AREA and RUN(AREA), but not channel pattern (Table 3-7). The variance component for RUN(AREA) was three times that for AREA, which suggests that differences between runs are greater than the differences between channel types. Tukey's HSD showed that remnant channels in the Control and Impact areas were significantly different (Tukey's HSD, $p < 0.01$) from new channels and connector runs (Figure 3-9).

Thickness of substrate-overlying deposits at the thalweg in remnant channels averaged 38 cm for transects in the Control area and 21 cm in the Impact area (Table 3-6). Thickness of substrate-overlying deposits at the thalweg was much less in new channels, which averaged 6 cm, and in connector runs, which averaged 2 cm. For thickness of substrate-overlying deposits at the thalweg, box plots were symmetrical except in the new river channels, but the range of values generally overlapped, suggesting homogeneity of variances (Figure 3-8C). We transformed substrate-overlying deposits thickness at the thalweg by taking the natural logarithm $(X + 1)$. Transformed values for substrate-overlying deposits thickness at the thalweg were significantly different for AREA and RUN(AREA) but not for channel pattern (Table 3-7). The variance component for RUN(AREA) was two times that for AREA, which suggests that differences among runs were greater than differences among channel types. Tukey's HSD showed that the thickness of substrate-overlying deposits at the thalweg in remnant channels from the Control and Impact area were significantly different (Tukey's HSD, $p < 0.01$) from each other, and from new channels and connector runs (Figure 3-9).

Reference conditions

In pre-channelization aerial photographs, point bars occurred on the insides of 329 of 330 river meanders with an arc angle >70°. The largest point bars occurred on curves downstream of long straight river runs.

Baseline values for the three metrics used to characterize the substrate-overlying deposits were quite different from those of the reference condition (Figure 3-9). The reference condition for mean thickness of substrate-overlying deposits was 5 cm, which was much less than values for remnant river channels and between the values for new and connector channels. Percent of samples without substrate-overlying deposits averaged 55% in the reference condition, which was much higher than the values for remnant river channels, new channels, and connector runs. Thickness of substrate-overlying deposits at the thalweg was 9 cm in the reference condition, which was less than half the value for the Impact area and less than a third of the value for the Control area. Thickness of substrate-overlying deposits at the thalweg at the reference area was about twice the values for the new channels and connector runs.

Table 3-6. Mean (SE) values for mean thickness (cm) of substrate-overlying deposits, percent of samples without substrate-overlying deposits (%), and thickness of substrate-overlying deposits at the thalweg thickness (cm).

Area	Run	n	Thickness	Percent	Thalweg
Control	Ice Cream Slough Run	6	27 (4.34)	0 (0)	44 (9.02)
	Rattlesnake Hammock Run	5	23 (4.94)	1 (0.95)	33 (13.74)
	Persimmon Mound Run	10	18 (1.48)	1 (0.71)	38 (6.98)
	Control Total	21	22 (1.93)	1 (0.40)	38 (5.11)
Impact	UBX Run	5	13 (1.65)	0 (0)	6 (1.20)
	Montsdeoca Run	18	11 (1.03)	1 (1.19)	13 (2.92)
	Oxbow13 Run	11	24 (1.77)	2 (0.99)	47 (7.82)
	Micco Bluff Run	28	14 (1.15)	7 (1.59)	25 (3.18)
	MacArthur Run	24	11 (0.74)	0 (0.25)	15 (3.49)
	Impact Total	86	14 (0.71)	3 (0.65)	21 (2.17)
New	Fulford Run	3	0 (0.04)	56 (4.47)	0 (0.00)
	Strayer Run	6	7 (2.29)	4 (1.39)	14 (4.96)
	Oxbow13 (recarved)	3	1 (0.43)	28 (11.77)	0 (0.17)
	Loftin Run	5	2 (0.86)	10 (4.80)	4 (1.76)
	New total	17	4 (1.07)	19(5.32)	6 (2.26)
Connector	Montsdeoca-Fulford Connector	1	2	28	0
	Strayer-Fulford Connector	1	1	41	4
	Oxbow13-Micco Connector	1	4	33	1
	Loftin-Micco Connector	2	6 (1.46)	11 (0.36)	2 (0.50)
	Connector total	5	4 (1.27)	25 (6.01)	2 (0.68)

Table 3-7. Coefficient of determination (R^2), F statistics, variance components (S^2), and mean square error for nested ANOVAs on mean thickness of substrate-overlying deposits (Thickness), percent of samples without substrate-overlying deposits (Percent), and thickness of substrate-overlying deposits at the thalweg (Thalweg). Nested ANOVA included three factors: channel type (AREA), runs nested within area RUN(AREA), and channel pattern (PATTERN). Values in parentheses are the percentage of the total variance explained by the variance component.

		AREA		RUN(AREA)		PATTERN		
Metric	R^2	F[1]	S^2	F[1]	S^2	F[1]	S^2	MSE
Thickness	0.62	5.7*	27.7 (22)	7.5**	65.8 (53)	0.3	0 (0)	30.3 (24)
ArcPercent	0.65	5.3*	0.02 (22)	9.5**	0.05 (58)	0.8	0 (0)	0.02 (20)
Lthalweg	0.54	23.4*	0.5 (20)	5.4**	1.1 (46)	1.6	0.08 (3)	0.8 (31)
df		3		9		1		115

[1] Significant differences are indicated by * ($p < 0.05$) and ** ($p < 0.01$).

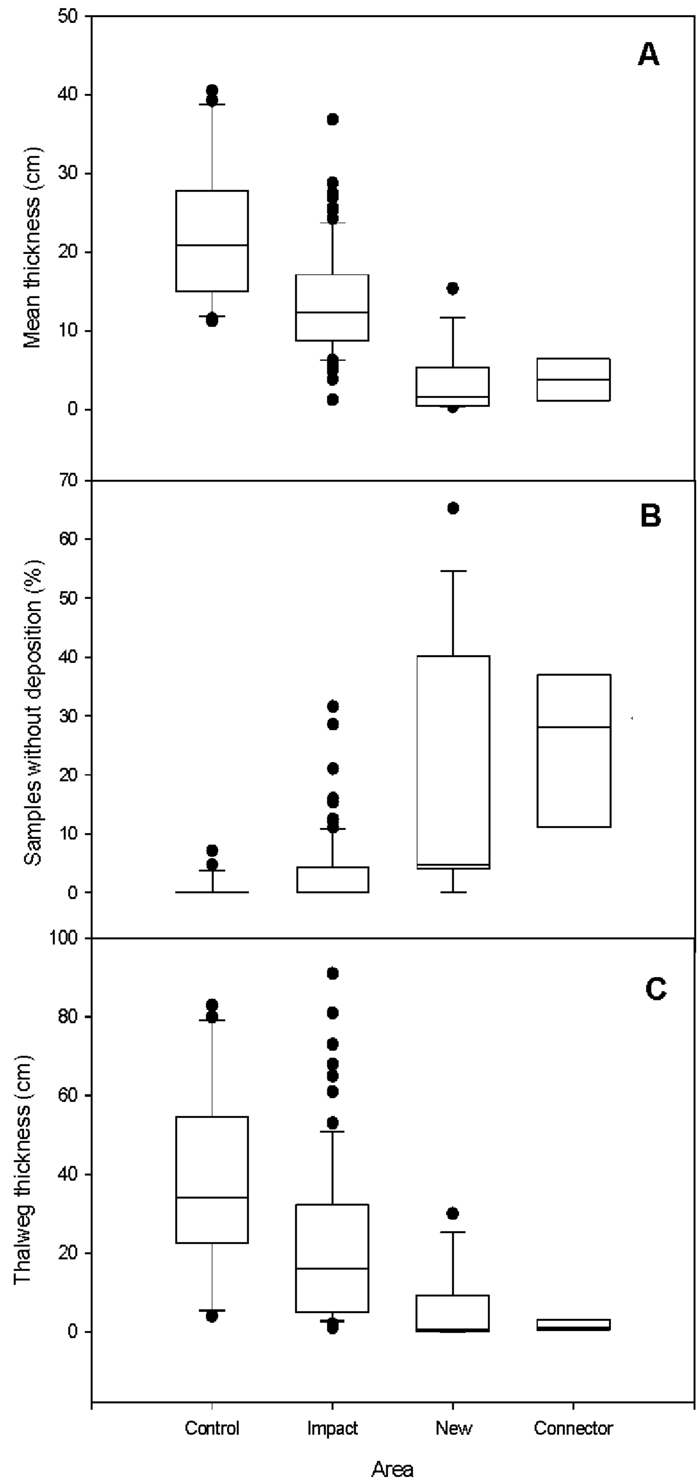


Figure 3-8. Box plots of mean thickness (A), percent of samples without deposition (B) and thalweg thickness (C) for each channel type. Horizontal lines represent the median values, the ends of the boxes represent the 25th and 75th percentiles, and the error bars indicate the 10th and 90th percentiles. Circles indicate values outside the 10th and 90th percentiles.

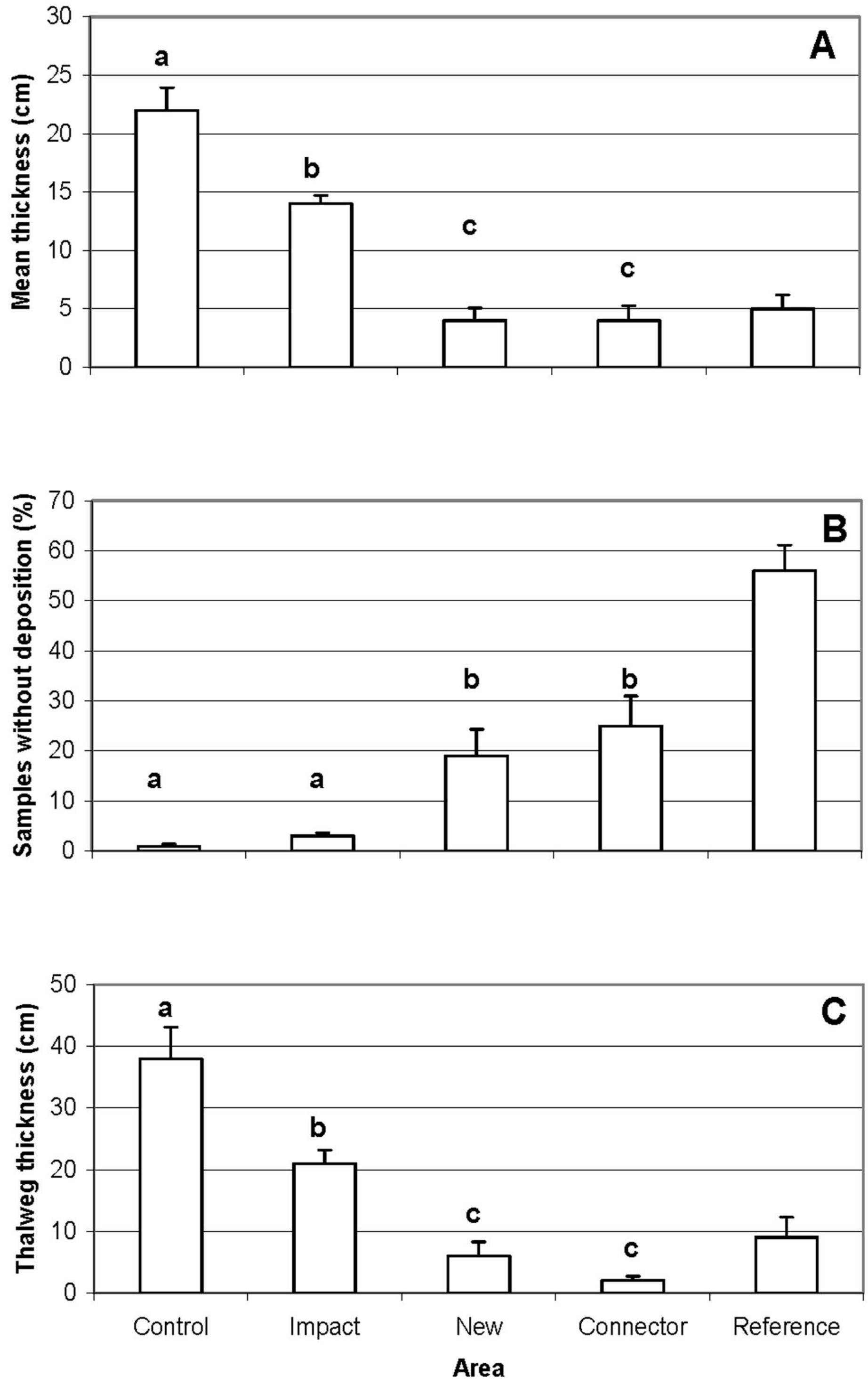


Figure 3-9. Mean (+SE) values for mean thickness of substrate-overlying layer on a transect (A), percentage of samples without a substrate-overlying layer on a transect (B), and thickness of the substrate-overlying layer at the thalweg of each transect (C) for control, impact, new and connector channel types. Reference is for the Demonstration Project data. Different letters identify which baseline channel types were significantly different (Tukey's HSD, $p<0.01$).

Prospective Power Analysis

Estimates of the minimum detectable difference showed similar relationships to sample size and β for mean thickness of substrate-overlying deposits, percent of samples without substrate-overlying deposits, and thickness of substrate-overlying deposits at the thalweg (Figure 3-10). For all three metrics, the minimum detectable difference decreased with increasing sample size, especially for sample sizes <10. Minimum detectable difference increased with increasing values of β until $n = 10$, and then β had little effect on the minimum detectable difference.

We compared the minimum detectable difference from the power analysis with the changes that would need to occur in order for the baseline values in the Impact area to attain the reference values. For this comparison, we used the minimum detectable difference for each metric when $n = 80$, $\alpha = 0.05$, and $\beta = 0.1$. To attain the reference value of 5 cm, the baseline value for mean thickness of substrate-overlying deposits of 14 cm would have to decrease by 9 cm, which is larger than the minimum detectable difference of 3 cm. To attain the reference value of 56%, the baseline value for percent of samples without substrate-overlying deposits of 3% would have to increase by 53%, which is larger than the minimum detectable difference of 6%. To attain the reference value of 9 cm, the baseline value for thickness of substrate-overlying deposits at the thalweg of 21 cm would have to decrease by 12 cm, which is larger than the minimum detectable difference of 9 cm. For all three metrics, the minimum detectable difference is smaller than the expected change based on the difference between the baseline value for the Impact area and the reference value, so the current sampling design should be adequate to detect these changes in the Impact area if they occur.

Riparian Characteristics

Riparian soil core samples from Control and Impact areas were similar in composition (Figure 3-11). In the Control area, sand-muck combinations comprised 31.7% of the cores, followed by muck (28%), sand (21%), mucky sand (11.7%), and sandy muck (10.8%). In the Impact area, sand-muck combinations comprised 22.94% of the layers, followed by sand (25%), muck (17.42%), mucky sand (15.77%) sandy muck (11.90%) and peat (7.10%).

Color analyses of riparian soils in both Control and Impact areas indicated large amounts of organic material. A black color (10yr 2/1) was the most prevalent and accounted for 17% of the layers in the Control area and 7% in the Impact area. The next largest percentage was dark gray (10yr 3/1), which was present in 13% of the layers in the Control area and 11% in the Impact area (Table 3-8).

Approximately 100 plant species were identified at riparian soil core sample locations. Only 16 species were present in more than 5% of sample locations in each remnant river run (Table 3-9), and nine of those were present in both the Control and Impact area. *Vitus spp* (wild grape) was found in the highest percentage in the Impact area followed by *Myrica cerifera* (Wax myrtle) and *Paspalum notatum* (Bahia grass). In the Control area, *Panicum hemitomon* (maidencane) was the most common species found, but *Baccharis halimifolia* (salt bush) and *Paspalum notatum* were common in Rattlesnake Hammock, and *Urena lobata* (Caesar weed) was abundant in Ice Cream Slough.

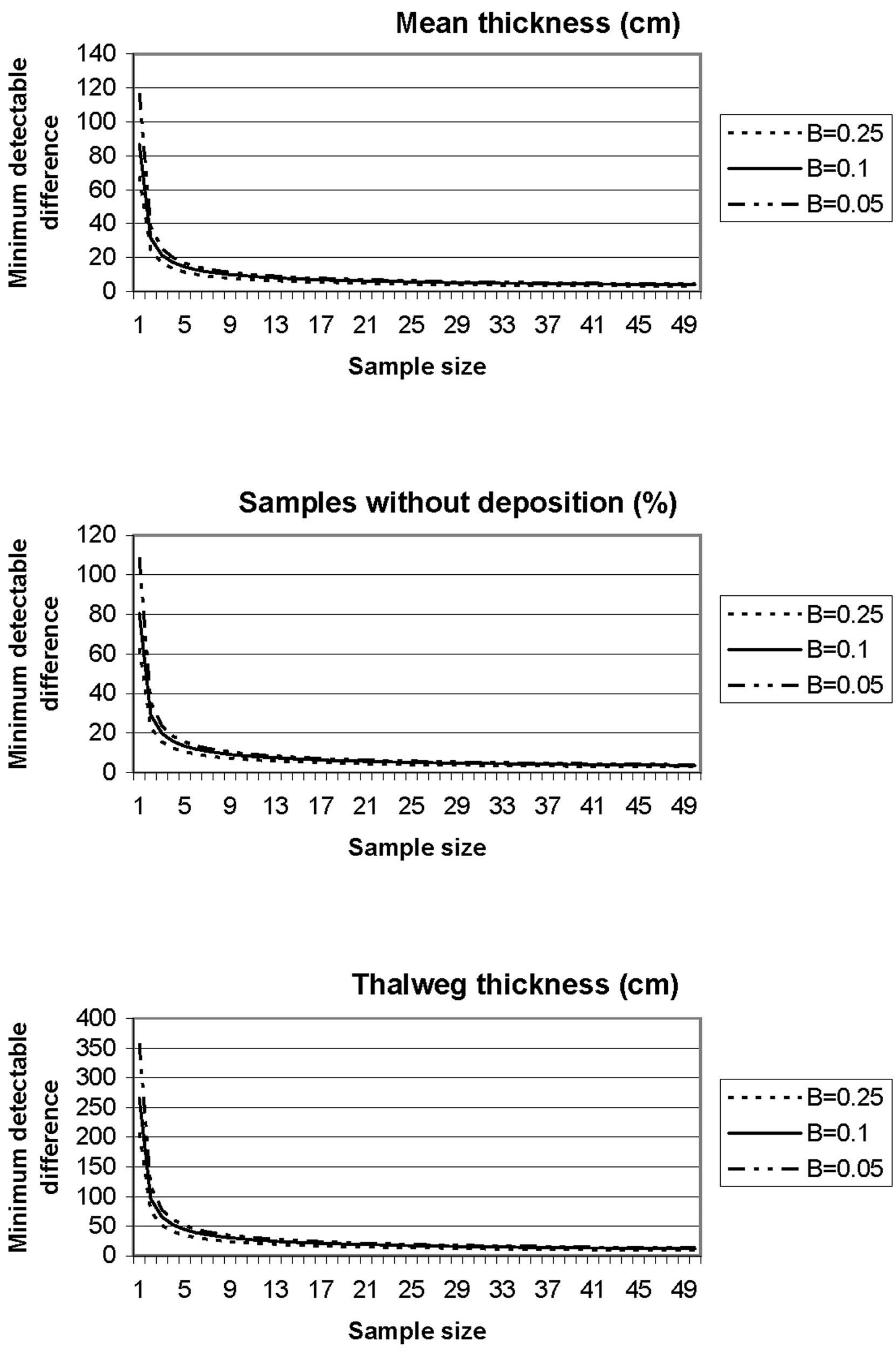


Figure 3-10. Minimum detectable difference versus sample size from a power analysis on mean thickness of the substrate-overlying layer (A), percent of samples without substrate overlying deposits (B), and thalweg substrate-overlying layer thickness (C), when $\alpha = 0.05$ and for three levels of β.

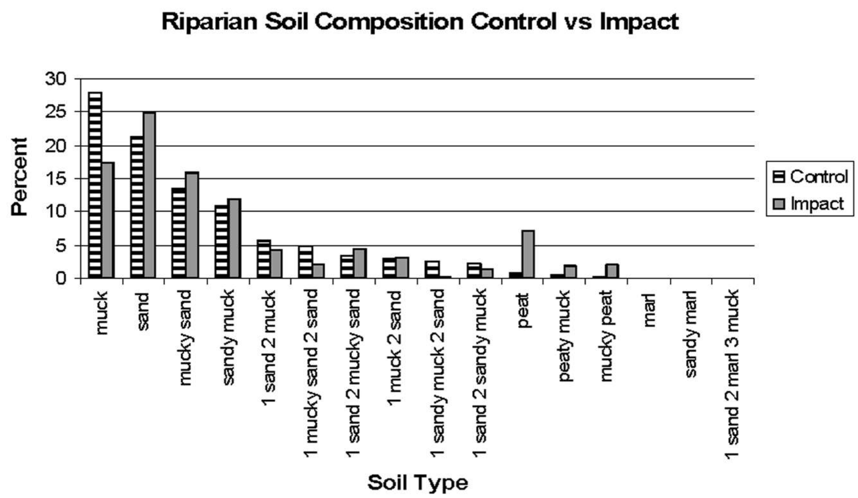


Figure 3-11. Riparian soil composition, Control vs. Impact. Percentages are based on total number of core samples taken from the Control and Impact area.

Table 3-8. Percent of sampled layers belonging to Munsell soil colors (hue, value, chroma) for riparian soils in all core samples in the Control and Impact areas. Others category includes all soil colors represented by fewer than 1% of all samples.

Color	Munsell Color	Percent	
		Control	Impact
Black	GL1 2.5/N	4	
Dark Brown	7.5yr 3/3		2
Dark Brown	7.5yr 3/2		3
Very Dark Gray	7.5yr 3/1		4
Black	7.5yr 2.5/1		2
Reddish Gray	2.5y 5/1	1	
Dark Reddish Gray	2.5y 3/1		1
Reddish Black	2.5y 2.5/1		1
White/Very Dark Gray*	10yr 8/1, 10yr 3/1*	3	
White	10yr 8/1	1	4
Light Gray/Very Dark Gray*	10yr 7/2, 10yr 3/1*		2
Light Gray	10yr 7/2	3	4
Light Gray/Dark Gray*	10yr 7/1,10yr 4/1*	1	
Light Gray/Very Dark Gray*	10yr 7/1,10yr 3/1*	4	2
Light gray	10yr 7/1	2	2
Gray	10yr 6/1		3
Gray	10yr 5/1		2
Dark Gray	10yr 4/1	2	5
Very Dark Grayish Brown	10yr 3/2	5	2
Very Dark Brown/Gray*	10yr 3/1,10yr 6/1*	2	1
Very Dark Brown/ Grayish Brown*	10yr 3/1, 10yr 5/2*	1	
Very Dark Gray	10yr 3/1	13	11
Very Dark Brown	10yr 2/2		2
Black/Light Brownish Gray*	10yr 2/1,10yr 6/2*	1	
Black	10yr 2/1	17	7
	Others	38	40

* Layered soil containing two colors.

Table 3-9. Percentage of samples with different riparian vegetation species present for each run of the Impact and Control areas. Only vegetation species present in more than 5% of the samples from any river run are shown.

	Impact			Control			
Species	MacArthur	Micco	Oxbow 13	Montsdeoca	Persimmon	Rattlesnake	Ice Cream
Acer rubrum	2.93	6.5	0	0			
Ambrosia artemisifolia					1.58	6.67	5.21
Baccharis halimifolia					4.42	14.67	1.04
Eupatorium capillifolium	0.49	1.42	5.98	0.24	2.21	6	4.17
Hydrocotyle umbellata					1.26	8	3.13
Ludwigia peruviana					6.62	0	6.77
Myrica cerifera	21.52	13.01	5.98	8.19	9.46	3.33	2.6
Panicum hemitomon	4.16	4.27	5.13	2.89	8.52	12.67	17.19
Paspalum notatum	0	11.99	10.26	8.91	4.1	12	3.13
Pteridophyta spp.	6.6	5.08	0	0.24			
Rubus cuneifolius	2.69	0.61	7.69	4.82	9.46	6.67	0
Sambucus canadensis	1.22	2.64	6.84	5.06	4.42	1.33	8.33
Schinus terebinthifolious	0.49	6.91	5.13	10.6	0	1.33	6.25
Sida acuta					0.63	4.67	8.33
Urena lobata	5.38	6.5	7.69	5.54	3.47	0.67	10.94
Vitus spp.	17.11	6.71	23.08	10.12	5.68	0	2.6

DISCUSSION

Impacts of Channelization

Our study quantified the impacts of channelization on the geomorphology of the Kissimmee River. We attribute these impacts primarily to the near elimination of flow through channel remnants by diverting the flow of water from the river channel to the C-38 canal (Figure 3-1). While some flow occurs in remnant channels because of inputs from overland flow or small tributaries (e.g., Oak Creek in Micco Bluff Run, Istokpoga Canal in MacArthur Run), these flow events were probably of low volume and short duration. The resulting stagnation in remnant river channels suspended the natural channel-shaping geomorphic processes of sediment transport and deposition. Thus, it is not surprising that we found little evidence that channelization altered the overall shape of cross-sections of remnants of the natural river channel. While we lack pre-channelization data for comparison, we believe that morphometry based on the channel bed substratum (e.g., Z_{sub}, w/d_{sub}, shape of cross-sectional profile, and cross-sectional area to substratum) reflects the natural channel condition.

Channelization did affect two major aspects of river channel geomorphology: point bar formation and organic deposition within the river channel. Point bars regularly form along the inside of meander bends (Leopold 1994) and were present on all meander bends in pre-channelization aerial photographs of the Kissimmee River (D. Frei, personal observation). We did not observe point bars on any meander bends in post-channelization aerial photographs although remnant bars are evident in cross-sections of curved channels (e.g., Figure 3-2). We suggest that elimination of flow has halted sand transport and deposition required to extend point bars and that lack of flow and stabilized water levels have allowed vegetation to colonize extant point bars. In post-channelization aerial photographs, all meander bends were overgrown with vegetation. Thus, stabilized water levels and vegetation hid remnant point bars.

River channel substratum was altered by deposition of organic matter on the channel bottom. Our ability to interpret the presence of these deposits as an impact of channelization is based on three assumptions: (1) that we could distinguish between organic deposits and channel bed substratum, (2) that the substratum layer represents the pre-channelization channel bottom, and (3) that the substrate-overlying deposits represent relatively recent (post-channelization) deposition. These assumptions seem reasonable based on several pieces of evidence. First, composition of the substrate-overlying deposits differed from that of the substratum based on color and appearance so that the two layers could be distinguished easily. These differences were supported by color analysis of all sublayers and by chemical analyses of select sediment classes, especially marl and sand. Second, the substratum layer was composed primarily of sand

and mucky sand, and we know that sand of the same medium- and fine-sized particles was the primary constituent of river channel sediments prior to channelization (U. S. House of Representatives 1902, Warne et al. 2000). Also, our cross-sectional plots of the substratum layer appear to capture the shape of the historic channel and exhibit little evidence of sloughing of channel banks, and the shape of the cross-sectional profile for curved channels is consistent with the direction of the curve in plan view. Third, fine organic particles composing much of the substrate-overlying depositional layer are unlikely to have accumulated prior to channelization because flows competent to transport these particles (i.e., bankfull discharge) occurred regularly and for long periods of time (Toth et al. 2002). Fourth, changes in flow, which regulates entrainment and deposition of sediments, provide a mechanism to account for the presence of extensive deposits in the upper layer of the river channel bed post-channelization. Finally, this mechanism is consistent with observations during the Demonstration Project, which reestablished flow to several remnant river channels (Toth 1993).

The primary constituents of substrate-overlying deposits were organic muck and muck detritus. These organic sediments most likely were produced by decomposition of aquatic macrophytes that expanded their coverage of remnant river channels in the absence of flow. These macrophytes included: emergent species such as spatterdock (*Nuphar lutea*) and smartweed (*Polygonum densiflorum*); mat-forming species such as Cuban bulrush (*Scirpus cubensis*); and freely floating aquatic vegetation (FFAV) such as water hyacinth (*Eichhornia crassipes*) and water lettuce (*Pistia stratiotes*). As these plants die and begin to decompose they serve as a source of organic deposition. The high proportion of fine muck sediments relative to the coarser muck detritus in the substrate-overlying depositional layer is consistent with rapid breakdown of macrophytes in warm waters (reviewed in Grimshaw 2002). Freely floating aquatic vegetation was likely an important source of organic deposition because it could cover much of the midchannel area and was treated with herbicides beginning in 1983 (Grimshaw 2002). Herbicides have been used to control the cover of freely floating aquatic vegetation, and an average of 791 ha (range of 347 ha to 1578 ha) was treated between 1983 and 1998 (Grimshaw 2002). Initiation of maintenance control in 1988 significantly reduced the size of the area requiring herbicide treatment each year (Grimshaw 2002) and by limiting the amount of organic production probably reduced the amount of organic deposition each year (e.g., Joyce 1985).

Another component of the substrate-overlying deposits was inorganic marl. Chemical analysis showed that this material had the characteristics of marl (high carbonate content with clay minerals). Presence of dolomite suggests a clastic origin for the carbonate. When marl was present, it usually occurred between the substratum layer and the more recently deposited organic layers (see Digital Appendix on attached CD). Thus, it is possible that marl deposition was associated with construction of the C-38 canal. Core borings near S-65 (well OSF-52), S-65A (well PDF-20), and S-65C (well OKF-42) contained shell beds and other potential sources of carbonate material within the top 9 m (Shaw and Trost 1984), which might have been exposed during excavation of the canal. Dredging material from the canal may have suspended marl in the water and transported it into the remnant channel, or it may have become entrained in runoff from spoil mounds on the canal bank, which carried it to the remnant channel. Physical and chemical analyses show that marl from the substrate-overlying deposits differs from sand in the river channel and is more similar to spoil excavated from the canal. Differences in chemical composition may be related to differential weathering of marl deposited on the bank in exposed spoil piles and that deposited in remnant river channels.

New channels and connector channels were carved during Phase I of the restoration project, and their characteristics reflect design criteria and their relatively young age. These channels contain much thinner substrate-overlying deposits than remnant channels, and these thin deposits probably represent deposition transported from remnant channels when flow was restored.

We characterized the effects of post-channelization deposition on channel morphology (depth, width/depth ratio, cross-sectional area) by using the depth to substratum, which we believe represents the natural channel bottom, and the depth to the substrate-overlying deposits. Deposits tended to have relatively small effects on metrics that describe channel shape because the deposits are thin relative to the width and depth of the channel. We created three metrics to characterize directly the channel substratum, and the utility of these new metrics partially reflects how each is calculated. One is an average (mean thickness of substrate-overlying deposits), one is a percentage (percent of samples without substrate-overlying deposits), and one is a single value from a specific location on each transect (thickness of substrate-overlying deposits at the thalweg). Because it is an average, mean thickness of substrate-overlying deposits has the best sampling characteristics. It provides a precise estimate for a transect (standard error typically <20% of the mean) because of the high sample size of >20 samples for most transects and because it does not appear to violate the assumptions for ANOVA. We did not evaluate precision for percent of samples without substrate-overlying deposits or thickness of substrate-overlying deposits at the thalweg, because these metrics are not calculated as an average. Also, both metrics required transformation to meet the assumptions for ANOVA. All three had reasonable power to detect changes due to restoration based on the differences between the baseline values for the Impact area and the reference values. Thus, all three should be useful for restoration evaluation.

Substrate-overlying deposits represent an accumulation of organic matter over time, and the quantity of deposition at any point in time depends on the balance between inputs (import from upstream, macrophyte death) and losses (downstream export, microbial activity). We did not attempt to quantify the rate of change during the baseline period, and we sampled each transect only once to establish a baseline for evaluating changes during restoration. This approach implicitly assumes that the quantity of deposition remains fairly constant over the approximately three-year period that we sampled, which immediately preceded the restoration project. This assumption is reasonable because the inputs in any given year were likely to be small (e.g., Joyce 1995) compared to the total amount of deposition. Also, the change in loading with maintenance control for FFAV occurred a decade earlier.

We used ANOVA to investigate spatial variability of the three metrics at three scales: among channel types (AREA), among runs nested within channel types (RUN(AREA)), and among curved and straight channels (PATTERN). This statistical model accounted for at least 50% of the variation in three metrics: mean thickness of substrate-overlying deposits, the arc sin of percent of samples without substrate-overlying deposits and the natural logarithm of thickness of substrate-overlying deposits at the thalweg. AREA and RUN(AREA) were significantly different for all three metrics but PATTERN was never significant, and RUN(AREA) always accounted for at least twice as much of the variability as AREA, which suggests that runs are more variable than Impact and Control channel types. Mean values for new and connector channels were never different from each other, but they were different from the remnant channels, which probably reflect the young age of these constructed channels, and the lack of opportunity for macrophyte growth and detritus deposition to occur. Mean values for Control channels differed from those for Impact channels for metrics describing the amount of deposition (mean thickness of substrate-overlying deposits, thickness of substrate-overlying deposits at the thalweg) but not for percent of samples without substrate-overlying deposition. This difference between Control and Impact areas may reflect higher rates of deposition in the Control area, which is consistent with greater vegetation cover in the Control area (Bousquin 2005).

The thickness of substrate-overlying deposits varies in thickness between runs, which may reflect differences in inflows, riparian vegetation, and frequency of aquatic plant (weed) management activity in remnant river channels. Thinner deposits tend to occur in runs that have had at least some flow since channelization (i.e., MacArthur, Micco Bluff, Montsdeoca, and UBX). Also, these runs typically have greater coverage of tall riparian vegetation (shrubs and trees) along both riverbanks than runs with thicker substrate-overlying depositional layers, which are largely flanked by pasture (Table 3-10). Tall riparian trees and shrubs can shade aquatic vegetation and limit its growth. The thickness of substrate-overlying deposits also may be affected by aquatic plant management activities such as the use of herbicides to control vegetation cover, which may limit the amount of deposition.

The accumulation of organic matter in remnant river channels has important consequences for other components of the Kissimmee River ecosystem and thus, for ecological integrity. The near uniform covering of the river channel reduces the substratum diversity available for aquatic invertebrates (Harris et al. 1995) contributing to their low diversity (Koebel et al. 2005). It also reduces the area of appropriate nesting habitat for fish (Trexler 1995). Organic deposition also provides an abundant substrate for microbial respiration, which contributes to depressed concentrations of dissolved oxygen observed in remnant river channels during the baseline period (Colangelo and Jones 2005). We expect oxygen consumption by the substrate-overlying depositional layer to increase with temperature because of increased microbial activity. That increase in microbial activity helps explain the seasonal patterns of dissolved oxygen, although other factors such as seasonal solubility also are important. Oxygen consumption by organic deposits also helps account for the vertical gradient of decreasing oxygen concentration with depth (Perrin et al. 1982, Toth 1993, Colangelo and Jones 2005). Belanger et al. (1994) conservatively estimated oxygen consumption in core samples of benthic sediments for one remnant channel in Pool C (Impact area) and found an average consumption rate of 0.037 g O_2 m^{-2} h^{-1} (range of 0.003–0.094 g O_2 m^{-2} h^{-1}) for three dates in June–August 1994. It is difficult to interpret the significance of this estimate of the oxygen demand by benthic sediments without understanding its spatial and temporal variability or the relative magnitude of other processes (e.g., reaeration) that can influence dissolved oxygen concentrations. Nonetheless, oxygen consumption by organic deposits in the river channel does contribute to low oxygen concentrations in remnant river channels. It therefore indirectly affects redox conditions, which can profoundly influence biogeochemical processes, especially at the sediment water interface. The extent to which this occurs will require further study.

Table 3-10. Inflow and riparian characteristics of remnant river channels in the Control and Impact areas, new channels, and connector runs.

Control Area

Ice Cream Slough: West bank shrubs and pasture, east bank shrubs.

Rattlesnake Hammock: West bank primarily pasture with some shrubs, east bank shrubs. Very low inflows from a culvert that drains Rattlesnake Hammock Marsh.

Persimmon Mound: West bank shrubs, east bank primarily pasture with some shrubs. Some minor tributary inflow.

Impact Area

UBX: Both banks covered primarily with willow and wetland vegetation. Periodically transported low flow (<100 cfs) to a culvert at its south end during some post-channelization years but has not carried flow since 1990.

Montsdeoca: West bank primarily pasture with some shrubs, east bank primarily hardwoods and shrubs. Receives some inflow from drainage canals and a culvert in the S-65B tieback levee.

Oxbow 13: West bank pasture, east bank shrubs. Short run, close to C-38 throughout entire course.

Micco Bluff: Heavily wooded on west bank with mix of shrubs, hardwoods and palmettos. East bank primarily pasture with some hardwoods and shrubs. Receives tributary inflow from Oak Creek and Starvation Slough.

MacArthur: Most heavily wooded run in Pool C. Primarily bounded by shrubs and hardwoods. Receives some inflow from Istokpoga Canal.

Recarved runs

Fulford Run: West bank with Wax myrtle (*Myrica cerifera*) and Brazilian pepper (*Schinus terebinthifolius*).

Strayer Run: Mostly ungrazed pasture with dogfennel (*Eupatorium*).

Obow 13 RC: Mostly grazed Bahia grass (*Paspalum notatum*) pasture on the west bank and bare spoil of backfilled canal on the east.

Loftin Run: Wax myrtle (*Myrica cerifera*). Both sides have Primrose willow (*Ludwigia peruviana*) and some Brazilian pepper (*Schinus terebinthifolius*) and some Coastal plain willow (*Salix caroliniana*).

Connectors

Montsdeoca-Fulford Connector: Initially barren spoil on both sides of the channel.

Strayer-Fulford Connector: Initially barren spoil on both sides of the channel.

Oxbow 13 – Micco Connector: Initially barren spoil on both sides of the channel.

Loftin-Micco Connector: Initially barren spoil on both sides of the channel.

Expectations for the Restored River

Expectations for geomorphology in the restored river should focus on attributes that are likely to respond to restoration and have reference conditions. The restoration project is designed to reconnect remnant river channels and to reestablish a hydrologic regime that mimics pre-channelization conditions, which should initiate natural processes of sediment transport and deposition. These processes should create active point bars on most meanders. Also, flow should reduce the quantity of organic deposits on the channel substratum either by burial or by entrainment and export downstream to the canal. Reestablishing flow should also reduce the input of organic detritus by reducing vegetation cover. An expectation for point bar formation was developed based on aerial photography taken before channelization and states that "Point bars will form on the inside bends of river channel meanders with an arc angle >70°" (Frei et al. 2005). In the absence of historical data for the Kissimmee River, expectations for organic deposition in the river channel (Anderson et al. 2005) are based on data from the Pool B Demonstration Project such as were presented as the reference in Figure 3-9. This expectation states that "In remnant river channels, mean thickness of substrate-overlying river bed deposits will decrease by $\geq$65%, percent of samples without substrate-overlying river bed deposits will increase by $\geq$165%, and the thickness of substrate-overlying river bed deposits at the thalweg will decrease by $\geq$70%." There are two caveats for using Demonstration Project data for reference conditions. First, the Demonstration Project did not completely reestablish the pre-channelization hydrologic characteristics to remnant river channels. Second, when the three metrics for substrate-overlying deposits were calculated for several years of reference data, they showed consistent increases (percent of samples without substrate-overlying deposits) or decreases (mean thickness of substrate-overlying deposits, thickness of substrate-overlying deposits at the thalweg) for four years, indicating that the reference values may not have reached a new equilibrium with the new flow regime (Anderson et al. 2005). Thus, the reference condition for substrate-overlying deposits is conservative, and using Demonstration Project data as the reference condition may conservatively estimate ecological integrity in the Kissimmee River. The power analysis for mean thickness of substrate-overlying deposits, percent of samples without substrate-overlying deposits, and thickness of substrate-overlying deposits at the thalweg suggests that the existing design is more than adequate to detect changes in all three metrics from the baseline value to the reference values proposed in Figure 3-9.

Future Studies

River channels are dynamic systems, which are constantly undergoing adjustment to changes in inputs of water and sediment, and this was true of the Kissimmee River prior to channelization (Warne et al. 2000). We have proposed two attributes (point bar formation and organic deposition in the river channel) for which specific expectations have been developed to evaluate the restoration of the Kissimmee River. Future studies of river channel geomorphology also should consider changes in channel morphology that might represent continued adjustments to flow, especially if these changes impact other aspects of the river ecosystem. For example, reestablishing flow during the Demonstration Project resulted in one unexpected channel cutoff event in the remnant channel adjacent to weir 3 (Scarlatos et al. 1990). Sedimentation and stability of river channels were two major components outlined in the monitoring programs for the Kissimmee River Restoration Project (U. S. Army Corps of Engineers 1991).

Once flow is reestablished, point bar formation will be much more dynamic than during the baseline period, and additional sampling effort may be required to capture these changes. This effort may include more frequent aerial photography (e.g., yearly), which also might be used to capture changes in vegetation. We used aerial photography only to determine the presence or absence of active point bars. More detailed information about point bar dynamics may require establishing additional transects on a few meander bends that could be sampled with greater frequency (e.g., after major flow events, end of wet season). Such a study would help link geomorphic changes to hydrologic drivers.

Restoring ecological integrity to the Kissimmee River requires reestablishment of natural river functions. Once flow is reestablished, the river channel, especially in recarved sections, is likely to undergo a period of adjustment to the new hydrologic regime, which will be reflected in changes in several different characteristics (e.g., sinuosity, width–depth ratio, entrenchment ratio, meander patterns). While we do not have specific expectations for how these characteristics will change in the restored river, quantifying these changes may help characterize the ecological integrity of the restored Kissimmee River, and may provide insights into other changes observed during the restoration. Special consideration should be given to measuring the stability of new river channels and connector channels, which might be accomplished with visual observations and depth measurements using the permanent transects laid out for this study. Surveys that established elevations for fixed points would enhance the value of these transects.

Our results show that connector runs differ morphologically from remnant river channels, and these differences may influence ecosystem function. Connector runs are about twice as wide as remnant channels (Table 3-3), and assuming that both channel types have the same depth, mean velocity across the channel cross-section in a connector run will have to be about half that of the remnant channel to carry the same discharge. In connector runs, the lowest velocities are likely to occur in the shallowest portions of the channel, especially along the northern side. These shallow, low velocity areas may be colonized by macrophytes or by benthic algae and may accumulate small quantities of fine particulate organic matter. Thus, these areas may become hotspots of benthic productivity within the restored river channel, which can serve to attract larger invertebrates (e.g., crayfish and grass shrimp), fish, and wading birds. Ultimately, the influence of connector runs on higher trophic levels will depend on their contribution to the habitat mosaic within the restored river channel. Understanding this contribution to the restoration project will require a longitudinal view of the river, which we have only begun to develop in this baseline geomorphology study.

Future studies should consider how geomorphology is linked to at least four other components of the ecosystem. First, hydrologic changes are expected to drive changes in the geomorphology of the restored river, and the linkage should be clarified. Second, changes in the quantity of organic deposition within the river channel is likely to be influenced by the amount of vegetation within the river channel, which is being measured on the same transects in the river channel vegetation study (Bousquin 2005). Third, reducing the amount of organic deposition within the river channel should reduce sediment oxygen demand, and changes in the quantity of deposition might be linked to changes in water column dissolved oxygen. Direct measurements of benthic respiration may clarify this linkage. Finally, reduction in the thickness and extent of the organic deposition in the river channel should enhance habitat quality for fish and invertebrates, and this linkage might be quantified.

ACKNOWLEDGEMENTS

Andrew G. Warne assisted with the original design of this study. Many people, especially R. Harper Carroll IV and Kim Kanter, spent long hours collecting core samples. Rene Hartless produced the graphs of the channel cross-sections. Colleen Chennell measured the channel cross-sectional areas. Bonnie Rose produced the maps. Discussions with Stephen G. Bousquin helped clarify relationships to the channel vegetation study. Jose J. Valdes helped clarify local geology. This chapter was benefited by comments from Arthur C. Benke, Clifford N. Dahm, Louis A. Toth, Joseph W. Koebel Jr. Jose J. Valdes, Martha Nungesser, Gary E. Williams, and David J. Colangelo.

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CHAPTER 4

DISSOLVED OXYGEN IN THE CHANNELIZED KISSIMMEE RIVER AND SEVEN REFERENCE STREAMS

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ABSTRACT: Dissolved oxygen (DO) is an important component for evaluating success of the Kissimmee River Restoration Project because it is essential to the metabolism of most aquatic organisms and because chronicly low DO concentrations have been observed since channelization. Seven remnant river channel stations and two canal (C-38) stations were sampled continuously, monthly, or weekly between March 1996 and June 1999. Remnant river channel DO data were used to establish a baseline for comparison with reference data. These data were used to estimate pre-channelized conditions, and will be compared with post-construction data collected in the future. Reference streams were chosen for their similarities to the pre-channelized Kissimmee River. Water quality stations covered a large geographic area and provided the scale needed for in-depth DO regime studies. Baseline period mean DO concentrations were consistently low at all stations; however, dry season (December–May) concentrations were slightly higher than wet season (June–November) concentrations. Dissolved Oxygen depth profiles exhibited a clinograde pattern at certain times of the year. Mean DO concentrations for the reference streams ranged from 2.4 to 6.0 mg/L during the wet season and from 3.7 to 7.4 mg/L during the dry season. Comparison of reference streams to baseline data suggests that channelization changed DO regimes in the river channel substantially. Reference stream data were used to develop a restoration expectation for changes in DO concentrations in the reconnected river channel after restoration of flow. Based on reference data, post-restoration DO concentrations are expected to increase from <1–2 mg/L to 3–6 mg/L during the wet season and from 2–4 mg/L to 5–7 mg/L during the dry season. Mean daily concentrations are expected to be greater than 2 mg/L more than 90% of the time. DO concentrations within 1 m of the channel bottom are expected to exceed 1 mg/L more than 50% of the time.

INTRODUCTION

Dissolved oxygen (DO) is one of the most frequently used indicators of water quality because it is easy to understand and relatively simple to measure (Belanger et al. 1985). Dissolved oxygen is essential to the metabolism of most aquatic organisms and can influence growth, distribution and structural organization of aquatic communities (Wetzel 2001). Oxygen concentration also affects the solubility and availability of many nutrients and can impact the productivity of aquatic ecosystems (Wetzel 2001).

Channelization of the Kissimmee River transformed the flowing blackwater river into a central drainage canal (C-38) composed of a series of reservoir-like pools. Flows through remnant river channels were eliminated (Koebel et al. 1999), allowing aquatic vegetation to encroach upon open water areas (Bousquin 2005) and oxygen-depleting organic sediments to accumulate over the river's sandy substrate (Anderson et al. 2005). The nine meter deep canal also drained the adjacent floodplain, thereby reducing the ratio of surface area to volume of water and possibly limiting the ability of the system to be re-aerated through wind and flow-induced mixing (Loftin et al. 1993). Chronically low DO concentrations in the remnant river channel became apparent after channelization and elimination of continuous flow. For these reasons, DO studies were identified as an important component of the Kissimmee River Restoration Evaluation Program.

Restoration of continuous, variable flow through reconnected river channels is expected to flush flocculent organic matter from the river channel bottom and increase DO concentrations by reducing biochemical and sediment oxygen demand and increasing atmospheric aeration. Continuous flow should restrict/preclude mid-channel growth of aquatic macrophytes and reduce the potential for deposition of organic matter over mid-channel substrates.

Baseline conditions were established by measuring DO in the remnant river channel after channelization. Reference conditions were determined by selecting seven nearby rivers and streams as reference sites. Preference was given to sites with plentiful DO data (collected preferably during the same period of record as baseline data). Baseline data were then used to develop a restoration expectation for DO in the Kissimmee River channel.

Objectives

The objectives of this study were to:

(1) Establish baseline and reference conditions for assessing effects of restored hydrology on DO regimes within the river channel;
(2) Quantify the impacts of channelization on DO in the river channel by comparing reference and baseline conditions; and
(3) Link assessments of baseline and reference data with restoration expectations for DO in the river channel.

METHODS

Baseline Conditions

Study Site

Baseline DO conditions were measured within remnant river runs and canal stations within Pool C (impact) and Pool A (control) (Figure 4-1). Monitoring sites were selected to cover a large geographic area. Canal stations near water control structures S-65A and S-65C monitored DO concentrations of water flowing into and out of the area to be restored. Sampled remnant river runs were approximately 20–30 m wide and 2–3 m deep, with little or no flow. Riverbed substrate consisted of flocculent, unconsolidated organic material (Anderson et al. 2005). River channel aquatic vegetation was dominated by ***Salvinia minima***, *Scirpus cubensis*, *Ludwigia peruviana* and *Nuphar lutea*. Approximately two thirds of the area between channel banks was vegetated (Bousquin 2005).

Continuous Data

Dissolved oxygen and water temperature were monitored continuously with a YSI 600R multiparameter water quality sonde (YSI Inc., Yellow Springs, Ohio) at three remnant river run stations in the impact and control areas (KRBN, KRDR and KREN; Figure 4-1). Sondes were fixed at a depth of approximately one meter at each station. Each sonde was wired to a Campbell CR10 datalogger (Campbell Scientific, Logan, Utah) and programmed to record data at 15 minute intervals (Figure 4-2). Data were uploaded automatically to the South Florida Water Management District (SFWMD) base station via radio signal every evening. Sonde calibration and maintenance were performed weekly according to YSI calibration procedures. Monitoring began at each station between July and October 1997.

Monthly Data

Dissolved oxygen was sampled monthly at seven remnant river run stations (KREA91, KREA92, KREA93, KREA94, KREA95, KREA97, KREA98) and at two canal stations (S65A and S65C) (Figure 4-1) in the control and impact areas. Samples were taken mid-channel at a depth of 0.5–1.0 m with a multiparameter water quality sonde generally about mid-day. Monitoring began at each station between March 1996 and December 1997.

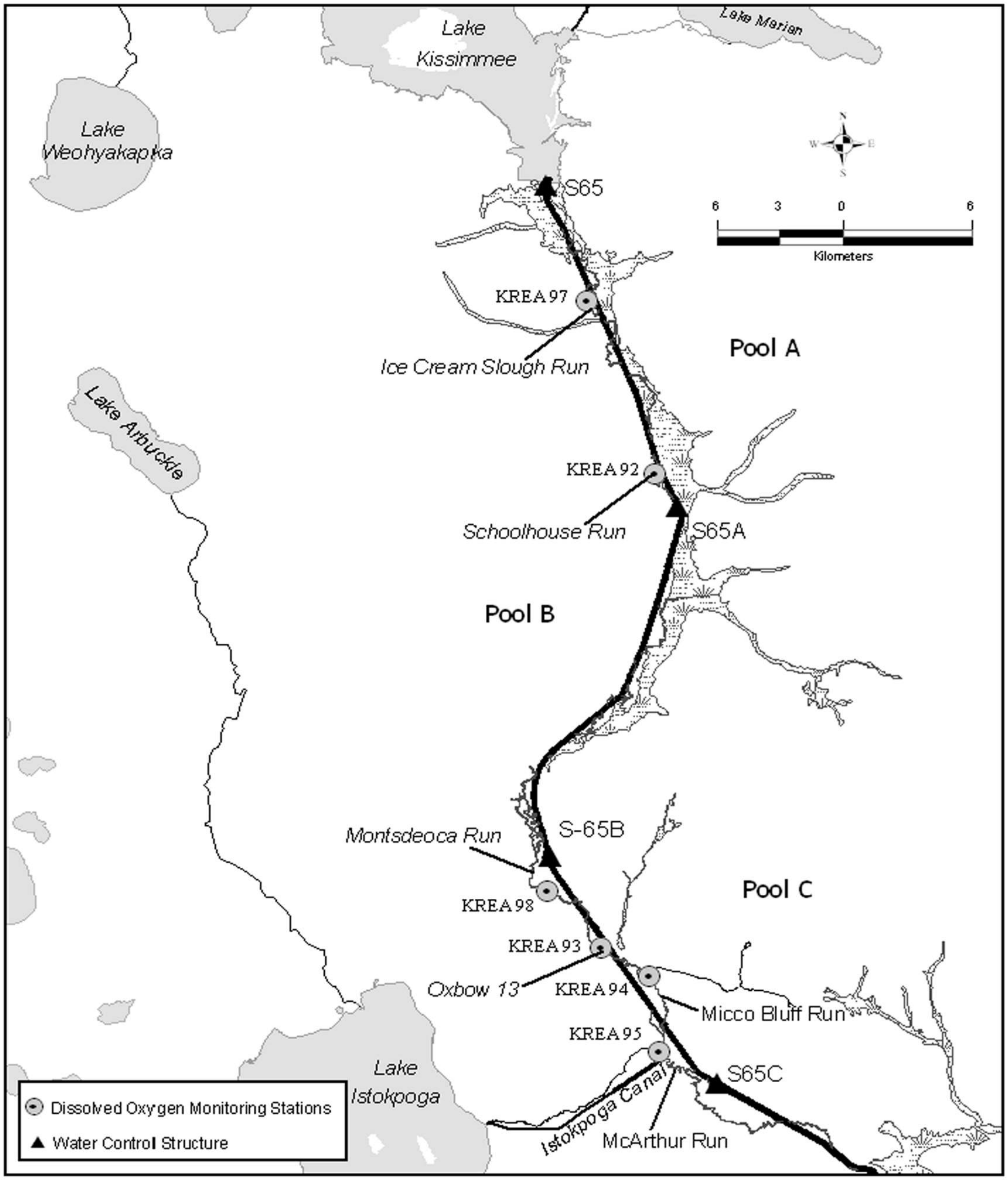


Figure 4-1. Dissolved oxygen monitoring stations along the channelized Kissimmee River.

Weekly Data

Weekly DO profiles were sampled to record changes in DO along a depth gradient as part of baseline data collection for Phase I construction monitoring (Colangelo and Jones 2005). Depth profiles were taken at two stations (K05 and K07) within Micco Bluff Run in Pool C (Figure 4-1) with a YSI 6920 multiparameter water quality sonde (YSI Inc., Yellow Springs, Ohio). Measurements were recorded at 0.5 m, 1.0 m, and every 1.0 m interval to within 0.5 m of the channel bottom. Monitoring began at each station in April 1999.

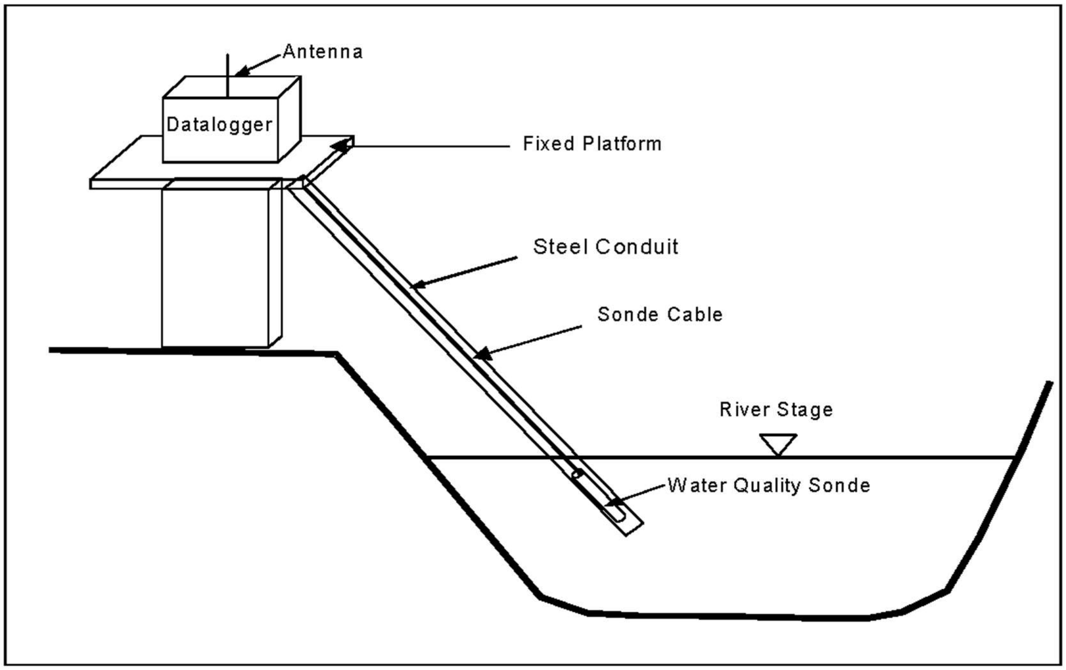


Figure 4-2. Continuous water quality monitoring station design.

Statistics

Sampling for this study was designed according to the before-after-control-impact (BACI) statistical design (Stewart-Oaten et al. 1986), which will be used to evaluate change. Data from all stations underwent quality assurance/quality control (QA/QC) processing prior to inclusion in the baseline data set. An average of 19.8% of data (% of days) collected at stations KRBN (21.6%), KRDR (19.7%) and KREN (18.1%) were eliminated by the QA/QC process. Equipment calibration data were referenced when DO data seemed unusually high, low, or erratic. Diagnostic readings (specific to the YSI DO sensor) recorded during calibration were used to determine whether values should be discarded. Dissolved oxygen values exceeding 100% saturation were discarded if associated Chlorophyll *a* data showed that algae were not present in sufficient quantities to cause supersaturation. Erratic data, such as changes in DO concentration >5 mg/L within 30 minutes with no change in water temperature, were discarded because these patterns are symptomatic of equipment failure.

Seasonal variation in DO was evaluated by comparing wet season and dry season values at each station because oxygen solubility decreases with increased water temperature. Nonparametric statistics were used because DO data were not normally distributed. The Wilcoxon rank-sum test was used to compare two-sample data (wet season vs. dry season and impact vs. control) and the Kruskal-Wallis test was used for multiple comparisons between remnant river channel stations within and among impact and control areas. Post hoc multiple comparisons were made using Dunn's test. Statistics were computed using SAS ver. 8.0 (SAS Institute, Cary, NC). All comparisons were considered significant at the $p < 0.05$ level.

Statistics were computed using mean daily DO values from stations KRBN, KRDR, and KREN (average of 15 minute interval data, 96 values each day), weekly values from stations K07 and K05 and monthly values from KREA91, KREA92, KREA93, KREA94, KREA95, KREA97, KREA98, S65A and S65C.

Diel DO curves were plotted for stations KRBN, KRDR and KREN using the average of readings recorded at each quarter hour (Eastern Standard Time) during the day (e.g., the 6 am DO reading at station KRBN was averaged for each day during the entire dry season). Curves were plotted for wet and dry season at each station.

Reference Conditions

Because no DO data were collected before channelization, reference conditions were derived from data for seven free-flowing, blackwater, south Florida streams. It was important to find streams where DO had been measured frequently throughout the year because DO concentrations change seasonally due to differences in water temperature and community metabolism. At least 11 samples were collected over a minimum of one year at each stream. Some streams were sampled for more than ten years (Table 4-1).

Table 4-1. Station data at seven reference sites and in the channelized Kissimmee River. Measurements were taken generally at mid-day, mid-channel with a dissolved oxygen probe at 0.5 m depth, at intervals ranging from weekly to bi-monthly.

Water Body	Station ID	County	Period of Record (mm/yy)	Freq.[1]	# Samples
Reference Sites					
Fisheating Creek	FECSR78	Glades	04/73–02/99	W-M	447
Arbuckle Creek	ARBKSR98	Highlands	02/88–02/99	BiM	86
Lake Marian Creek	DLMARNCR	Polk	04/82–09/85	M	37
Tiger Creek	ETIGERCR	Polk	04/82–06/85	M	33
Josephine Creek	JOSNCR17	Highlands	02/88–02/99	M-BiM	85
Boggy Creek	ABOGG	Osceola	08/81–03/99	M	202
Catfish Creek, S. Branch	ROSALIEC	Polk	11/84–09/85	M	11
Kissimmee River					
Ice Cream Slough Run (Pool A)	KREA 97	Polk	11/96–03/99	M	27
Rattlesnake Hammock Run (Pool A)	KREA 91	Polk	03/96–03/99	M	29
Schoolhouse Run (Pool A)	KREA 92	Polk	03/96–03/99	M	31
Montsdeoca Run (Pool C)	KREA 98	Highlands	03/96–03/99	M	14
Oxbow 13 (Pool C)	KREA 93	Highlands	03/96–03/99	M	29
Micco Bluff Run (Pool C)	KREA 94	Okeechobee	03/96–03/99	M	28
MacArthur Run (Pool C)	KREA 95	Highlands	12/97–03/99	M	31

[1] W = Weekly; M = Monthly; BiM = Bi-Monthly

Study Sites

Tables 4-2 and 4-3 summarize the physical and chemical characteristics of reference sites and the pre-channelized Kissimmee River. All reference streams are free-flowing blackwater Florida streams located within 145 km of each other and within 65 km of the Kissimmee River. Each reference stream has a low gradient (<6.5 cm/km) and a mean water temperature between 21.4 and 25.0°C. The chemical characteristics of these streams also are comparable. Six of the seven streams are lake fed and all streams empty into lakes.

Arbuckle Creek is located in Highlands and Polk Counties in central Florida, approximately 16 to 24 km west of Pools B and C of the Kissimmee River (Figure 4-3). Arbuckle Creek begins at the southern end of Lake Arbuckle and flows generally southeast for 39.8 km to Lake Istokpoga. The southernmost section of the creek has been channelized. Flow within Arbuckle Creek is unregulated and varies with the stage of Lake Arbuckle (Milleson 1978). Land use in the Arbuckle Creek watershed includes beef and dairy cattle as well as citrus production (Germain 1994). The water quality monitoring station is located at the southern end of the stream.

Table 4-2. Physical characteristics of reference streams and the pre-channelized Kissimmee River. Temperature data represent mean values.

Stream	Length (km)	Gradient (cm/km)	Drainage Area (ha)	Temp (deg. C)	Flows Into
Arbuckle Creek	39.8	6.2	381	24.99	Lake Istokpoga
Boggy Creek	18.8	2.4	88.8	21.41	E. Lake Tohopekaliga
Fisheating Creek	85.3	2.2	-	24.98	Lake Okeechobee
Josephine Creek	19.3	5.5	143	24.57	Lake Istokpoga
Lake Marion Creek	13.5	2.8	-	22.07	Lake Hatchineha
Catfish Creek, S. Branch	13	-	-	22.78	Lake Hatchineha, Lake Rosalie
Tiger Creek	3.7	3.6	-	23.61	Lake Kissimmee
Pre-channelized Kissimmee R.*	166	6.0 – 9.0	-	-	Lake Okeechobee

Sources: Bass (1983).
* = Source: Koebel (1995).

Table 4-3. Mean values for the period of record for chemical characteristics of reference streams (Table 4-1) and the pre-channelized Kissimmee River (One sample in 1955).

Stream	Sp. Cond. (µS/cm)	pH	Hardness (mg $CaCO_3$/L)	TP (mg/L)	TN (mg/L)
Arbuckle Creek	134.99	6.31	40.53	0.10	1.18
Boggy Creek	134.50	6.58	37.11	0.08	0.79
Fisheating Creek	220.97	6.60	60.21	0.17	1.76
Josephine Creek	147.42	6.09	45.42	0.04	1.25
Lake Marion Creek	144.97	6.58	54.88	0.07	1.54
Catfish Creek, S. Branch	168.27	6.69	59.05	0.05	2.15
Tiger Creek	104.10	5.69	21.83	0.05	0.95
Pre-channelized Kissimmee R.*	-	-	21.00	-	-

Source: SFWMD dbhydro database (mean values).
*= Source: Parker et al. (1955).

Boggy Creek is located within Orange and Osceola Counties in central Florida (Figure 4-4). Boggy Creek follows a 16 km channelized course from Lake Warren to Boggy Creek road, south of the Orlando International Airport. The remainder of the creek has not been channelized. There are two major branches of Boggy Creek. The east branch is the main branch, with its headwaters beginning at the southern lobe of Lake Conway and running for 19 km before emptying into East Lake Tohopekaliga. The west branch flows from Lake Jessamine to Boggy Creek Swamp, receiving runoff primarily from surrounding residential areas. The drainage landscape between Boggy Creek Swamp and East Lake Tohopekaliga is primarily citrus (SFWMD 2000a). The sampling station is located approximately one mile northwest of the northern shoreline of East Lake Tohopekaliga.

Fisheating Creek is an extensive riverine swamp system flowing for 85 km through Glades County before emptying into Lake Okeechobee (Figure 4-3). Fisheating Creek is currently the only free flowing tributary of Lake Okeechobee and for the most part has not been greatly impacted by human activities. Land use in and around the floodplain is mostly rangeland. Habitat types include cypress sloughs/mixed hardwood swamp forest, emergent marshes, willow thickets and open-water ponds (SFWMD 2000b). The sampling station is located at the bridge crossing of State Road 78. Data from the early 1940s suggest that chemical composition of water samples from Fisheating Creek and the Kissimmee River were similar before channelization (Parker et al. 1955).

Josephine Creek is located in Highlands County and flows east from Lake Josephine for 19 km before emptying into Lake Istokpoga (Figure 4-3). Land use in the Josephine Creek watershed includes rangeland and various wetlands. Water quality samples were taken approximately midway along the length of the creek.

Lake Marion Creek is located in Osceola and Polk Counties and flows southeast from Lake Marion and Snell Creek for 13.5 km before emptying into Lake Hatchineha (Figure 4-5). Land uses in this area are predominantly wetlands with some rangeland and agriculture near the mouth of the creek (Guardo 1992). The water quality sampling station is located at the mouth of the creek.

Catfish Creek is in Polk County and flows east-southeast approximately 13 km from Lake Pierce to Lake Hatchineha (Figure 4-5). The stream flows through the mostly wooded, Allen David Broussard Catfish Creek Preserve State Park. Water quality samples were collected midway along the creek.

Tiger Creek is located in Polk County and flows northeast between Tiger Lake and Lake Kissimmee (Figure 4-5). The Tiger Creek watershed is mostly wetlands (Guardo 1992). Water quality samples were taken about midway along Tiger Creek.

Comparison Methods

Mean DO concentrations for the wet season and dry season were calculated for each reference stream and for each remnant river channel station (grand wet or dry season mean for the period of record for each stream or remnant channel in the Kissimmee River). Dissolved oxygen concentrations from reference streams and the channelized Kissimmee River were compared using the Kruskal-Wallace test because data did not fit the normal distribution. The percentage of samples exceeding specific DO concentrations for each stream and remnant river channel station during the wet and dry season also were calculated for comparison. Only monthly data from the channelized Kissimmee River were used for comparison with reference streams because data from reference streams were collected using similar methods.

Although no water column profile data have been examined for reference streams (in most cases data do not exist), it is assumed that oxygen values near the bottom of the channel are usually higher when stream flow is present. This was observed during the Pool B Demonstration Project when weirs across C-38 diverted flow to adjacent remnant river runs. Although oxygen concentrations remained low, there was some evidence of more uniform DO profiles during the summer (Rutter et al. 1989). This information was used to help develop the restoration expectation for DO in the river channel.

RESULTS

Baseline Conditions

Continuous DO Data

Water temperature followed a predictable seasonal pattern and was similar at all three stations (Figure 4-6). Dry season water temperatures were cooler than wet season temperatures. DO concentrations were well below 100% saturation at all stations throughout most of the baseline period.

Mean daily DO concentrations for the period of record at stations KRBN, KRDR and KREN were 1.60 ± 0.08 mg/L (± 1 SE), 1.25 ± 0.06 mg/L and 1.33 ± 0.07 mg/L, respectively (Figure 4-7) and were significantly different between stations KRBN and KRDR, and KRBN and KREN (Kruskal-Wallis test, Dunn's test, $p < 0.05$). However, differences of less than 1.00 mg/L are likely not ecologically significant.

Mean daily DO concentrations at stations KRBN, KRDR, and KREN were <2 mg/L for 77–82% of the baseline period and <5 mg/L for 93–97% of the baseline period (Figure 4-8).

Wet season DO concentrations were significantly lower (Wilcoxon rank-sum test, $p < 0.05$) than dry season DO concentrations at each station, except KRBN (Table 4-4). Mean daily DO concentrations were usually <2 mg/L within remnant river channels during the wet season (Figure 4-6). Although mean DO concentrations at stations KRDR and KREN were significantly higher during the dry season than the wet season, mean DO concentrations did not vary by more than 1 mg/L seasonally.

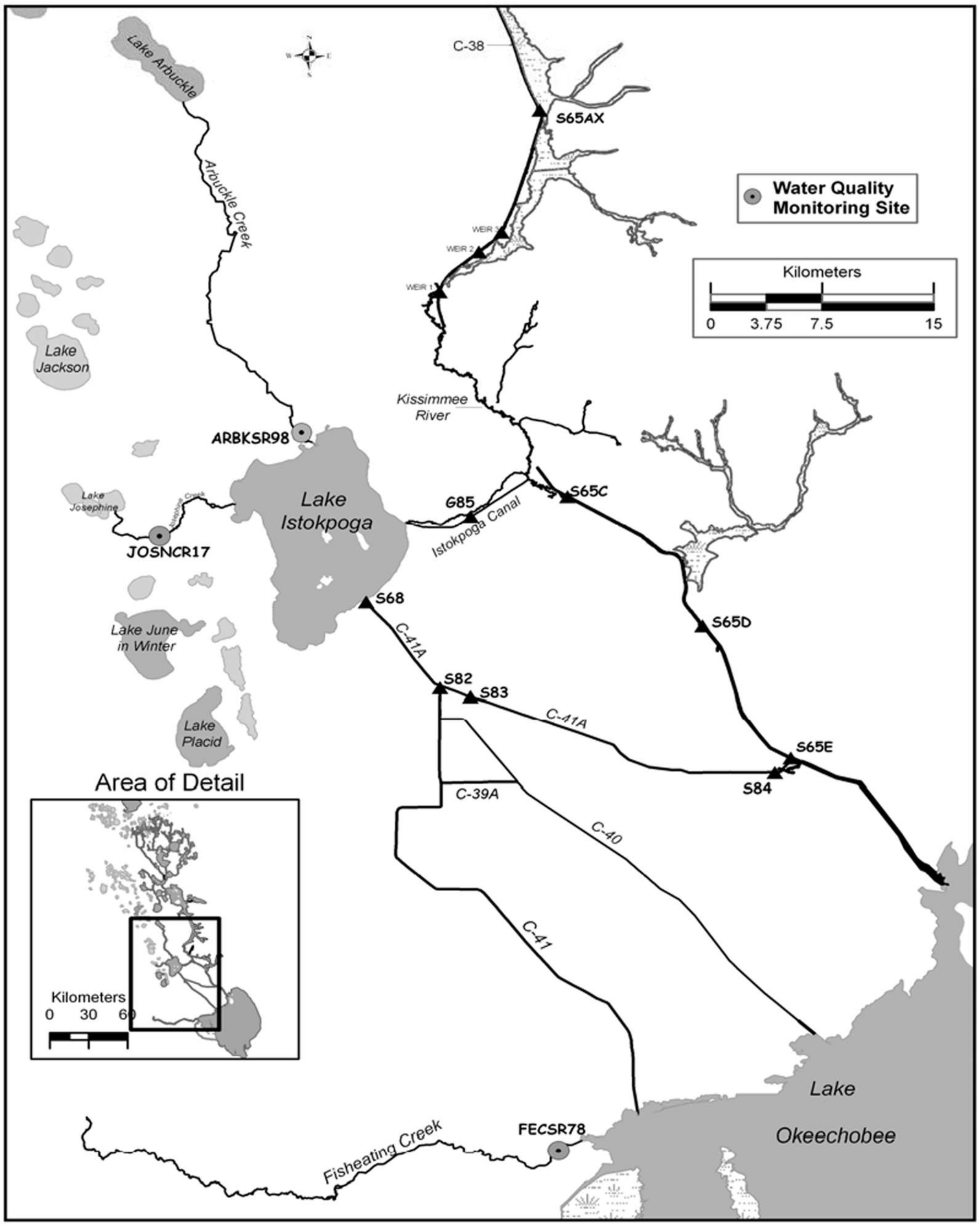


Figure 4-3. Location of Arbuckle, Fisheating and Josephine Creeks.

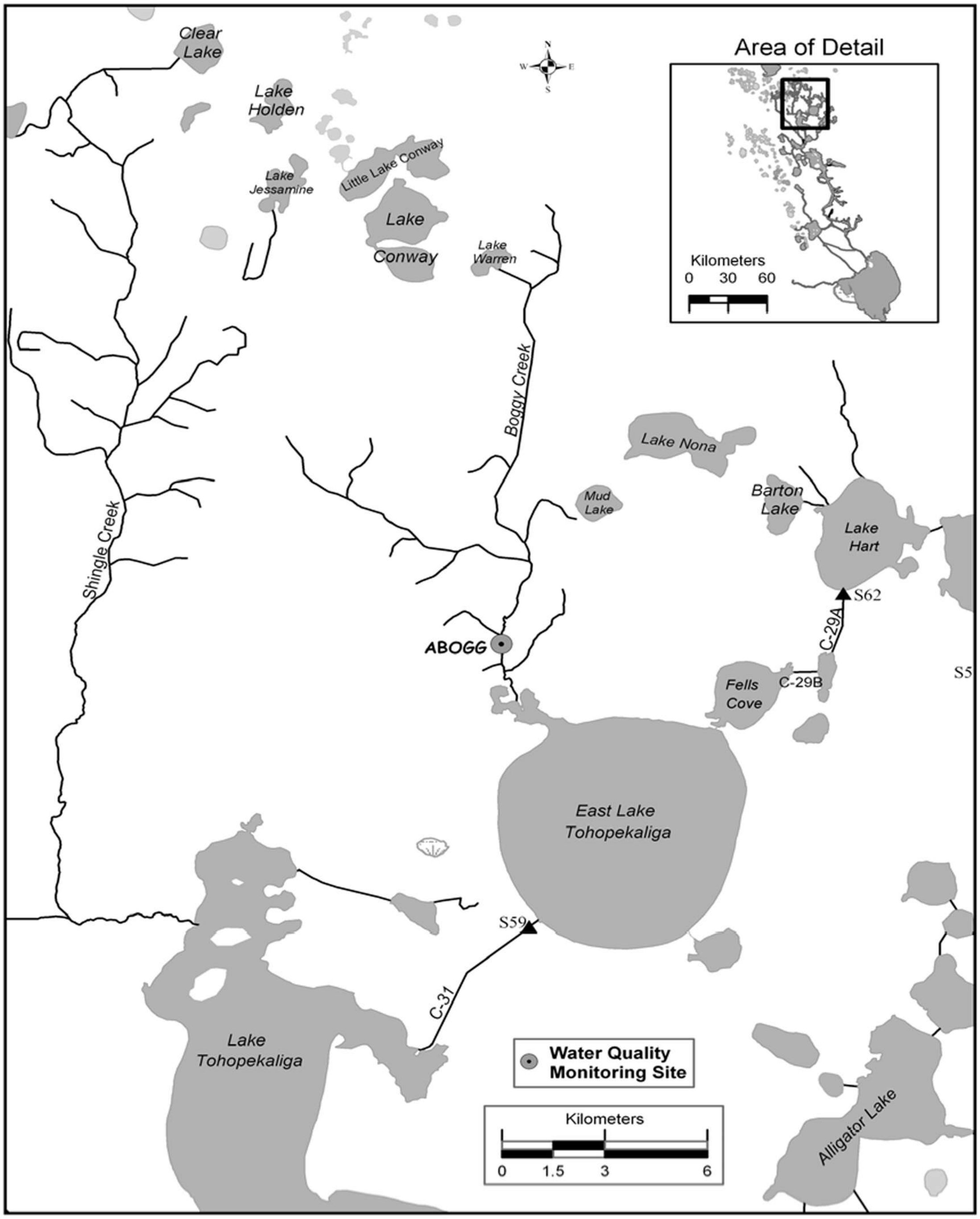


Figure 4-4. Location of Boggy Creek.

Diel DO curves (averaged by season over the period of record) within the channelized system showed little change over the diel period. However, dry season diel patterns were more variable than wet season curves at all stations. Figure 4-9 shows diel DO curves at station KRBN which is representative of stations KRDR and KREN as well. To illustrate variation in diel patterns on a daily scale, daily maximum and minimum DO concentrations at remnant river channel stations were plotted (Figure 4-10). Maximum and minimum daily DO values within remnant river channels varied by more than 2 mg/L 14% of the time and by more than 4 mg/L 2% of the time.

Weekly DO Data

Dissolved oxygen concentrations decreased rapidly with depth at station K05 within Micco Bluff Run from May through late June 1999 (Figure 4-11). The gradient at station K07 was not as strong as at station K05. During July–October 1999, DO concentrations became more uniform with depth and decreased to <2 mg/L throughout the water column at both stations. Dissolved oxygen concentrations near the channel bottom were often <1 mg/L.

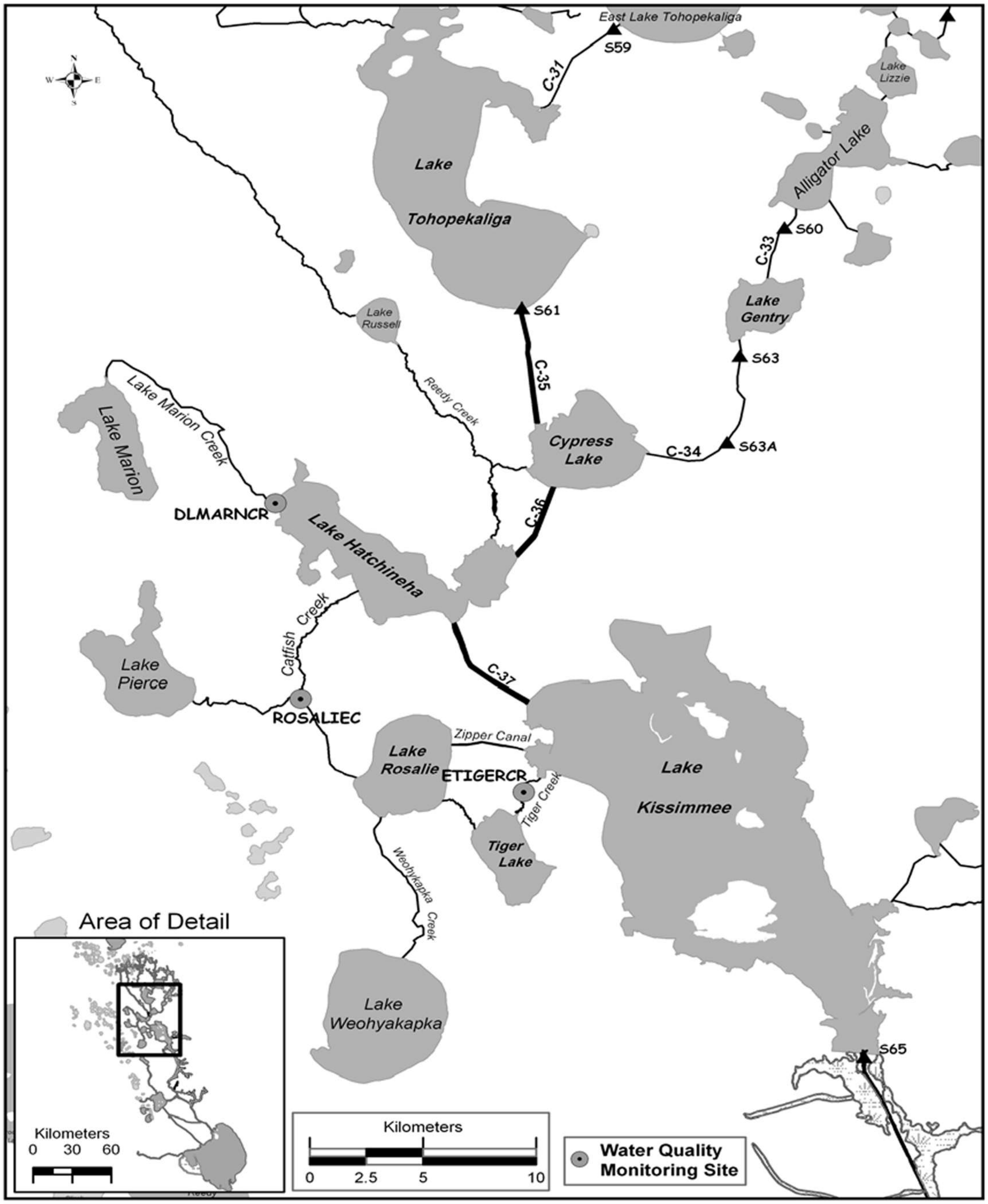


Figure 4-5. Location of Catfish, Tiger and Marion Creeks.

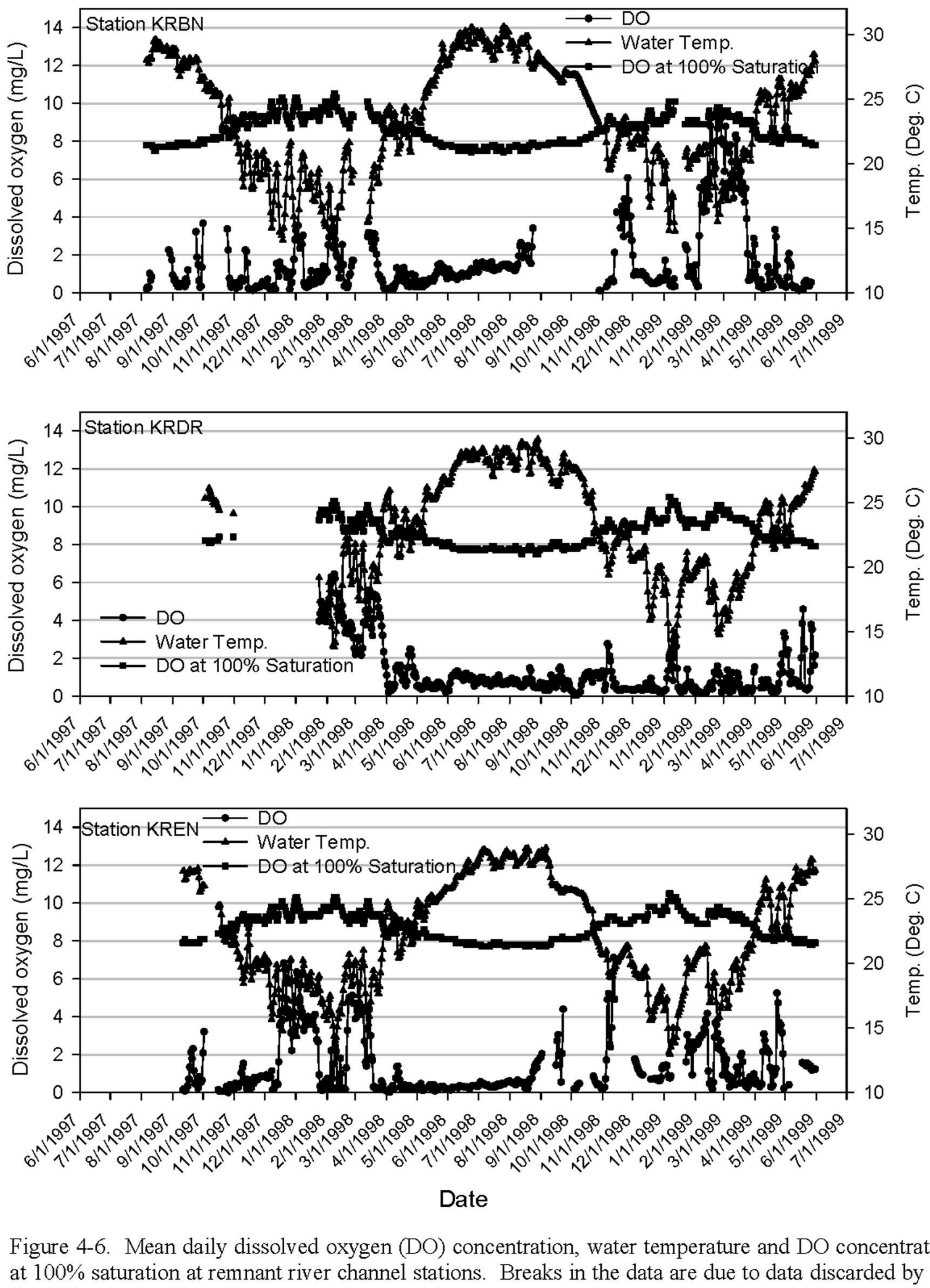


Figure 4-6. Mean daily dissolved oxygen (DO) concentration, water temperature and DO concentration at 100% saturation at remnant river channel stations. Breaks in the data are due to data discarded by the QA/QC process.

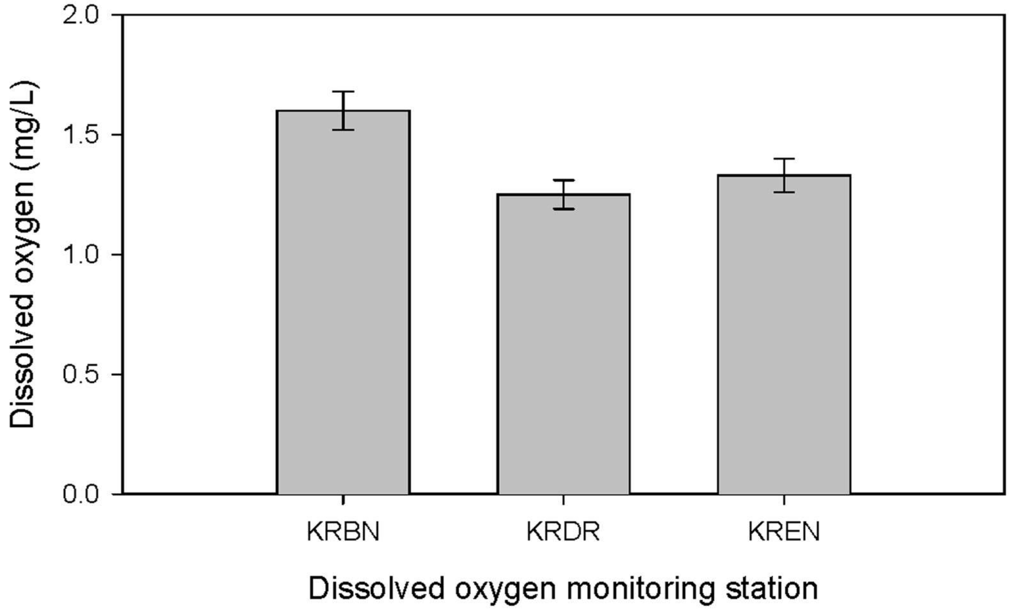


Figure 4-7. Mean daily dissolved oxygen (DO) concentration (± 1 standard error) at remnant river channel stations during the baseline study period. Means were calculated using average daily DO concentrations (average of 96 values per day) at each station.

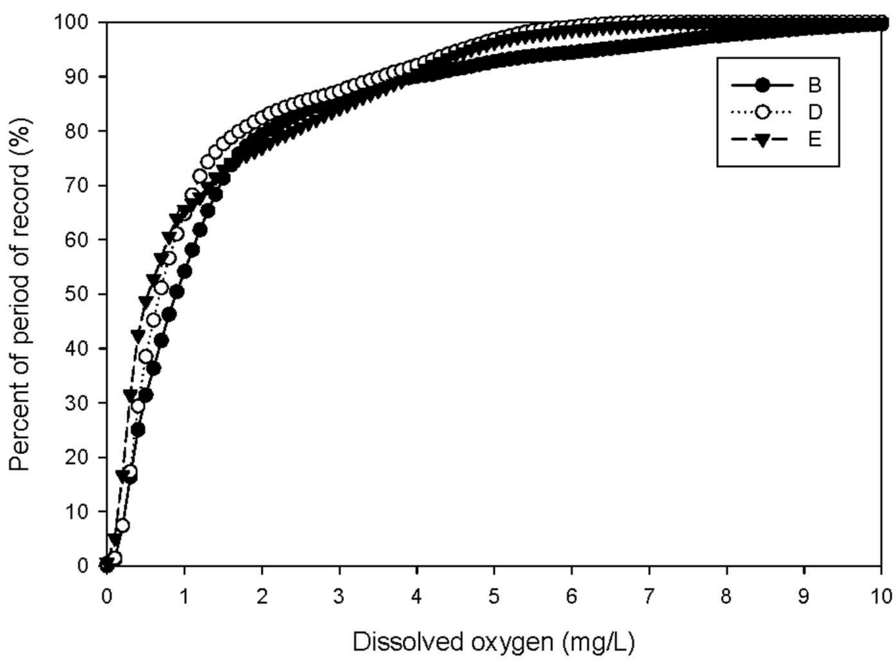


Figure 4-8. Cumulative frequency diagram of dissolved oxygen concentrations at stations KRBN, KRDR and KREN (July 1997–June 1999).

Monthly DO Data

Monthly DO data indicate that concentrations in remnant river runs were often extremely low and concentrations <2 mg/L were common (Figure 4-12). Percent oxygen saturation at all stations was usually <30%. Mean DO concentrations were similar at all stations (Kruskal-Wallis test, $p < 0.05$). Mean DO concentrations (Figure 4-13) at each station ranged from 0.7–1.9 mg/L for the wet season. Mean dry season DO values (2.5–3.8 mg/L) were higher (Wilcoxon rank-sum test, $p < 0.05$) than wet season values. Mean monthly DO concentrations were similar for the control and impact areas (Wilcoxon rank-sum test, $p > 0.05$)

Values at S-65A and S-65C followed similar patterns, but DO tended to be higher at S-65A in 1996–1997 and lower in 1998–1999 (Figure 4-14). Mean monthly values were low during the summer months when water temperatures were highest. However, DO was usually well below saturation levels and fell below 30% saturation during the summer.

Table 4-4. Comparison of mean DO concentrations (± 1 standard error) during the wet (June 1–November 30) and dry (December 1–May 31) seasons of the baseline sampling period.

Station	Season	N	Mean
KRBN	Wet	179	1.30 ± 0.08
	Dry	338	1.79 ± 0.12
KRDR	Wet	183	0.77 ± 0.03
	Dry	305	1.53 ± 0.09 *
KREN	Wet	197	0.78 ± 0.08
	Dry	319	1.66 ± 0.09 *

* = significant difference at the $p < 0.05$ level, Kruskal-Wallis test, Dunn's test, $p < 0.05$

Disolved Oxygen concentrations were significantly different at S-65A and S-65C. During the first half of the baseline period, S-65A was sampled more often in the afternoon (~ 1:00–4:00 pm), while S-65C was sampled more often in the morning (~ 9:00–11:00 am). During the latter half of the baseline period, the reverse occurred. When S-65A was sampled in the afternoon, 78% of DO measurements were higher at S-65A than S-65C.

However, even though DO was usually greater at S-65C late in the baseline period (when afternoon sampling predominated), it was often lower prior to July 1998 (i.e., the relationship between time and DO becomes ambiguous in the middle of the baseline period). Consequently, when S-65C was sampled in the afternoon and S-65A was sampled in the morning, less than half (47%) of DO measurements from S-65C were greater than values from S-65A. Nearly identical trends were obtained when only wet season samples were considered.

Reference Conditions

Mean DO concentrations ranged from 3.7 to 6.0 mg/L during the wet season and from 5.4 to 7.4 mg/L during the dry season (Figure 4-13).

Dissolved oxygen concentrations were significantly higher during the dry season than during the wet season (Wilcoxon rank-sum test, $p < 0.05$). In five of the seven streams, DO was >5 mg/L in more than 50% of the samples. In all seven of the reference streams more than 90% of the samples had concentrations greater than 2 mg/L (Figure 4-15).

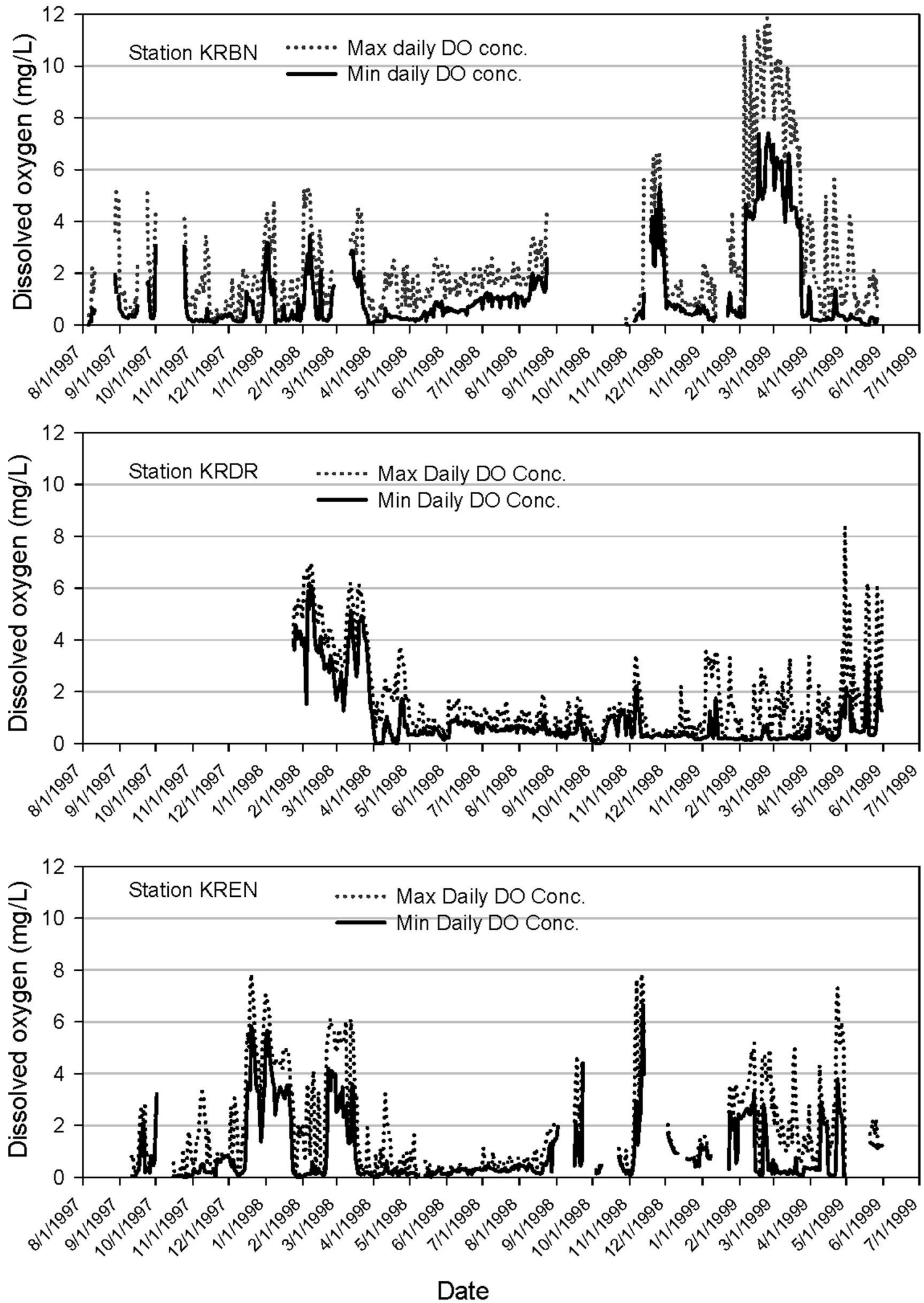


Figure 4-10. Maximum and minimum daily dissolved oxygen concentrations at remnant river channel stations during the baseline study period.

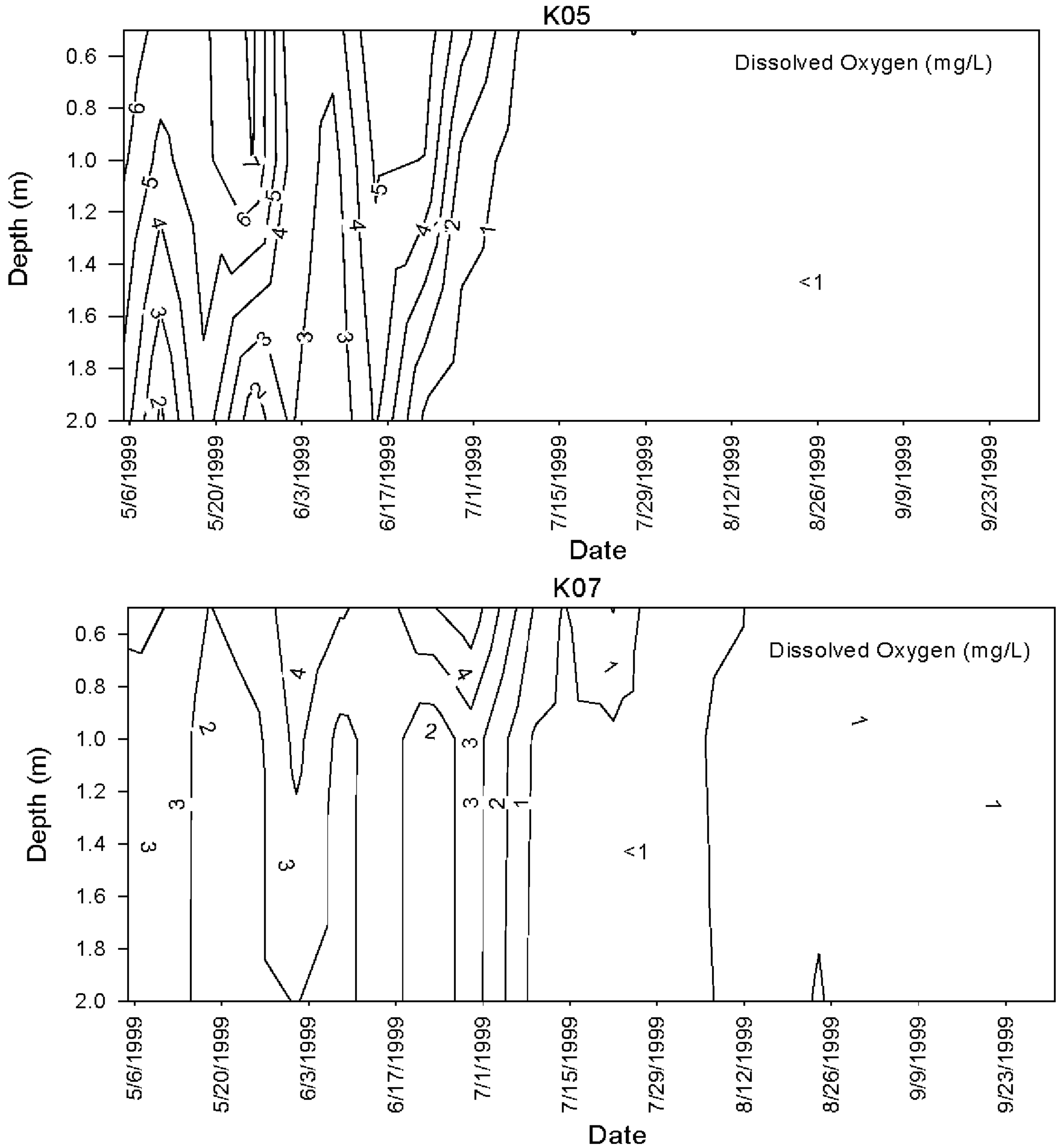
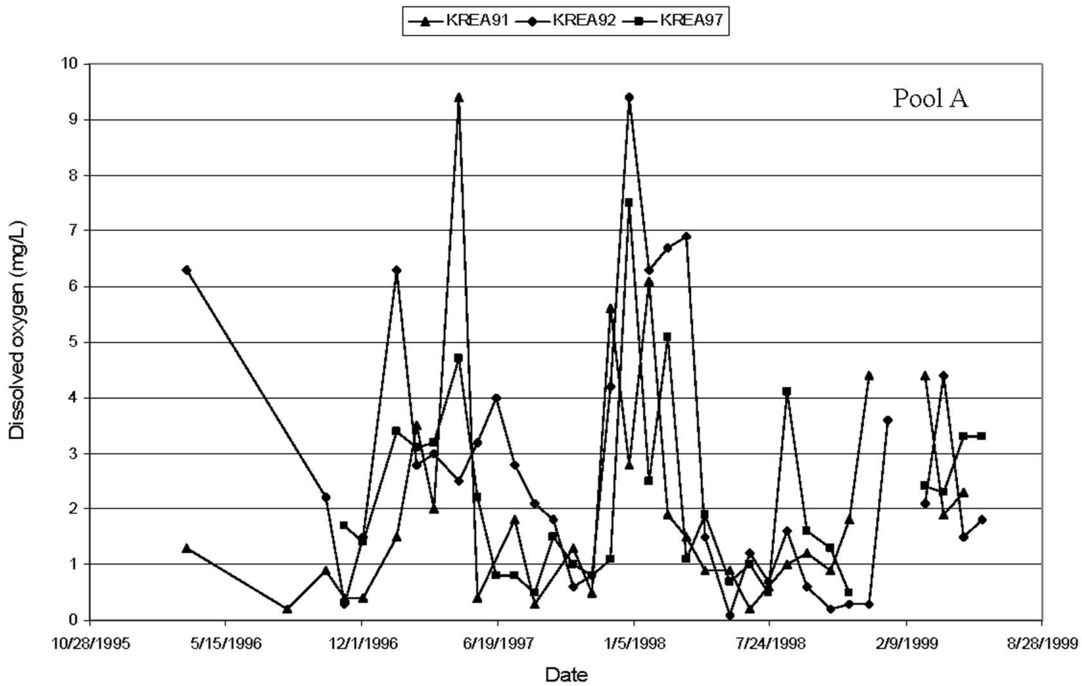


Figure 4-11. Vertical gradient of dissolved oxygen concentrations (mg/L) within Micco Bluff Run during May–October 1999.



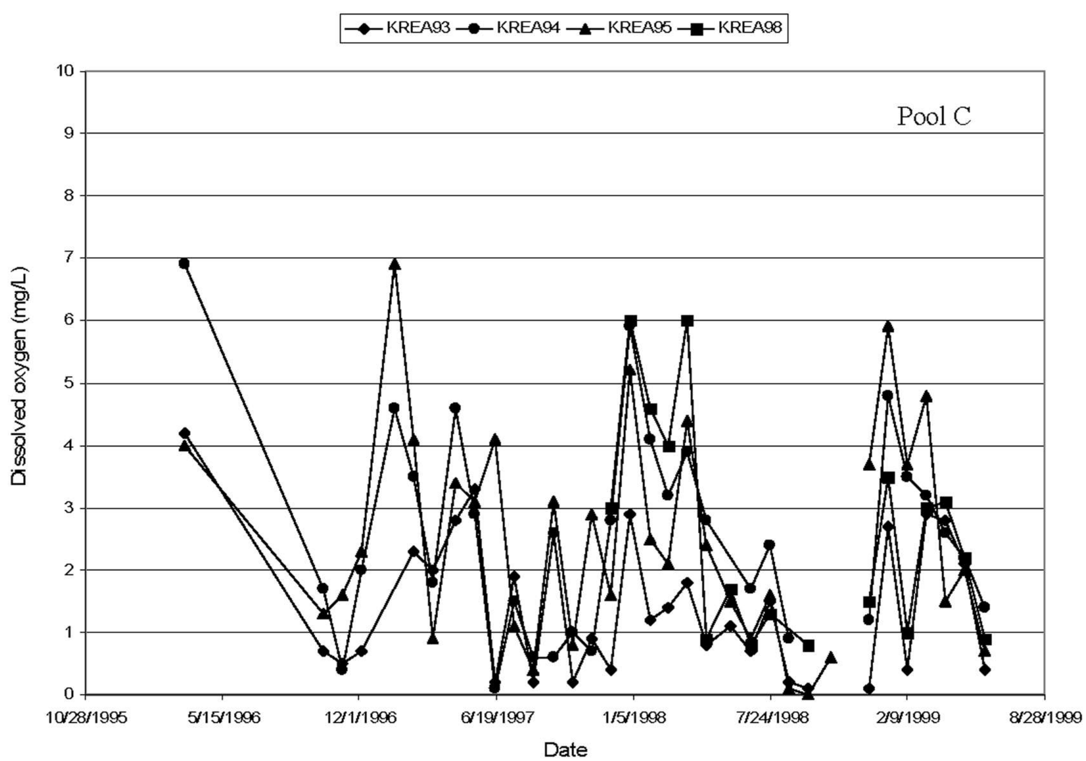


Figure 4-12. Monthly dissolved oxygen concentrations at 0.5 m in remnant river runs of Pools A and C.

Baseline-Reference Condition Comparisons

Mean DO concentrations from reference streams were significantly greater than those from the channelized Kissimmee River (Kruskal-Wallace test, $p < 0.05$). This difference is likely ecologically significant because fish and other aerobic aquatic organisms may become stressed at concentrations less than 2 mg/L (Moss & Scott 1961, Davis 1975, Smale and Rabeni 1995, Matthews 1998)

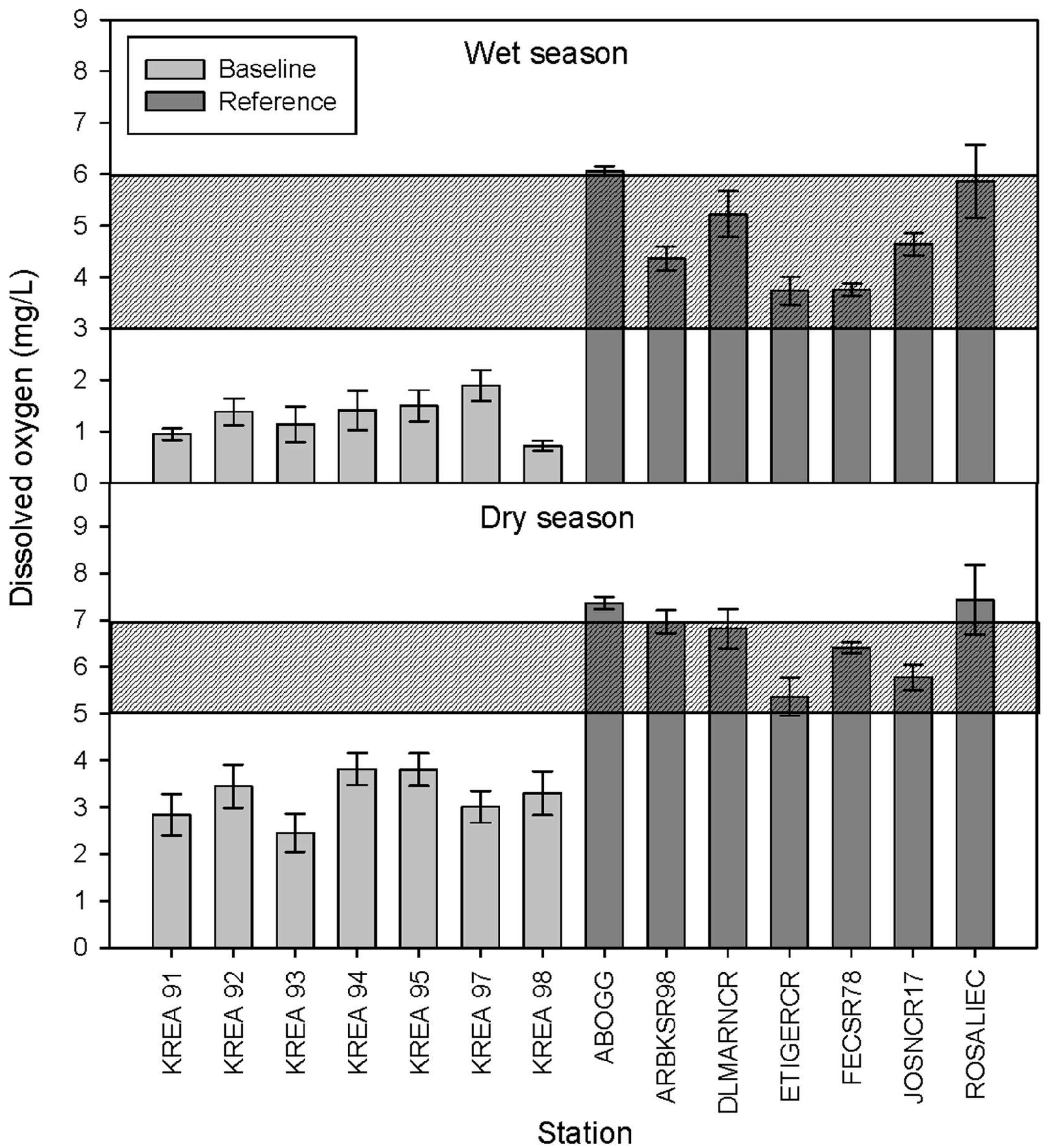


Figure 4-13. Mean (± standard error of the mean) dissolved oxygen (DO) concentrations in free-flowing, blackwater, south Florida streams and remnant runs of the channelized Kissimmee River during the wet and dry season. Cross-hatched area represents expected range of DO concentrations in the Kissimmee River after restoration.

Mean wet season DO concentrations were 4.2 mg/L for all reference streams combined and 1.3 mg/L for the channelized Kissimmee River. Mean dry season DO was 6.1 mg/L and 3.1 mg/L for reference streams and the channelized Kissimmee River, respectively. Grand means for DO concentrations in the channelized Kissimmee River and reference streams were 2.2 mg/L and 5.2 mg/L, respectively.

DISCUSSION

Baseline Conditions

Data collected continuously, weekly, and monthly showed that DO concentrations in the channelized Kissimmee River were persistently low. Continuous monitoring is the most accurate method for measuring changes in DO because concentrations can be monitored at short time intervals over a long period of time (years). However, continuous monitoring is expensive and can not be accomplished at the same spatial scale as weekly or monthly monitoring. Monthly monitoring is useful because a large area can be spot-sampled in a short period of time (several hours), which can show the extent of spatial variation within a system. Weekly water column profiles also are useful because they can be collected over a large area, and show the extent of variation within the water column. These monitoring strategies will be useful for assessing changes in DO dynamics as flow is restored.

Continuous Data

The relatively high percentage of DO data that were discarded by the QA/QC procedure was largely due to equipment failure, calibration error, or extremely low DO concentrations, which tend to foul the sensor and cause erratic readings. YSI Inc. reported that versions of their Rapid-Pulse DO sensors could expire prematurely, thus corrupting any data collected after the sensor stopped operating properly. This problem has been corrected in newer versions of the sensor. Much of our discarded data may be attributed to these technical problems.

Dissolved oxygen concentrations were low at all remnant river channel stations and all stations followed the same general trends. Greater DO concentrations at station KRBN than at stations KRDR and KREN may have been largely due to data collected during February 1999–April 1999, when DO concentrations at station KRBN were near 100% saturation, while DO concentrations at stations KRDR and KREN remained relatively low (Figure 4-6). This localized increase in DO concentration may have resulted from an increase in algal photosynthesis in the water column. Chlorophyll *a* concentrations (an indicator of algal biomass) in Oxbow 13 (near KRBN) during this time period were higher than in all other Pool C runs (Jones 2005). Maximum and minimum daily DO concentrations during this period differed as much as 3–4 mg/L, which is indicative of a diel pattern that may occur during an algae bloom (Wetzel 2001).

Dissolved Oxygen concentrations remained well below 100% saturation for most of the baseline period. Water bodies with large quantities of organic matter (both sediment and dissolved) usually have oxygen concentrations that are appreciably below the saturation point (Wetzel 2001). Dissolved Oxygen values observed within remnant river channels were well below typical DO concentrations for streams in the region and for reference streams (Friedemann and Hand 1989).

Dissolved oxygen concentrations were greater during the dry season due to cooler water temperatures than during the wet season. Solubility of oxygen is affected nonlinearly by temperature and increases in colder water (Wetzel 2001). Additionally, oxygen demand (water column and sediment respiration) is generally lower because of cooler water temperatures (Bott et al. 1985). Wind-induced aeration also is generally higher during the dry season than during the wet season due to cold fronts (causing increased wind speed) that pass through the region.

Dissolved oxygen concentrations were related to time of day, suggesting that diel changes in water column DO concentrations were linked to photosynthesis and respiration by phytoplankton and aquatic macrophytes, and decomposition of organic bottom sediment by bacteria. However, diel oxygen curves in humic colored waters such as the Kissimmee tend to be flater than in other systems because reduced light penetration (Belanger et al. 1985) results in reduced oxygen production in the water column.

Weekly Data

Dissolved oxygen concentrations in remnant river channels exhibited a vertical gradient during spring and early summer 1999. Vertical stratification is likely caused by bacterial decomposition of organic matter along the bottom of the remnant river channel, coupled with low or no flow. Vertical stratification of DO also followed a thermal gradient. Thermal stratification of remnant river channels was similar to a warm monomictic lake (which circulates freely in the winter and stratifies during the summer (Wetzel 2001)). Warm, less dense water in the epilimnion develops thermal resistance to mixing with the cooler more dense water of the hypolimnion during early to mid-summer. As DO in the hypolimnion is depleted, a clinograde oxygen profile is formed. By mid-summer, DO concentrations throughout the water column have been depleted. These patterns are similar to data collected between July and October 1989 (Toth 1993, Koebel 1995).

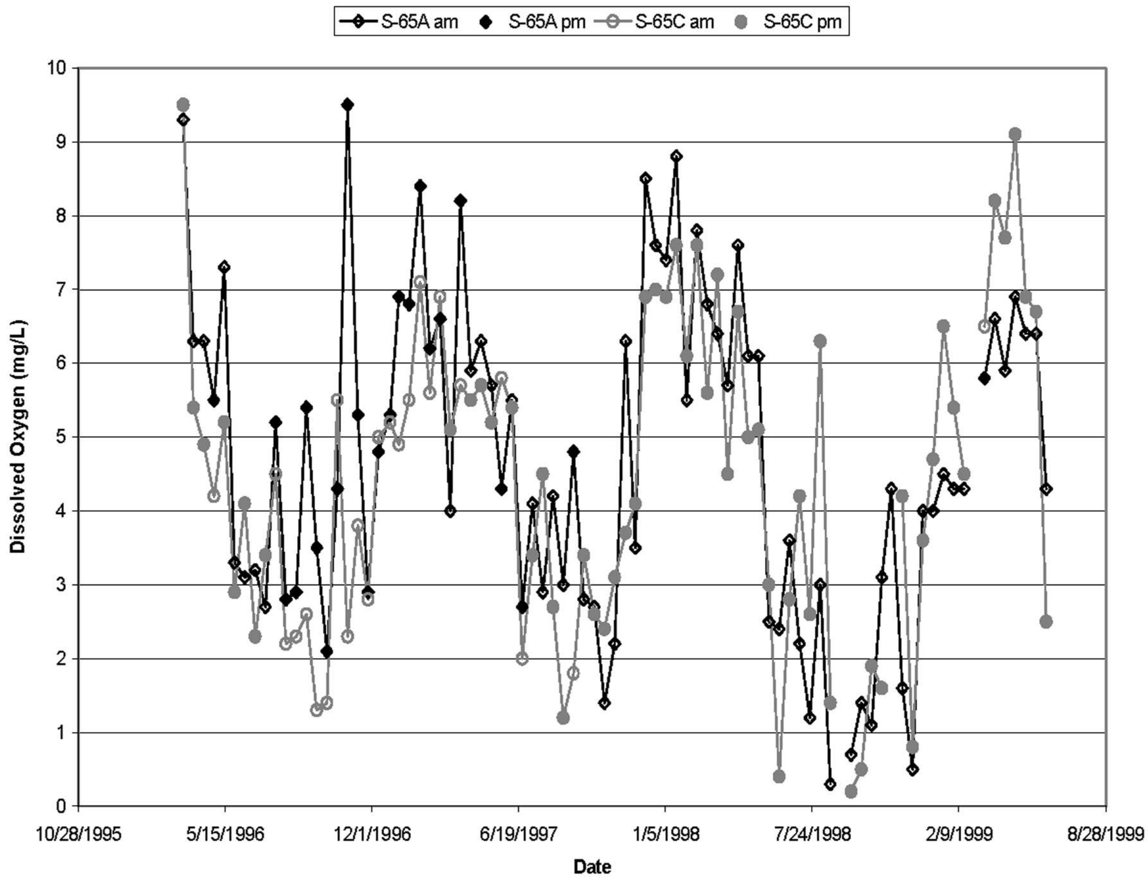


Figure 4-14. Dissolved oxygen concentrations at S-65A and S-65C at 0.5 m.

Monthly Data

The relationship between DO concentration and the time of day measurements were taken may be related to phytoplankton communities. Common cyanobacteria in C-38, including *Anabaena sp.* and *Microcystis aeruginosa*, tend to rise to the surface on calm, sunny days. As these phytoplankton produce oxygen, DO concentrations increase during the day. Therefore, near-surface DO measurements would tend to be higher in the afternoon than in the morning, especially when more phytoplankton are present. Chlorophyll *a*, an indicator of phytoplankton biomass, was usually greater at S-65A in 1996. Whether this was due to more algal biomass in the water column or phytoplankton rising to the surface during the day is unknown, as only near-surface samples were collected. In either case, afternoon sampling at this structure resulted in higher Chlorophyll *a* concentrations compared to S-65C.

Dissolved Oxygen concentrations at remnant river stations varied considerably. Schoolhouse Run (KREA92) in Pool A and MacArthur Run (KREA95) in Pool C often had higher DO concentrations than other runs even though they were usually sampled earlier in the day. These two runs are at the lower ends of

Pools A and C and tend to be wider and less choked with vegetation. This may have allowed more solar insolation and greater phytoplankton productivity and wind-induced reaeration than in other runs.

These results demonstrate the importance of considering time of day, among other factors, when comparing DO data from different locations. Post-construction samples will be taken at approximately the same time of day as baseline samples.

Data collected during the baseline period are adequate for assessing restoration of DO regimes within the Kissimmee River. Continuous monitoring resulted in reliable data on a 15 minute frequency, while weekly and monthly sampling covered a wide geographic area. Post-construction data collection began in June 1999 and will continue until some time after the Upper Basin regulation schedule is implemented.

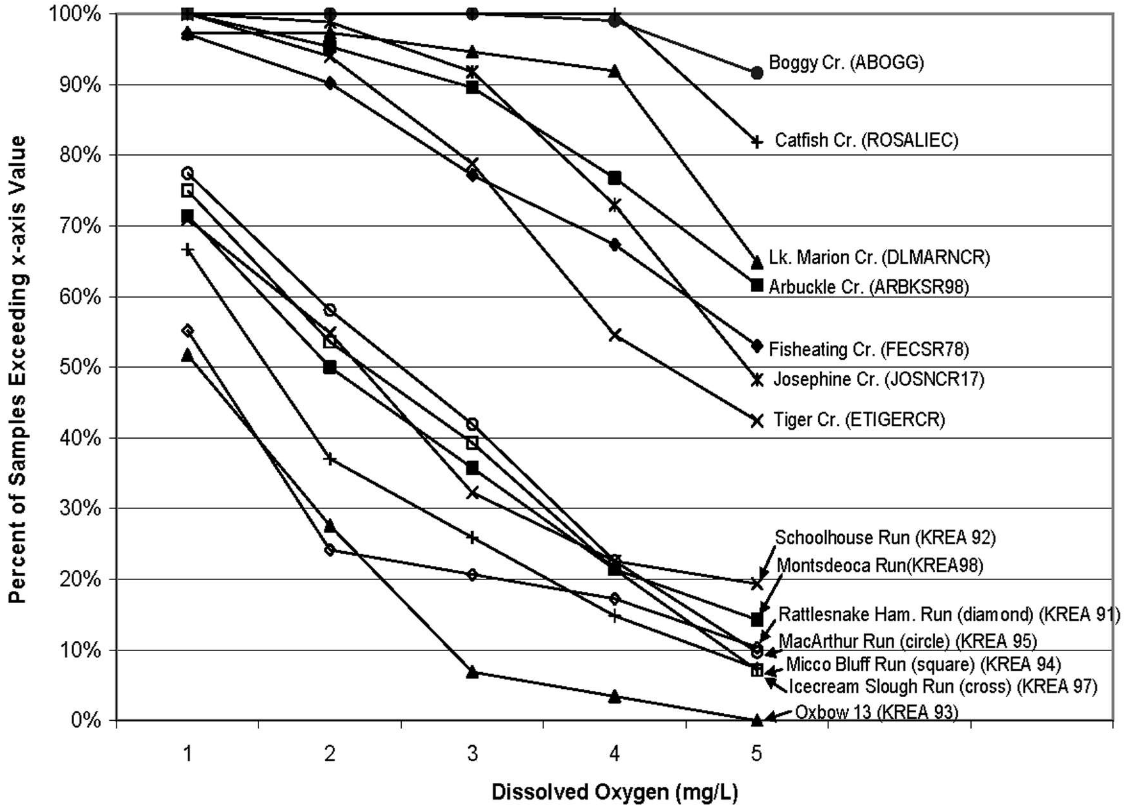


Figure 4-15. Dissolved oxygen concentrations at 0.5 m depth in south Florida reference streams and remnant river runs of the channelized Kissimmee River.

Reference Conditions

The reference streams are most similar to the pre-channelized Kissimmee River in stream gradient, proximity to the Kissimmee River and stream type. Reference stream gradient varied from 2.2 to 6.2 cm/km compared to the pre-channelized Kissimmee, which had a gradient varying from 6 to 9 cm/km (Table 4-2). Choosing streams with low gradients was important because DO concentration is affected by aeration through turbulent flow. All reference streams were located within 65 km of the Kissimmee River and therefore had similar climatic conditions and water temperatures. Water temperature is one of the most critical factors affecting DO concentrations, because the solubility of oxygen in water increases with a decrease in temperature. All reference sites were sand-bottomed blackwater streams with moderate to low impact from human activity.

Reference site data may be limited for several streams because the period of record was relatively short (<2 years). Also, the period of record does not overlap for all streams making it difficult to compare data among sites. Reference streams were shorter and narrower and had smaller drainage areas than the pre-channelized Kissimmee River. Flow velocity and water depth of reference sites may also differ from the pre-channelized Kissimmee River. Additionally, some sections of reference streams receive more shading from riparian vegetation than the pre-channelized Kissimmee River.

These reference streams may not completely represent conditions that existed in the pre-channelized river. However, due to similarities in flow, watershed characteristics and water quality, these streams likely approximate oxygen regimes in the river before channelization, they are the best analog available.

Expectations

Based on comparisons of baseline and reference data, mean daytime concentration of DO in the Kissimmee River channel is expected to increase from <1–2 mg/L to 3–6 mg/L during the wet season and from 2–4 mg/L to 5–7 mg/L during the dry season. Mean daily concentrations are expected to be greater than 2 mg/L more than 90% of the time. Dissoved Oxygen concentrations within 1 m of the channel bottom are expected to exceed 1 mg/L more than 50% of the time.

Other Studies

Other factors under investigation are relationships between DO concentrations and precipitation, stage recession rates and groundwater inputs within the channelized system. Belanger (1994) found that critically low DO conditions occurred when the previously dry floodplain was inundated and rapidly drained. Toth (1988) hypothesized that a September 1988 fish kill caused by extremely low DO was likely linked to rapid drainage of water from the floodplain into remnant river channels and C-38. Seepage of groundwater also may contribute to low DO concentrations. Concentrations of ammonium-N and BOD in groundwater can be high and may represent a significant source of oxygen uptake (Belanger 1994). Dissolved oxygen data collected during the baseline period will be used with groundwater seepage, rainfall and flow data to investigate these relationships.

A study to monitor water quality during Phase I of the Kissimmee River Restoration project was initiated in June 1999 (Colangelo and Jones 2005). The objective of this study was to monitor changes in water quality as flow was diverted into remnant river runs, and as the old river channel was flushed. Additionally, vertical gradients in dissolved oxygen and turbidity upstream and downstream of the construction were monitored. Phosphorus concentrations and loads downstream of the construction also were monitored.

Future research may include a water column community metabolism study similar to that performed by McCormick et al. (1997) in the Florida Everglades. Diel DO data can be used in conjunction with water temperature and oxygen diffusion rates to estimate baseline and post-construction gross primary production and aerobic respiration within the water column. This type of study can provide information about the changing ecological health of the system.

ACKNOWLEDGMENTS

The authors thank Ben Bolan, R. Harper Carroll IV, Pat Davis, Kim Kanter and Lisa Kreiger for their assistance with data collection and equipment calibration and Bonnie Rose for map preparation. We also would like to thank Lou Toth, David H. Anderson, Stephen G. Bousquin, Gary E. Williams, Mike Chimney, and Jim Laing for comments and suggestions that improved this report and the associated restoration expectation.

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CHAPTER 5

WATER QUALITY IN THE CHANNELIZED KISSIMMEE RIVER

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ABSTRACT: To document water quality conditions in the channelized Kissimmee River, remnant river run and C-38 stations in Pools A and C were monitored for three years prior to Phase I construction. The range of median turbidity values was low (1.2–3.0 NTU), and median concentrations of chlorophyll *a* (3.3–17.3 mg/m^3), total phosphorus (0.034–0.071 mg/L), total nitrogen (1.12–1.33 mg/L), specific conductance (110–277 µS/cm), and pH (5.88–6.87) were moderate. Chlorophyll *a* concentrations rose above 40 mg/m^3 occasionally, indicating the presence of algal blooms. Variations in color, organic carbon, specific conductance, chloride, alkalinity, and pH reflected seasonality in headwater and tributary discharges. Higher ionic content in some river runs may be indicative of agricultural inflows. Small, but statistically significant differences existed between Pool A and Pool C runs. Chlorophyll *a*, turbidity, total phosphorus (TP), and alkalinity were higher in Pool A, while dissolved inorganic nitrogen, specific conductance, and chloride were higher in Pool C. Runoff from ditched tributaries increased phosphorus concentrations in one Pool C run. In contrast, a Pool A run receiving inflow from a floodplain marsh had more moderate concentrations. Total phosphorus concentrations at the upper four C-38 structures followed the trend of concentrations in Lake Kissimmee, which increased slightly after declining in the 1980s. A larger increase at S-65 may have been caused by phosphorus release and wind-induced sediment resuspension in the lake following artificial drawdown and hydrilla control, muck and tussock removal or dredging in the lake's south end, local impacts at the water control structure, or inputs of phosphorus near the lake's outlet. Elevated concentrations of phosphorus at S-65, coupled with high discharges from a succession of storms, resulted in disproportionately large phosphorus loading from S-65 in 1998. However, agricultural watersheds of Pools D and E remained the most concentrated source of phosphorus in the Kissimmee Basin. Concentrations at S-65D and S-65E were substantially higher than at the upstream structures. After restoration, turbidity and total suspended solids (TSS) concentrations are expected to remain low. As stated in the expectation compendium (Jones 2005), mean turbidity in the restored river channel should not differ significantly from mean turbidity in similar south Florida streams (3.9 NTU), and the median TSS concentration should not exceed 3 mg/L.

INTRODUCTION

Water quality is an important component of habitat quality, and its influence on fish, aquatic invertebrates, vegetation, microorganisms, and ecological dynamics is well known. Accordingly, the quality of water can be an important determinant of species presence, diversity, abundance, reproduction, and productivity. Water quality also can be altered by physical modifications and shifts within biotic communities, so restoration of habitat can potentially improve water quality, which in turn may promote desired changes in the biota.

In addition, many water quality parameters respond quickly to physical and hydrologic changes and can serve as early indicators of changes in habitat quality. Water quality can be monitored frequently and rapidly to support adaptive management during restoration and recovery.

Water quality investigations also can supplement other studies of the Kissimmee River and its restoration (Bousquin et al. 2005), which document changes in species composition and habitat characteristics and distribution. Inclusion of water quality data in these evaluations can provide understanding of mechanisms leading to ecological change. For these reasons, water quality is an important component for assessing restoration of the Kissimmee River ecosystem.

Perhaps the most significant benefit of restoration with respect to water quality is the expected increase in dissolved oxygen concentrations. To document daily as well as seasonal variations in oxygen, additional monitoring was conducted. For this reason, dissolved oxygen has been discussed separately in the previous chapter (Colangelo and Jones 2005). The present chapter discusses baseline and reference conditions for water quality parameters other than dissolved oxygen, with special attention given to phosphorus, turbidity and total suspended solids (TSS).

Impacts of Channelization on Water Quality

Much of the initiative to restore the channelized Kissimmee River derived from concern that the C-38 flood control canal formed a conduit for rapid downstream transport of nutrients from the river's headwater basin. However, initial studies identified agricultural runoff, particularly from beef cattle ranching and dairy farming, as the primary cause of elevated phosphorus loads in the channelized system (Federico and Brezonik 1975, Florida Department of Environmental Regulation 1976, Lamonds 1975). In addition to facilitating direct entry of nutrients from surrounding agricultural lands by eliminating the system's natural capacity to filter and retain nutrients, the dredging of C-38 and ditching of lateral tributaries allowed cattle to graze along and within waterways throughout the floodplain. The most severe phosphorus runoff problems existed in the watersheds of Pools D and E. Soon after channelization, Lamonds (1975) found that the mean phosphorus concentration at S-65E (0.08 mg/L) was three times higher than at S-65, and the watersheds of Pools D and E contributed 75% of the phosphorus load originating in the river basin (excluding Lake Kissimmee). However, nitrogen concentrations did not appear to be related to intensity of agricultural land use. Nitrogen concentrations in C-38 ranged from 1.00 to 2.00 mg/L and decreased slightly between S-65 and S-65E.

Drastically modified flow patterns and channel morphometry also contributed to poorer water quality in the canal and remnant runs. The canal's water column was often stratified in summer, and water near the canal bottom was lower in dissolved oxygen and higher in phosphorus and ammonium-nitrogen (Lamonds 1975). These lower oxygen concentrations probably resulted from greater oxygen demand, anaerobic inputs from groundwater and tributaries, lack of consistent flow, and the canal's deep, box-cut shape. Stagnant conditions in remnant runs led to growth of aquatic vegetation and accumulation of organic sediment, which contributed to poor benthic habitat and chronic anoxic conditions. Specific conductance doubled from the 1950s to the early 1970s due to unusually low discharges, diversion of Lake Istokpoga outflow away from the river, and an increase in groundwater contributions (Lamonds 1975). From measurements of phytoplankton chlorophyll *a*, Rutter et al. (1989) concluded that the channelized system functioned more like a eutrophic reservoir than a natural river. Following periods of discharge through the system, phytoplankton responded to the influx of nutrients and formed blooms after water control structures were closed.

Objectives

The objectives of this chapter are to:

(1) Establish baseline (channelized) and reference (pre-channelized) conditions for assessing the effects of restored flow on water quality;
(2) Compare remnant river channel and C-38 water quality in Pools A and C during the three-year period before backfilling began in June 1999; and
(3) Quantify phosphorus concentrations and loads throughout C-38, including temporal trends and seasonal differences over a 26-year record.

METHODS

Baseline Conditions

Comparison of Water Quality in Pools A and C

Water quality data were compared from seven remnant river runs and two C-38 stations in Pools A and C for the period of March 18, 1996 to June 8, 1999. One station was sampled in each remnant run (Figure 5-1). Sampling of Rattlesnake Hammock Run (KREA 91) and Schoolhouse Run (KREA 92) in Pool A, and Oxbow 13 (KREA 93), Micco Bluff Run (KREA 94), and MacArthur Run (KREA 95) in Pool C began in March 1996. Ice Cream Slough Run (KREA 97) in Pool A was added to the monitoring program in November 1996, and Montsdeoca Run (KREA 98) in Pool C was added in December 1997. Data from C-38, sampled at S-65A and S-65C, were included to evaluate the quality of water entering and leaving the area to be restored. These data also were compared to remnant run data to determine the degree of similarity.

Samples from the remnant river runs were collected from a boat in mid-channel usually once per month. Sampling of C-38 was done on the upstream side of the water control structures. The canal samples were usually collected every two weeks, but the frequency ranged from weekly to bi-monthly.

At each station, water was sampled at a depth of zero to 0.5 m with a plastic bucket or Van Dorn bottle. Subsamples were transferred to polyethylene bottles. Unfiltered subsamples were analyzed for turbidity, TSS, chlorophyll *a*, total organic carbon (TOC), total phosphorus (TP), total Kjeldahl nitrogen (composed of organic nitrogen and ammonium), and alkalinity. Subsamples for color, dissolved organic carbon (DOC), soluble reactive phosphorus (SRP), nitrate- and nitrite-nitrogen, ammonium-nitrogen, and chloride were filtered through a 0.45 micron, polycarbonate membrane filter. Subsamples for TP, TOC, DOC, and nitrogen analyses were preserved immediately with 50% sulfuric acid to pH <2. Samples were transported on ice and refrigerated until analysis. Specific conductance and pH were measured *in situ* at a depth of 0.5 m with a Hydrolab®. Sampling and analytical methods are described in detail in the South Florida Water Management District (SFWMD) Comprehensive Quality Assurance Plan (SFWMD 1999).

Water quality data from certain lateral tributaries also are included in this report when needed to interpret water quality data from the remnant runs. These tributary stations include Rattlesnake Slough (KREA 89), which flows into Rattlesnake Hammock Run, two unnamed tributaries (KREA 99 and KREA 100) flowing into Montsdeoca Run, and Starvation Slough (KREA 83) and Oak Creek (KREA 96), which flow into Micco Bluff Run. These stations were visited monthly, but grab samples were collected only when flow was observed. The sampling method was the same as described above.

Data values below the detection limit of the analytical method were set to half the detection limit before performing statistical analysis. Because data from most stations were not normally distributed, significant differences ($p <0.05$) between stations and pools were tested with the nonparametric Kruskal-Wallis test and Dunn's test using SAS, v.8 (SAS Institute, Cary, NC). Linear regressions of total phosphorus concentrations versus time were performed after log-transforming the data.

Only three years of baseline data are available for the river runs, but the water quality record for C-38 extends back to June 1973. The Kruskal-Wallis test was used to compare these earlier data to the 1996–1999 data set to determine if the latter data were significantly different from the longer period of record.

C-38 Phosphorus Concentrations and Loads

Further analysis of phosphorus data was based on grab samples from the six C-38 structures (S-65 to S-65E) since 1973 and from Lake Kissimmee since 1982. Monthly mean concentrations were calculated for months when multiple samples were collected. Summary statistics (e.g., annual and seasonal means) were then calculated from the monthly mean values.

Computation of total phosphorus loads involved grab samples supplemented by automatic sampling. Grab samples were only included in loading calculations if the structures had been open at the time of sample collection (i.e., there was flow through the pool). Samples also were taken with automatic samplers from the upper half of the water column on the upstream side of the structures. The autosampler took 80 ml water samples every 144 minutes (ten samples per day) and composited them in a pre-acidified, 1 liter, polyethylene bottle over a 24-hour period. These unrefrigerated, daily-composite samples were transported weekly to the lab for analysis. If two or more samples (grab or automated) were collected on a given day, the data were averaged to yield a daily mean concentration.

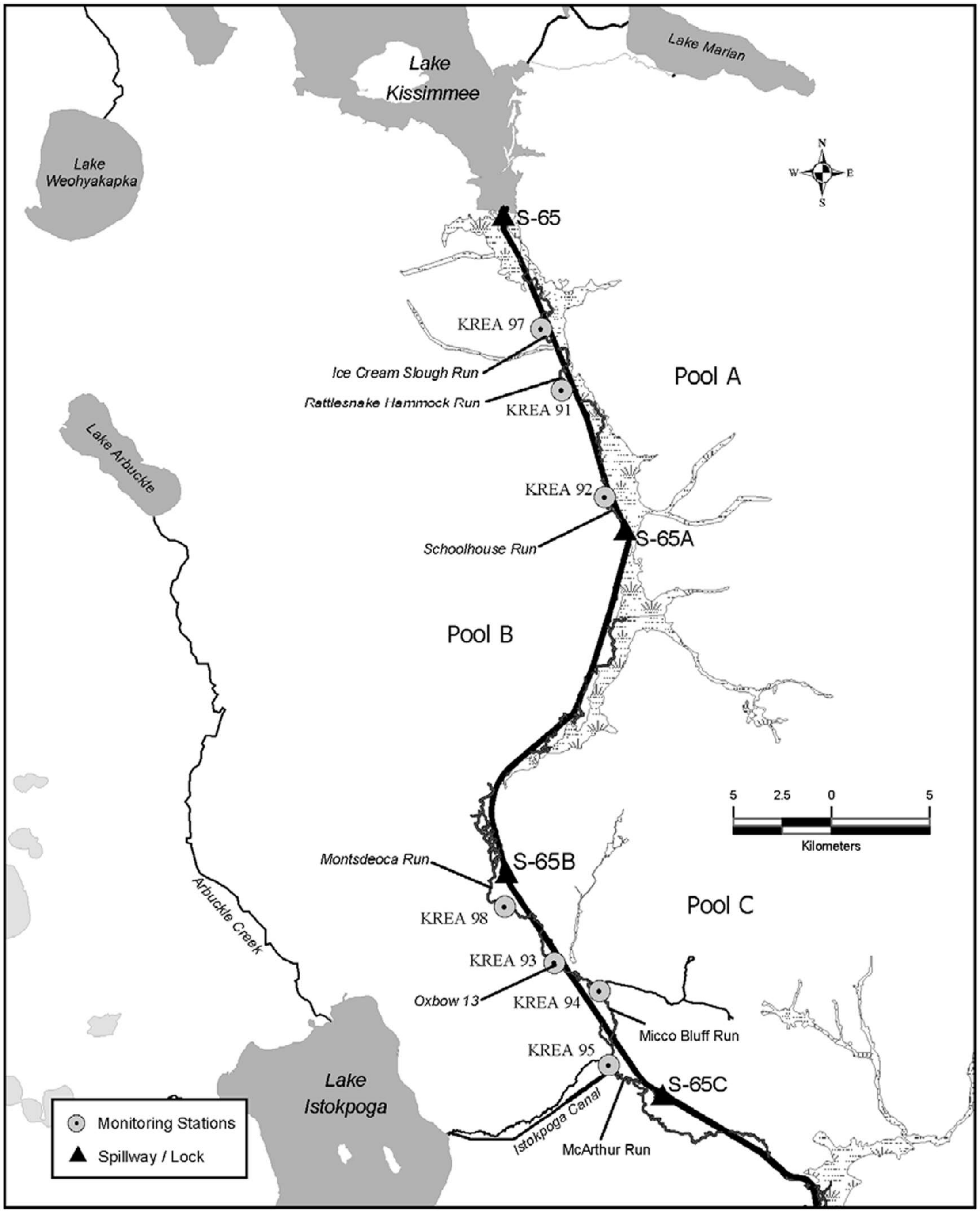

Figure 5-1. Locations of grab sample water quality monitoring stations (KREA91–KREA98) in Pools A and C.

Multiple samples collected the same day included grab samples collected on both upstream and downstream sides of the structures during earlier years.

After the TP concentration data set was established, concentrations for days between adjacent sampling dates were estimated by interpolation to provide estimated or measured TP concentrations for each day. If automated composite samples were collected between grab sample dates, measured TP values were available for each day of sampler operation and interpolation was not necessary.

Total phosphorus loads were calculated by multiplying daily TP concentration by daily discharge at each structure. Discharges were estimated by the U. S. Geological Survey (USGS) (S-65, S-65E) and SFWMD (S-65A, S-65B, S-65C, S-65D). Daily loads were summed by month and by year. To facilitate evaluation of long-term trends, discharge-weighted (D-W) TP concentrations were calculated by dividing annual TP load by annual discharge.

Reference Conditions

Little information is available on the quality of water in the pre-channelized Kissimmee River. The earliest description (Love 1955) of water quality in the pre-channelized ecosystem (1940–1941) characterized the river's water as soft (hardness ranging from 17 to 26 mg/L), highly colored (110 units on the platinum-cobalt scale), and low in total dissolved solids (61 to 80 mg/L). Daily monitoring at the SR 70 bridge west of the town of Okeechobee showed little variation in concentrations of dissolved constituents, indicating no or limited impact from surface runoff, although flow during this period (1940–1941) was only moderate (~1000–3000 cfs). However, runoff of particulates from the watershed may have been insignificant even during extreme flooding. The river was fed by upper basin lakes that overflowed through wide, shallow marshes during periods of heavy rainfall. Floods were characterized by slow changes in stage, low flow velocities, and long periods of recession. Floodwaters were relatively clear and little silt was left after floods passed (Bogart and Ferguson 1955). In addition to headwater flow from Lake Kissimmee, which supplied 58% of total river discharge (Bogart and Ferguson 1955), river flow was maintained by groundwater seepage from aquifers underlying upland areas (Parker 1955). Surface runoff was inhibited by flatness of the terrain, abundant vegetation, and permeable sandy soil. Consequently, the Kissimmee River carried relatively clear water, although it was capable of moving considerable quantities of loose sand during seasonal floods (Parker 1955). These characteristics suggest that watershed runoff did not carry much suspended material and did not significantly influence turbidity in the pre-channelized river. Any turbidity present would have been due to internal factors such as plankton, suspended detritus, or sediment erosion during extreme discharges. However, in a flowing, blackwater river surrounded by dense vegetation, phytoplankton blooms were probably rare.

Turbidity

Among the water quality parameters discussed in this chapter, a restoration expectation has been developed only for turbidity and TSS. Because no turbidity and TSS data exist from the pre-channelized Kissimmee River, reference conditions have been derived from similar streams in south Florida (Table 5-1). Data for these eight free-flowing, blackwater streams (Table 5-2) came from the SFWMD's database and were collected and analyzed in a manner similar to the baseline data. As described by Colangelo and Jones (2005), these streams and their watersheds share some features of the former Kissimmee River (e.g., low topographic relief, sandy substrate, presence of swamps or marshes, low velocity), although other characteristics such as watershed size, discharge, watershed development, and artificial drainage may differ.

Phosphorus Loading

Reference data are also lacking for phosphorus, the other parameter of primary interest. The only phosphorus data from the pre-channelized river are from three samples collected in 1952 (Odum 1953). One sample was taken on August 19, 1952 at a location believed to be the USGS gauge located south of SR 60 in what is now Pool A. The TP concentration in this sample was 0.060 mg/L. As documented later in this chapter, this concentration is similar to concentrations in Lake Kissimmee and C-38 (Pool A) after channelization. Sample concentrations collected two months earlier at two downstream stations were much lower. A sample collected on June 16, 1952 at Ft. Basinger, in what is now Pool D, had a concentration of 0.002 mg/L. Another sample collected on June 22, 1952 at SR 78 near the river's mouth had a concentration of 0.012 mg/L. Discharge on that date averaged 1,450 cfs (Joyner 1974), which is close to the mean daily discharge (2,231 cfs) for 1929–1960. Multiplying 0.012 mg/L by the mean daily discharge yields a mean loading rate of 65 kg/day or 24 metric tons per year (Mt/y). These data suggest that TP concentrations and loads were very low in the lower part of the river compared to values measured decades later in Pools D and E.[1]

Table 5-1. South Florida Water Management District data sets for Florida streams used as reference sites for turbidity and total suspended solids.

Water Body	SFWMD Station ID	County	Period of Record (month/year)	Frequency
Fisheating Creek	FECSR78	Glades	4/73–2/99	Weekly - Monthly
Arbuckle Creek	ARBKSR98	Highlands	2/88–2/99	Bi-Monthly
Lake Marian Creek	DLMARNCR	Polk	4/82–9/85	Monthly
Reedy Creek	CREEDYBR	Osceola	4/85–3/99	Monthly
Tiger Creek	ETIGERCR	Polk	4/82–6/85	Monthly
Josephine Creek	JOSNCR17	Highlands	2/88–2/99	Monthly - Bi-Monthly
Boggy Creek	ABOGG	Osceola	8/81–3/99	Monthly
Catfish Cr.-S. Branch	ROSALIEC	Polk	11/84–9/85	Monthly

Table 5-2. Summary of turbidity and total suspended solids data for Florida stream reference sites.

	Turbidity (NTU)				TSS (mg/L)[1]		
Water Body	N	Median	Mean ± Std. Error	Max.	N	Median	Max.
Fisheating Creek	393	1.6	3.8 ± 0.9	290.0	365	<3.0	986.7
Arbuckle Creek	85	2.9	3.4 ± 0.2	14.4	39	<3.0	24.0
Lake Marian Creek	37	2.0	4.5 ± 1.9	70.0	13	4.0	15.0
Reedy Creek	150	1.3	2.0 ± 0.2	18.9	99	<3.0	58.0
Tiger Creek	33	3.9	3.9 ± 0.3	8.7	12	3.0	8.0
Josephine Creek	85	2.2	2.4 ± 0.2	10.5	39	<3.0	14.0
Boggy Creek	204	2.0	6.5 ± 2.8	570.0	116	<3.0	416.0
Catfish Cr.-S. Branch	11	3.8	4.8 ± 0.8	11.1	4	4.5	11.0

1 = Most total suspended solids values were below detection limit (usually <3.0 mg/L). Consequently, means and standard errors for TSS are not shown.

RESULTS

Results are presented in two parts. The first part focuses on data collected during a period of approximately three years before backfilling began in June 1999. Water quality is compared between remnant river channel and C-38 sampling stations in Pools A and C. The second part of the results examines phosphorus concentrations and loads throughout C-38, including temporal trends and seasonal differences over a 26-year record.

[1] The Kissimmee River phosphorus data from 1952 should be treated with caution. Total phosphorus concentrations reported in the range of 0.002-0.012 mg/L may have been below the minimum detection unit of analytical methods commonly used over 50 years ago.

General Water Quality of Pools A and C

Appendix 5-1A presents descriptive statistics for data collected at each station. Because data for most parameters were not normally distributed, median values are emphasized in the text below.

Turbidity and Total Suspended Solids

Turbidity was usually below 10 NTU in all remnant runs (Appendix 5-2A). There was a slight tendency for higher values in summer, which probably reflect higher densities of phytoplankton. Median turbidity values were 1.2 to 2.5 NTU. Although variation was small, turbidity values in Pool A runs were significantly higher than in Pool C runs (Kruskal-Wallis test; $p < 0.05$). As indicated below in the Planktonic Chlorophyll *a* section, this difference may have been related to more phytoplankton in Pool A runs, which could be mainly attributed to brief algal blooms. Concentrations of TSS in remnant channels were 25 mg/L or less, and were usually below the detection limit of 3 mg/L.

Similar levels of turbidity and TSS were measured in C-38. However, turbidity was significantly higher (Kruskal-Wallis test; $p < 0.05$) at S-65A (median 3.0 NTU) than at S-65C (median 2.0 NTU) (Appendix 5-3A). Higher phytoplankton densities were probably responsible for most instances of higher turbidity at S-65A. The unusually high turbidity value (87 NTU) in May 1998 (Appendix 5-3A) coincided with the dense algal bloom observed at this station (Appendix 5-4A).

Compared to 1973–1996, turbidity in 1996–1999 was not significantly different at S-65C, but was slightly higher at S-65A (Kruskal-Wallis test; $p < 0.05$). Total suspended solids concentrations at the two structures did not differ significantly between the two time periods (Appendix 5-5A).

Judging from a comparison of these baseline data with data from the eight reference streams, channelization has not increased turbidity and TSS concentrations significantly. Aside from occasional algal blooms that raised turbidity under stagnant conditions, suspended particulates in the channelized river system remained low. Median turbidity and TSS values in these reference streams also were low (1.3–3.9 NTU and <3.0–4.5 mg/L, respectively; Table 5-2). These values were probably typical of the former Kissimmee River due to the characteristics of the river and its watershed mentioned earlier. Therefore, because both baseline values and comparable reference values have been low, turbidity and TSS are not expected to change significantly after restoration. As stated in the expectation compendium (Jones 2005), mean turbidity in the restored river channel should not differ significantly from overall mean turbidity in the eight reference streams (3.9 NTU), and median TSS concentration should not exceed 3 mg/L.

Planktonic Chlorophyll a

Concentrations of planktonic chlorophyll *a* were usually low in all remnant runs, but occasionally rose above 40 mg/m^3 (Appendix 5-6A), indicating the presence of algal blooms. Blooms were found in six of the seven runs. Median chlorophyll *a* values ranged from 3.3 mg/m^3 in Montsdeoca Run to 17.3 mg/m^3 in Ice Cream Slough Run (Appendix 5-1A).

Chlorophyll *a* concentrations were significantly greater in Pool A runs than in Pool C runs (Kruskal-Wallis test; $p < 0.05$). Also, chlorophyll *a* was significantly higher in Oxbow 13 and Ice Cream Slough Run compared to the other runs in their respective pools (Dunn's test; $p < 0.05$). Concentrations in Oxbow 13 tended to be slightly higher than in the other Pool C runs toward the end of the three-year period (Appendix 5-6A). In Ice Cream Slough Run, the difference was due to a bloom in which chlorophyll *a* reached a maximum of 120.7 mg/m^3 on June 24, 1998 (Appendix 5-6A). This bloom persisted in Ice Cream Slough Run until October. Rattlesnake Hammock Run also had a bloom in June 1998.

The June 1998 bloom in the remnant runs was preceded by a major bloom in C-38, where chlorophyll *a* exceeded 300 mg/m^3 at S-65A on May 12, 1998 (Appendix 5-4A). The spring bloom in Pool A was followed by a bloom in Pool C, where chlorophyll *a* exceeded 105.9 mg/m^3 at S-65C on July 7, 1998 and 75.6 mg/m^3 on July 23, 1998 in Micco Bluff Run.

Chlorophyll *a* concentrations in C-38 (Appendix 5-4A) were usually similar to those in the remnant runs. Except for the bloom period of 1998, they remained at ≤ 50 mg/m^3. The median concentration at S-65A (12.0 mg/m^3) was 50% higher (Kruskal-Wallis test; $p < 0.05$) than the concentration at S-65C (8.0 mg/m^3).

Color and Organic Carbon

Color was variable in remnant runs (Appendix 5-7A) and C-38 (Appendix 5-8A), ranging from 30 to 561 Pt-Co units. Rattlesnake Hammock Run had significantly (Dunn's test; $p < 0.05$) more color (median = 151 Pt-Co units) than the other two Pool A runs. In Pool C, Micco Bluff Run was significantly higher in color (median = 142 Pt-Co units) than MacArthur and Montsdeoca Runs. Rattlesnake Hammock Run received inflow that tended to be higher in color. Color at the outlet of Rattlesnake Slough (sampled three times in 1996) was 166–191 Pt-Co units. Micco Bluff Run received inflow from Starvation Slough and Oak Creek, which had median color values of 195–255 Pt-Co units (July 1996–August 1998 data). Other remnant runs, such as Montsdeoca Run, received tributary inflow that was lower in color and may have originated partially from pumped groundwater (B. Jones, SFWMD, personal observation).

Color values at the two C-38 stations, while similar to those in the remnant runs, followed a seasonal pattern. The highest color occurred in the wet season (summer of 1996, 1997, and 1998) when S-65 was closed but S-65C was open to pass local inflows (Appendix 5-8A). Color was more moderate during the dry season when discharges were either low or were predominantly from Lake Kissimmee.

Color was a good indicator of organic carbon concentrations. Correlations between color and TOC were significant at every station (r^2 = 0.41 to 0.92; $p < 0.05$). Consequently, TOC displayed the same temporal patterns, and differences and similarities between stations (Appendix 5-9A and Appendix 5-10A). Median TOC concentrations in remnant runs and C-38 ranged from 17.0 to 22.8 mg/L. Nearly all (>92%) organic carbon was in the dissolved form.

Color and organic carbon did not differ significantly between Pool A and Pool C remnant runs. These parameters also did not change significantly at S-65A and S-65C between 1973–1996 and 1996–1999 (Appendix 5-5A) (Kruskal-Wallis test; $p < 0.05$).

Phosphorus

With few exceptions, TP concentrations were moderate at all stations. Concentrations were significantly higher in Pool A runs than Pool C runs (Kruskal-Wallis test; $p < 0.05$), but this difference was due to lower concentrations in Montsdeoca Run. Median concentrations in river runs ranged from 0.034 mg/L in Montsdeoca Run to 0.071 mg/L in Micco Bluff Run, and medians for S-65A and S-65C were 0.067 and 0.056 mg/L, respectively.

In Pool A, Ice Cream Slough Run and Schoolhouse Run had significantly higher concentrations than Rattlesnake Hammock Run (Dunn's test; $p < 0.05$), and greater values were measured toward the end of the baseline sampling period (Appendix 5-11A). Concentrations in Rattlesnake Hammock Run may have been lower due to inflow from Rattlesnake Slough. Although water quality monitoring at the slough's outlet (KREA 89) ended in August 1996, 112 samples taken from 1986 to 1996 had TP concentrations (median = 0.041 mg/L, mean = 0.046 mg/L, range = 0.006–0.179 mg/L) similar to those measured in this run during 1996–1999.

Total phosphorus in Micco Bluff Run was significantly higher than in the other three runs in Pool C (Dunn's test; $p < 0.05$). In June 1997, TP in this run increased to more than 0.4 mg/L (Appendix 5-11A) following a week of precipitation totaling more than 7 cm. Samples taken on the same date from its two tributaries, Oak Creek and Starvation Slough, also were high in phosphorus (Appendix 5-12A). Although only a few samples were collected from these tributaries during 1996–1999, they indicate the potential for these tributaries to impact phosphorus concentrations in Micco Bluff Run.

Total P concentrations were essentially similar at S-65A and S-65C (Appendix 5-13A), even though they were statistically higher at S-65A (Kruskal-Wallis test; $p < 0.05$). In May 1998, higher TP at S-65A coincided with a spring algal bloom. A second, higher peak in August 1998 coincided with higher turbidity (but low chlorophyll *a*) that might have originated from suspension of lake sediment near S-65. Apparently, this turbidity plume drifted down to S-65C, where a smaller peak was measured the following month. Another phosphorus/turbidity spike appeared at S-65A in December 1998.

On average, concentrations of soluble reactive phosphorus were less than half the total P. However, except for the very high TP value at S-65A in August 1998, SRP was proportionately higher whenever TP concentrations rose. For example, when TP peaked in Micco Bluff Run in 1997 and Schoolhouse Run in 1998, SRP accounted for 77–79% of the total phosphorus. This percentage decreased as TP declined to background levels. The same relationship was present in C-38. Between August 1998 and March 1999, when phosphorus concentrations were slightly elevated in the canal, SRP made up approximately 60–90% of TP.

Total phosphorus and SRP at S-65A and S-65C were significantly higher (Kruskal-Wallis test; $p < 0.05$) in 1996–1999 than in 1973–1996 (Appendix 5-5A). Concentrations at these monitoring stations followed the trend at the Lake Kissimmee outflow and increased after 1994 (See Phosphorus Concentrations and Loads below).

Nitrogen

Total nitrogen (TN) concentrations, calculated from total Kjeldahl nitrogen and nitrate/nitrite analyses, ranged from 0.50 to 2.34 mg/L in the remnant runs and showed no temporal trends or significant differences between stations or pools during the 1996–1999 baseline period (Appendix 5-14A). Median concentrations among monitoring stations ranged from 1.12 to 1.33 mg/L over these three years. With the exception of one data point (a value of 5 mg/L coinciding with the dense algal bloom at S-65A in 1998), the range of concentrations in C-38 was similar (Appendix 5-15A), and the median TN concentration was 1.13 mg/L at both S-65A and S-65C.

The dissolved inorganic nitrogen (DIN) fraction (nitrate, nitrite, and ammonium ions) comprised 2–14% of the total nitrogen and was significantly greater in Pool C runs than in Pool A runs (Kruskal-Wallis test; $p < 0.05$). This difference was due to concentrations in Montsdeoca and MacArthur Runs, which were significantly higher than concentrations in other runs of Pool C (Dunn's test; $p < 0.05$). Montsdeoca Run received occasional discharge from Pool B (through Culvert S-65BX2 in the S-65B tieback levee) and two ditches draining a nearby pasture and citrus grove. Median DIN concentrations in samples collected from these ditches in 2001 were 1.62–1.96 mg/L, indicating that they probably were the source of higher DIN concentrations (mostly nitrate) in Montsdeoca Run. The citrus grove was irrigated with groundwater from the Floridan Aquifer, and excess water was apparently released to the run. However, 11 samples taken from a nearby Floridan Aquifer well (OKF-42) at S-65C had lower DIN concentrations (median = 0.41 mg N/L; 99% ammonium-N) than the ditches (Florida Department of Environmental Protection, 2000), thus pointing to the influence of agricultural land use on these ditch concentrations. Groundwater seepage directly into Montsdeoca Run also has been investigated as a potential source of nitrogen, but Belanger et al. (2001) have found that mean nitrate/nitrite concentrations in shallow groundwater wells (0.023–0.075 mg N/L) on the channel bank were lower than in the channel (0.463 mg N/L). MacArthur Run is another run that occasionally received substantial citrus and pasture drainage via its main tributary, the Istokpoga Canal. However, no water quality data from the canal and its inflows are available to determine why MacArthur Run was higher in DIN.

Dissolved inorganic nitrogen concentrations in C-38 were slightly higher than in most remnant runs and tended to be lowest in the spring and summer (Appendix 5-16A). This seasonal pattern also may have been present in remnant runs (Appendix 5-17A). Compared to 1973–1996, significantly higher DIN concentrations were present at S-65A and S-65C during 1996–1999 (Appendix 5-5A), but total nitrogen did not differ significantly between the two periods (Kruskal-Wallis test; $p < 0.05$).

Other parameters

Significant differences in specific conductance, chloride, and alkalinity existed between Pool A and Pool C runs due to particular conditions in certain runs. Specific conductance and chloride were higher in Pool C runs, while alkalinity was higher in Pool A runs (Kruskal-Wallis test; $p < 0.05$).

Median values of specific conductance at river channel stations ranged from 110 to 227 μS/cm, which can be classified as moderate for a Florida freshwater system. Ice Cream Slough Run, Montsdeoca Run, and MacArthur Run had significantly higher values (Dunn's test; $p < 0.05$) than the other runs in their respective pools (Appendix 5-18A). Specific conductance in Montsdeoca and MacArthur Runs appears to be related to chloride concentrations (Appendix 5-19A), indicating the importance of agricultural inputs to these runs. Median chloride concentrations in two ditches feeding Montsdeoca Run were over 80–100 mg/L (SFWMD unpublished data, 2001), about twice the level in Montsdeoca Run (42.2 mg/L). The higher DIN concentrations in Montsdeoca Run also appear to be somewhat related to specific conductance. Higher values of specific conductance in Ice Cream Slough Run were related to alkalinity (Appendix 5-20A) instead of chloride, suggesting a different source, possibly groundwater-related. Median alkalinity (61.9 mg $CaCO_3$/L) in Ice Cream Slough Run was nearly twice the median at any other station.

Specific conductance, chloride, and alkalinity values in the other four runs did not vary as much and resembled levels found in C-38 (Appendix 5-21A, Appendix 5-22A, and Appendix 5-23A). As with color and organic carbon data, some slight seasonal variations (e.g., specific conductance in C-38), may indicate wet-dry season cycles.

Values of pH (range = 4.80–8.15) also varied seasonally (Appendix 5-24A and Appendix 5-25A). Lowest values were measured at S-65A and S-65C, but these two stations had significantly higher mean values (~ 6.8) than the remnant runs (5.95–6.47). Lower mean values in the remnant runs may reflect more influence of tributary inputs and surrounding vegetation and soils. More neutral values in C-38 are probably due to higher pH in Lake Kissimmee. Mean pH at S-65 was 7.33 during the 1996–1999 period.

Compared to 1973–1996, specific conductance, chloride, and alkalinity were significantly lower at S-65A and S-65C in 1996–1999, and pH was higher at S-65A (Kruskal-Wallis test; $p < 0.05$) (Appendix 5-5A).

Phosphorus Concentrations and Loads

Total Phosphorus Concentrations in C-38 and Lake Kissimmee

Long-term trends in monthly mean TP concentrations at S-65, S-65A, S-65B, and S-65C generally reflect the trend in Lake Kissimmee (Appendix 5-26A and Appendix 5-27A). However, concentrations at these structures, particularly at S-65, were sometimes higher than in the lake. Linear regressions on the log-transformed data indicated significant increases in TP from June 1973 to May 1999. While concentrations at these four structures were generally similar, S-65 exhibited the greatest increase ($r^2 = 0.20$; $p < 0.01$), and the slope of the upward trend decreased progressively downstream. Most of the increase occurred after 1994.

Because C-38 transports phosphorus loads from Lake Kissimmee, an understanding of factors affecting lake phosphorus concentrations is important for assessing past and future loading from C-38. Until the 1980s, effluents from wastewater treatment plants raised TP concentrations in Lake Tohopekaliga, Lake Cypress, Lake Hatchineha, and the northern portion of Lake Kissimmee (Williams 2001). The influence on Lake Kissimmee during this time is indicated by the difference in TP concentrations between the lake's northern monitoring station (E02) and central station (E04) (Appendix 5-26A). As these effluents were diverted away from the lakes, TP concentrations declined in Lake Tohopekaliga and the lakes downstream (James et al. 1993, 1994, Williams 2001). Later, hydrilla expanded across these lakes. Hydrilla was first reported in Lake Kissimmee in 1977 (Williams 1990), but coverage remained below 20 percent until 1991. Coverage then increased each year until 1995, when it reached 52% (SFWMD et al. 1997). Management efforts were confined to controlling only the heaviest hydrilla growth until 1997, when lake-wide treatment with fluridone herbicide became possible. By October 1998, hydrilla occupied only 19% of Lake Kissimmee (Florida DEP Bureau of Invasive Plant Management, unpublished data). Total phosphorus, chlorophyll *a*, and turbidity in Lake Kissimmee were relatively low during the period of hydrilla expansion, but increased in the year following lake-wide treatment (Appendix 5-28A). The mean lake TP concentration in 1998 was 0.062 mg/L, which was its highest since 1987.

Total phosphorus concentrations at S-65 have followed the upward trend in Lake Kissimmee, but have been frequently higher than mid-lake concentrations in recent years. From 1989–1999, the highest TP concentrations at S-65 (~ 0.15–0.50 mg/L) have occurred in 1989, 1992, 1996, and 1998 (Appendix 5-26A). The 1989 and 1992 phosphorus pulses traveled downstream and were detectable at S-65A, S-65B, and S-65C (Appendix 5-26A and Appendix 5-27A). Monthly mean TP (0.175–0.463 mg/L) and SRP (0.099–0.348 mg/L) at S-65 were exceptionally high in samples taken from June to August 1992. The magnitude and duration of this event suggests discharge of a concentrated source of phosphorus to the south end of Lake Kissimmee. In 1996, high TP values coincided with dredging, vegetation removal, and low water levels during a managed drawdown of the lake. Because discharges were low during the time of greatest impact, concentrations downstream of S-65 were not greatly affected. In 1998, higher TP concentrations at S-65 might have been due to local factors that will be mentioned in the following Phosphorus Loads section.

While phosphorus concentrations and loads at S-65A, S-65B, and S-65C depended largely on concentrations in Lake Kissimmee, especially at higher discharges, local influences on concentrations at S-65 could have affected loading calculations at this structure at certain times. Higher TP values could also occur occasionally during low- or no-flow conditions due to algal blooms in C-38, but in general, TP concentrations at these structures were moderate in comparison to concentrations at S-65D and S-65E (Appendix 5-27A). Over a 25-year period (1974–1998), mean annual TP concentrations at the four upper structures were similar to each other (0.051–0.055 mg/L), despite higher concentrations at S-65 in the late 1990s (Appendix 5-29A). Mean monthly TP values (Appendix 5-30A) appear to show that concentrations increased in the summer, especially at S-65D and S-65E. Concentrations at S-65 and S-65A also appear to increase. However, if 1992 and 1996 TP values are excluded, average concentrations at S-65 and S-65A become lower relative to S-65B and S-65C, and the resulting plots more closely resemble plots of median monthly TP values (Appendix 5-30A). Consequently, to avoid the bias of years with unusually high values, seasonal means were calculated from the median monthly values. Resulting wet season means (June–November) were higher than dry season means (December–May), and these differences were greater at the lower structures (Appendix 5-31A).

Phosphorus Loads

Mean monthly discharges and phosphorus loads in C-38 followed a distinct seasonal pattern (Appendix 5-32A), reflecting the schedule of regulatory releases from Lake Kissimmee, as well as rainfall-runoff during the wet season. Over 60 percent of discharges and loads from S-65 occurred during the dry season, when the Kissimmee lakes were lowered in preparation for rains in summer and fall. In the wet season, a higher proportion of discharge originated from the C-38 basin (Appendix 5-33A).

Annual phosphorus loads (Appendix 5-34A) at S-65 to S-65C rose during 1974–1998 due to increases in both discharges (Appendix 5-35A) and concentrations (Appendix 5-26A, Appendix 5-27A, and Appendix 5-29A). During the last three years, mean annual discharge from S-65 was almost 50 percent greater than in 1974–1995, and mean annual phosphorus loading increased by more than 360 percent. Discharge-weighted TP concentrations (Appendix 5-36A) rose from 0.043 mg/L (1974–1995) to 0.104 mg/L (1996–1998). Although D-W TP concentrations at S-65D and S-65E were higher in most years (Appendix 5-36A), they did not increase in 1996–1998. Consequently, loading from the upper basin became proportionately greater. From 1974 to 1995, discharge from S-65 accounted for 69% of discharge from S-65E, but only 30% of the phosphorus load. In 1996–1998, S-65 contributed 76% of the discharge and 68% of the phosphorus load from S-65E (Appendix 5-37A). In the latter period, phosphorus loads from S-65 exceeded loads from S-65A, S-65B, and S-65C due to higher concentrations at the headwater structure and possible underestimation of discharge at the downstream structures, particularly at S-65A during heavy flow in 1997–1998.

The high discharge in 1997–1998 resulted from an unusual succession of storms caused by the El Nino climatic phenomenon. Large releases were made through S-65 beginning in late November 1997. Discharges climbed to over 8000 cfs in December and January and over 10,000 cfs in February and March (Appendix 5-38A). Consequently, the river's headwater contributed most of S-65E's discharge (78%) and phosphorus loading (54%) in 1998. Nearly all (98%) of the 1998 loading from S-65 that year occurred in the first four months. Total phosphorus concentrations at S-65 were above average during this time and were higher than concentrations in Lake Kissimmee, which were between 0.04 and 0.05 mg/L (Appendix 5-38A). Concentrations in Lake Hatchineha, Lake Cypress, Reedy Creek, and Lake Tohopekaliga were similar to the values in Lake Kissimmee. Therefore, the upper basin was not the source of higher concentrations at S-65. Instead, local factors near S-65 appear to have affected phosphorus concentrations during this period. These factors may have included agricultural runoff, wind-induced suspension of sediment from a clear lake bottom formerly covered with weeds, construction activities or weed accumulation at S-65, or mechanical weed harvesting. Because these few months during 1997–1998 and the three-year period of 1996–1998 were unusual with respect to these local factors, the higher concentrations and loads measured during this time may not signify a lasting trend. In fact, more recent data (1999–2003) collected by the SFWMD indicate that annual mean TP concentrations at S-65 are closer to the long-term average (B. Jones, SFWMD, unpublished data). Consequently, the baseline condition for C-38 phosphorus loads was established using data from prior years (1974–1995).

During 1974–1995, phosphorus loading at S-65C and S-65D averaged 51 and 83 Mt/y, respectively. These amounts comprised 43% and 71% of the load at S-65E. Upstream of the restoration area, at S-65A, the mean loading rate was 42 Mt/y (Appendix 5-37A).[2]

[2] Discharges and phosphorus loads were estimated for the designated structures only and do not represent accurate estimates for the entire channel-floodplain system. Other significant, unmeasured discharges through the system may have occurred during this baseline period. Prior analysis conducted by the SFWMD during development of the 1997 Lake Okeechobee SWIM Plan Update (SFWMD 1997)

Because phosphorus loads can vary greatly from year to year, discharge-weighted concentrations (annual load divided by annual discharge) provide a more useful metric for evaluating effects of restoration. Annual mean D-W TP concentrations were 0.053 mg/L at S-65C (range = 0.033–0.087 mg/L) and 0.078 mg/L at S-65D (range = 0.047–0.141 mg/L) (Appendix 5-39A). Concentrations were greater during years of lowest flow (1981 and 1985).

DISCUSSION

1996–1999 Baseline Comparisons

Many statistically significant differences in water quality between remnant river runs were small or ecologically unimportant (e.g., turbidity), or due to transient events such as algal blooms (e.g., chlorophyll *a*). However, some of these differences demonstrate the impact of tributary runoff in the channelized system. For example, phosphorus concentrations in Micco Bluff Run were apparently elevated by pasture runoff from its tributaries. Although land use has not affected water quality in Pool C as severely as pasture and dairy runoff has in Pools D and E, evidence suggests that direct runoff from ditched tributaries has had measurable impacts on water quality in Pool C river channels.

Conversely, marsh inflow might have regulated phosphorus concentrations in Rattlesnake Hammock Run. The moderate TP concentrations in the outflow from Rattlesnake Hammock Marsh may be illustrative of water quality in restored wetlands, which may be capable of retaining substantial amounts of phosphorus as water moves between the channel and floodplain. Goldstein (1993) found that small wetlands could remove between 25% and 80% of the phosphorus they received. In the Everglades, marshes constructed to remove phosphorus from agricultural inflows reduced TP concentrations from 0.100 to 0.025 mg/L (SFWMD 2001). Moustafa et al. (1996) estimated that Boney Marsh had a mean annual phosphorus removal efficiency of 71%. The Boney Marsh experiment demonstrates that a significant proportion of phosphorus can be assimilated even when input concentrations are moderately low, as is typical of much of the Kissimmee River. Most studies of other constructed wetlands have reported comparable removal efficiencies (Gersberg et al. 1984, Godfrey et al. 1985, Cooper and Findlater 1990, Pride et al. 1990, Meiorin 1989, LaRock et al. 1991, Mitsch 1992, Moustafa 1999).

Backfilling of ditched tributaries with high TP concentrations and reestablishment of floodplain sloughs also should reduce phosphorus inputs to the river channel. Because most phosphorus in lateral inflows appears to be SRP, successful reductions in phosphorus inputs must occur through biological uptake on the floodplain, rather than entrapment of particulates. After restoration, phosphorus concentrations and other parameters should exhibit less variability in Pool C, as continuous flow and hydrologic interaction with the floodplain restores more consistent, high-quality water to the river. In Pool A, water quality may become more consistent as well if ditches are degraded, cattle are removed, and the canal receives continuous flow from Lake Kissimmee.

Baseline water quality in the remnant runs was similar to water quality in C-38, except for pH, which was somewhat higher in the canal. Differences within C-38 (at S-65A and S-65C) were insignificant, except for a few parameters — notably turbidity, chlorophyll *a*, TP (higher at S-65A), and DIN (higher at S-65C). However, these differences were small, and in most cases can be attributed to brief events such as algal blooms and turbidity in discharge from Lake Kissimmee. Consequently, if the quality of water improves significantly as it flows through the restored area, this change should be evident by comparing pre- and post-restoration data at the upstream (S-65A) and downstream (S-65C) stations. Likewise, the overall similarity of water quality in the Pool A and Pool C remnant runs allow the Pool A runs to serve as acceptable controls for comparison with the Pool C runs in the post-restoration evaluation.

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identified discrepancies in the water budgets of individual pools that were attributed to undocumented discharges through flanking structures along the pool tieback levees (Joseph Albers, SFWMD, personal communication). Consequently, further examination of operational records and analysis of hydrologic data would be needed to accurately estimate baseline discharges and loads from each pool. In addition, better understanding of channel-floodplain hydrology is needed for future evaluation of post-restoration phosphorus loads, which should take these unmeasured baseline discharges into account when doing pre- and post-restoration comparisons.

C-38 Phosphorus Concentrations and Loads

After declining in the 1980s and early 1990s, TP concentrations in Lake Kissimmee exhibited a small increase, which was most noticeable at S-65. Concentrations in C-38 followed the headwater trend. Possible factors contributing to these higher concentrations include: lake management activities (organic sediment and tussock removal, and dredging); response of Lake Kissimmee to artificial drawdown and hydrilla control (sediment phosphorus release and return to a plankton-dominated system); wind-induced sediment resuspension in the lake; local impacts at S-65 (weed accumulation and SR 60 bridge construction); or inputs of phosphorus near the lake's outlet. It is significant to note, however, that these potential causes did not always increase TP loading downstream, because S-65 was frequently closed when concentrations were greatest.

Elevated TP concentrations at S-65, coupled with very high discharges during a succession of storms, resulted in disproportionately large phosphorus loading from S-65 in 1998. This raised concern that urban development in Orange and Osceola Counties had accelerated eutrophication of the Kissimmee Chain of Lakes. However, TP concentrations in these lakes, including Lake Kissimmee, were lower than concentrations at S-65, thus pointing to causal factors near the structure. The prospect of a local influence near the outlet of Lake Kissimmee poses a more manageable problem for phosphorus control efforts, and the upward TP trend may prove to be a condition that can be rapidly reversed.

Although there is no obvious indication that urbanization has increased phosphorus loading from S-65, it remains a potential threat. Further land development as well as existing agricultural operations could affect future loading from the headwater lakes if nonpoint-source runoff is not controlled adequately. Consequently, efforts to monitor and control phosphorus runoff in the upper basin should continue.

Restoration of the Kissimmee River might reduce phosphorus loading downstream as the restored floodplain sloughs and marshes provide opportunity for retention and assimilation of phosphorus from the river's headwater- and watershed. The Kissimmee River is the largest inflow and source of phosphorus to Lake Okeechobee (SFWMD 2003), so reduced loading would significantly benefit lake eutrophication management. However, a reliable prediction of phosphorus load reduction is difficult due to insufficient knowledge of future river-floodplain hydrology and assimilation rates. As mentioned above, very little phosphorus data exist from the pre-channelized river, so it is unknown if net phosphorus retention occurred in the past. Likewise, not enough is known at this time about flow pathways, rates, volumes, and residence times in the restored river-floodplain to derive estimates of future phosphorus retention rates.

One indication that the floodplain may act to retain phosphorus comes from Boney Marsh, a 49 ha constructed wetland formerly located in Pool B. Over a nine-year period (1978–1986), this marsh received controlled but variable inflow with an average TP concentration of 0.056 mg/L. The marsh's monthly mean retention rate was 0.03 $g \cdot m^{-2} \cdot month^{-1}$ and TP concentrations in its outflow averaged 0.020 mg/L, representing a TP removal efficiency of 71% (Moustafa et al. 1996). It is difficult, however, to extrapolate these results to the entire river-floodplain system. Although this wetland represented a typical broadleaf marsh and the annual water regime was managed to generally resemble the average hydroperiod in the floodplain before channelization, it was not representative of the entire floodplain. Water depths in this marsh averaged 38 cm and fluctuated over a 90 cm range. The hydraulic residence time was 18 days (Mierau and Trimble 1988). Much of the water in the restored floodplain will be passing through areas with greater depths and shorter residence times.

Although a simple model of the restored river could be developed based on a set of assumptions of future discharges, phosphorus inputs, and assimilation rates, a large amount of uncertainty would be associated with calculations of future concentrations and loads. A simple model might not be able to deal with the spatial variability of soils, vegetation, and hydroperiod that could influence rates of assimilation and decomposition. Therefore, estimates of phosphorus concentration and load reductions have not been developed quantitatively in the form of an expectation.

Recommendations for Further Evaluation

The current monitoring design should be sufficient to detect changes in Pool C relative to Pool A. Monitoring of Pools A and C are expected to continue as described here, with grab samples collected at biweekly to monthly intervals at C-38 and river channel stations. These data will be supplemented by 24-hour monitoring of phosphorus collected by autosamplers in C-38. In addition, wetland stations were added to the water quality program in August 2001 as the floodplain in Pool C became inundated.

To improve evaluation of phosphorus loads, more accurate and reliable discharge estimates are needed. Also, possible factors contributing to higher concentrations should be examined more thoroughly. Studies of factors affecting phosphorus trends in Lake Kissimmee will be valuable, but an investigation of possible sources of phosphorus near S-65 is just as important. Because concentrations and loads at S-65B were similar to levels at S-65A and S-65C, a continuation of loading estimates at the location of S-65B (now demolished) is not necessary, but routine monitoring of water quality will continue at a nearby station in the river channel (KREA 98 in Montsdeoca Run).

More attention should be given to nutrient dynamics in the channelized and restored systems. Specific investigations should include phosphorus exchange between the river channel and floodplain, and assimilation and release of phosphorus in floodplain wetlands. Some preliminary work has been done to analyze phosphorus content in floodplain soil samples in Pool D. Further field collections, along with possible experimental studies, should be considered. Although it is too late to perform a field study of pre-restoration conditions in Pool C, an evaluation could be conducted using Pool D, with interpretations applied to Pool C as appropriate.

ACKNOWLEDGMENTS

The author thanks David J. Colangelo for assistance with statistical analysis, Cheol Mo for providing discharge and phosphorus loading data, and Lou Toth, Joseph W. Koebel, David H. Anderson, David J. Colangelo, Joseph Albers, and Zaki Moustafa for helpful comments on the manuscript. The author is also grateful to the staff of the SFWMD Environmental Resource Assessment Department, who collected and analyzed samples and managed the water quality monitoring program.

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CHAPTER 6

ALGAL SPECIES RICHNESS, DIVERSITY AND BIOMASS IN THE CHANNELIZED KISSIMMEE RIVER

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ABSTRACT: Algal studies were performed in the channelized Kissimmee River to establish baseline conditions for evaluating the effects of restoration of pre-channelization hydrology. Although no pre-channelization reference data exist, metrics for monitoring change in the algal community were selected based on existing literature and best scientific judgment. Baseline algal species richness and biomass (biovolume) were measured monthly from July 1999 through December 1999. Mean periphyton species richness and biovolume in remnant river channels of Pools A and C were 42.1 ± 2.0 (standard error of the mean) and 20.47 ± 9.05 mm^3/cm^2 respectively. Relative abundance of rheophilic periphyton (estimated by cell counts) was 28.7% ± 5.0%. However, none of the dominant species (>5% of total biovolume) were rheophilic. Periphyton biovolume was significantly related to dissolved inorganic nitrogen concentration. Mean species richness and biovolume of truly planktonic phytoplankton species were 22.5 ± 1.8 and 7.30 ± 2.85 mm^3/cm^2 per sample, respectively. After restoration of pre-channelization hydrology to the river channel, periphyton species richness and percentage of rheophilic species likely will increase, and species richness of truly planktonic phytoplankton likely will decrease.

INTRODUCTION

Algae are primary producers that form the base of the food web in many aquatic systems. Periphyton (algae attached to substrate) and phytoplankton (algae suspended in the water column) respond quickly to environmental change (Biggs 1996). Channelization of the Kissimmee River transformed the flowing river into a central drainage canal (C-38) composed of a series of reservoir-like pools. Examining changes in the algal community before and after backfilling of canal C-38 and restoration of flow to the river channel should be useful for understanding cause and effect relationships between physical restoration and ecosystem responses. For example, because algae are a major food source of many invertebrate and fish species, changes in algal communities may help explain changes in invertebrate and fish populations after the restoration project is complete.

During daylight hours, algae oxygenate the water column through photosynthesis, and at night, algae consume oxygen through respiration. Dissolved oxygen (DO) is necessary for the metabolism of most aquatic organisms, and DO concentrations in the channelized Kissimmee River can be extremely low, particularly during the warm, wet season (Colangelo and Jones 2005). Algal studies can help clarify the mechanisms driving DO dynamics.

Few studies examining algal communities in undisturbed, sub-tropical, blackwater river systems exist, and no periphyton or phytoplankton reference data exist from the Kissimmee River before channelization. This report focuses on the description of algal communities in the channelized Kissimmee River. Metrics to be monitored to detect changes in algal communities were chosen based on information available in the literature and best scientific judgment.

Objectives

The objective of this study was to establish baseline conditions for assessing effects of restored hydrology on algae species richness, diversity, and biovolume within the river channel.

METHODS

Baseline Conditions

Study Sites

The channelized Kissimmee River is characterized by non-flowing, stagnant conditions. Riverbed substrate consists of flocculent, unconsolidated organic material (Anderson et al. 2005). Littoral vegetation is dominated by *Salvinia minima*, *Scirpus cubensis*, *Ludwigia peruviana*, and *Nuphar lutea*. Approximately two thirds of the area between channel banks was vegetated (Bousquin et al. 2005).

Periphyton and phytoplankton species richness, diversity, and biovolume were quantified for remnant river runs in Pools A and C. Monitoring stations (five stations with three replicates each) were selected to cover several river reaches in each Pool (Figure 6-1). Sample sites were chosen to represent average habitat conditions (channel depth, littoral vegetation, etc.) in each Pool. Remnant river runs were approximately 20–30 m wide and 2–3 m deep with little or no flow.

Periphyton

Mean species richness and percentage of rheophilic species (cells) were chosen as metrics because baseline variability of these metrics was relatively low. Maximum species richness of periphyton often occurs in habitats with low to intermediate disturbance frequency (Connell 1978, Biggs et al. 1998, Sousa 1985), such as low to moderate changes in flow velocity and flooding. Restoration of stage and discharge to pre-channelization frequencies should increase disturbance frequency, which is very low under baseline conditions. Increased disturbance frequency in the river channel may allow an increase in periphyton species richness. Highest discharges in the pre-channelization Kissimmee River occurred during September–November with considerable annual variability (Toth 1993). Reestablishment of flow through the reconnected river channels should lead to an increase in the relative percentage of rheophilic species present in the river channel after flow is restored. Periphyton biovolume was used to determine species dominance based on biomass. The Shannon-Wiener diversity index (see following equation) was used as a measure of species diversity.

Shannon-Wiener Index

$$H' = \sum_{i-1}^{s} (p_i)\ (\log_2 p_i)$$

H' = information content of sample (bits/individual)
= index of species diversity
s = number of species
p_i = proportion of total sample belonging to ith species

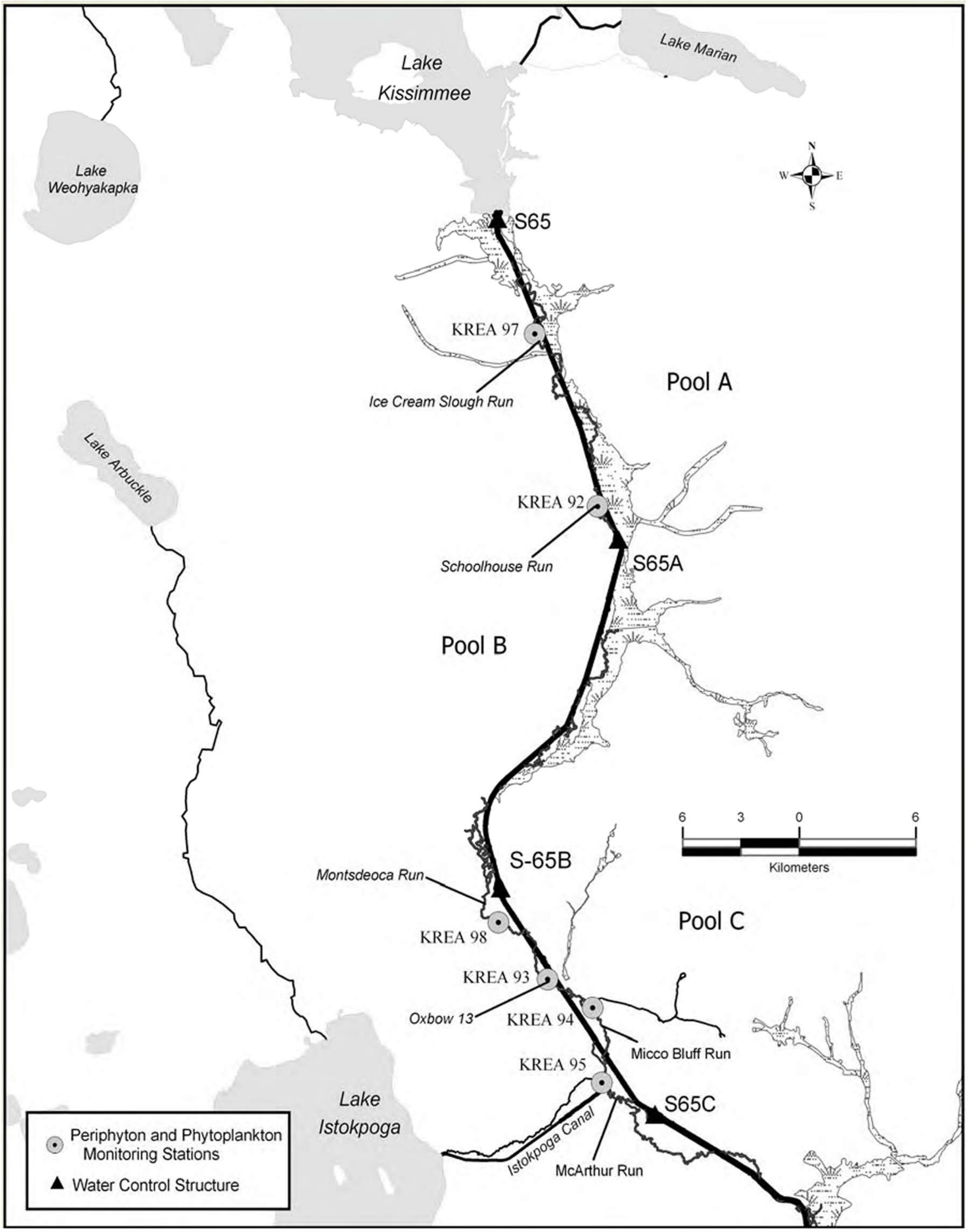

Figure 6-1. Periphyton and phytoplankton monitoring stations in Pools A and C of the Kissimmee River.

Periphyton were sampled monthly in Pools A (KREA92 and KREA97) and C (KREA93, KREA94, KREA95 and KREA98) (Figure 6-1) from August 1999 through December 1999 using artificial substrates (lightly sanded, clear acrylic rods) suspended from anchored floats. This method was developed following guidelines outlined in Barbour et al. (1999). Acrylic rods were approximately 6.5 cm long with a diameter of 1.2 cm. Floats had a diameter of 7.62 cm and were 3.81 cm long. Acrylic rods were attached to each float by inserting the rod into a hole in the center of the float. Floats were placed at the margins of and within littoral vegetation mats at each station (three floats per station) to maximize the range of light conditions present in the river channel. Acrylic rods were collected at the beginning of each month and replaced with new rods. At the time of collection, rod length (area exposed for periphyton growth) was recorded. Rods were kept in plastic sample bags on ice until they could be transported to the lab for processing. A small brush was used to scrape periphyton from each rod into a plastic tray. Approximately 100 ml of tap water was used to wash periphyton from the rod and brush. The sample was then mixed thoroughly and a 30 ml sub-sample was transferred to a 30 ml amber bottle. Lugol's solution (3.0% final volume) was added to preserve each sub-sample. Sample bottles were then refrigerated for future cell counts and taxonomic identification.

For algal identification and counts, samples were mixed thoroughly to suspend algae in the sample solution. Using a pipette, a Palmer-Maloney counting slide was filled with suspended periphyton sample. Samples were then examined on a compound microscope at 400X. Algae were identified to the lowest taxonomic level possible. Slides were scanned until at least 300 cells were identified and recorded along with the volume of water examined. Only cells that were viable (containing chloroplasts or protoplasm) were counted. To identify diatoms to species, a subset of each sample was taken and oxidized to clear diatom frustules of organic material. Cleaned frustules were dried on a cover slip and mounted with Naphrax mounting media to make permanent slides. These slides were examined under oil immersion at a magnification of 1000x. Cells were identified for each diatom form found in the corresponding Palmer-Maloney count. For example, if eight naviculoids were found, eight naviculoids were identified to species from the diatom slides. To estimate cell biovolume, average cell dimensions were measured during the counts or collected from the literature. The dimensions were applied to standard geometric shapes which approximate the shape of each taxon. Species richness was calculated by summing the species present in each sample.

Water quality sampling stations were located near algae sampling stations (Figure 6-1). Water samples were collected monthly and analyzed for soluble reactive phosphorus (SRP), dissolved inorganic nitrogen (DIN), turbidity, water temperature, pH and dissolved organic carbon (DOC) (Jones 2005). Results from these analyses were used to interpret results from periphyton and phytoplankton sample analyses. Table 6-1 summarizes water quality data used in this study.

Table 6-1. Mean water quality values in Pools A and C from August 1999 through December 1999.

Pool	Parameter	Value
A	DIN (mg/L)	0.01
	DOC (mg/L)	14.10
	Water Temp °C	23.21
	SRP (mg/L)	0.04
	Turb (NTU)	2.46
	pH	6.54
C	DIN (mg/L)	0.02
	DOC (mg/L)	22.22
	Water Temp °C	22.63
	SRP (mg/L)	0.03
	Turb (NTU)	6.34
	pH	6.19

Phytoplankton

Mean species richness of planktonic species was chosen as a metric to measure change because baseline variability was relatively low. After flow is restored, planktonic phytoplankton species richness is expected to decrease due to reestablishment of pre-channelization hydrologic conditions. Exclusively planktonic species should be flushed from reconnected river channels after flow is restored. However, because the floodplain and river channel will be hydrologically reconnected after flow is restored, suspended algae from the floodplain may enter the river channel as water levels fluctuate (Garcia de Emiliani 1997, Rojo et al. 1994).

Phytoplankton were sampled at the same stations as periphyton, monthly from July 1999 through December 1999. A vinyl tube (diameter = 2.5 cm, length = 100 cm) was vertically lowered into the water column to sample the surface meter of water. Stoppers were placed on the ends of the tube before the tube was pulled out of the water. The contents of the tube were then placed into a clean bucket and mixed. A 1.0 L plastic bottle was filled from the bucket and preserved with Lugol's iodine solution (3.0% final volume). Samples were kept cool and dark during transport. In the lab, the 1.0 L samples were concentrated to 20 mls using Imhof funnels. Plankton was allowed to settle for 48 hours. Methods for phytoplankton identification and biovolume calculation were identical to those used for periphyton. Biovolume of truly planktonic phytoplankton species was used to determine species dominance based on biomass. The Shannon-Wiener diversity index was used as a measure of species diversity.

Data Analysis

Summary statistics (mean, standard deviation, standard error of the mean, minimum and maximum values, and coefficient of variation (CV %)) were computed for algae species richness and cell biovolume. The Shannon-Wiener function (a measure of species diversity) was calculated for each sample. Linear regression was used to determine relationships between water quality variables and algae biovolume. Power analysis was used to calculate minimum detectable differences for all metrics. SAS version 8 (SAS Institute, Cary, NC) was used for all statistical analyses.

RESULTS

Baseline Conditions

Power analysis indicated the minimum detectable (90% of the time) increase in species richness of periphyton, percentage of rheophilic periphyton species, and minimal detectable decrease in species richness of truly planktonic phytoplankton was 2.0 species, 3.0% and 2.2 species, respectively. Complete lists of all periphyton and phytoplankton identified during this study are included in the Appendices 6-1A and 6-2A.

Periphyton Species Richness and Biovolume

Mean species richness in remnant river channels of Pools A and C was 46.1 ± 1.9 (standard error of the mean) and 38.1 ± 2.0 per sample, respectively (Table 2). Mean monthly periphyton species richness varied considerably (range = 28–52) (Figure 6-2); however, variability within each Pool on each sample date was low.

Species richness was generally higher during October, November, and December than during August and September. The mean Shannon-Wiener diversity index for samples collected in Pools A and C was 2.5 ± 0.1 and 2.8 ± 0.1 respectively.

Total rheophilic species collected in Pools A and C were 33 and 37 species, respectively. Mean relative abundance of rheophilic species (by cell count) was 22.6 ± 1.7% in Pool A and 34.8 ± 3.3% in Pool C (Table 6-2).

Mean periphyton biovolume in Pools A and C was 21.86 ± 6.5 mm^3/cm^2 and 19.08 ± 11.6 mm^3/cm^2, respectively (Table 6-2). Periphyton species comprising >5% of the total biovolume were considered dominant. Dominant species included *Gomphonema gracilis*, *Oedogonium* spp., and *Spirogyra* spp. in Pool A and *Oedogonium* spp., *Schizomeris leibleinii*, and *Spirogyra* spp. in Pool C (Table 6-3). *Spirogyra* spp. accounted for the majority of the biovolume present in both Pools. Temporal variability and mean periphyton biovolume was particularly high during October (Figure 6-3). The high mean biovolume value recorded during October in Pool C can be attributed to one extremely high value at station KREA98. None of the dominant periphyton species were rheophilic.

Regression analysis revealed periphyton biovolume increased significantly with DIN (r^2 = 0.37, p = 0.02). No significant relationships were found between biovolume and SRP, turbidity, water temperature, pH or DOC. Shannon-Wiener index values also were not significantly related to any of the water quality variables.

Table 6-2. Mean periphyton species richness, relative abundance (based on cell counts) of rheophilic species, mean biovolume, Shannon-Wiener index, and summary statistics in Pools A and C from August 1999–December 1999. Values in parentheses represent the standard error of the mean.

Species Richness	Pool A	Pool C
Mean	46.1 ± (1.9)	38.1 ± (2.0)
n	27	24
max	64	62
min	21	22
CV%	21.1	25.4
% Rheophilic	22.6 ± (1.7)	34.8 ± (3.3)
Biovolume (mm^3/cm^2)		
Mean	21.86 ± (6.5)	19.08 ± (11.6)
n	27	24
max	124.77	282.60
min	0.07	0.10
CV%	155.61	297.09
Shannon-Wiener	2.5 ± (0.1)	2.8 ± (0.1)

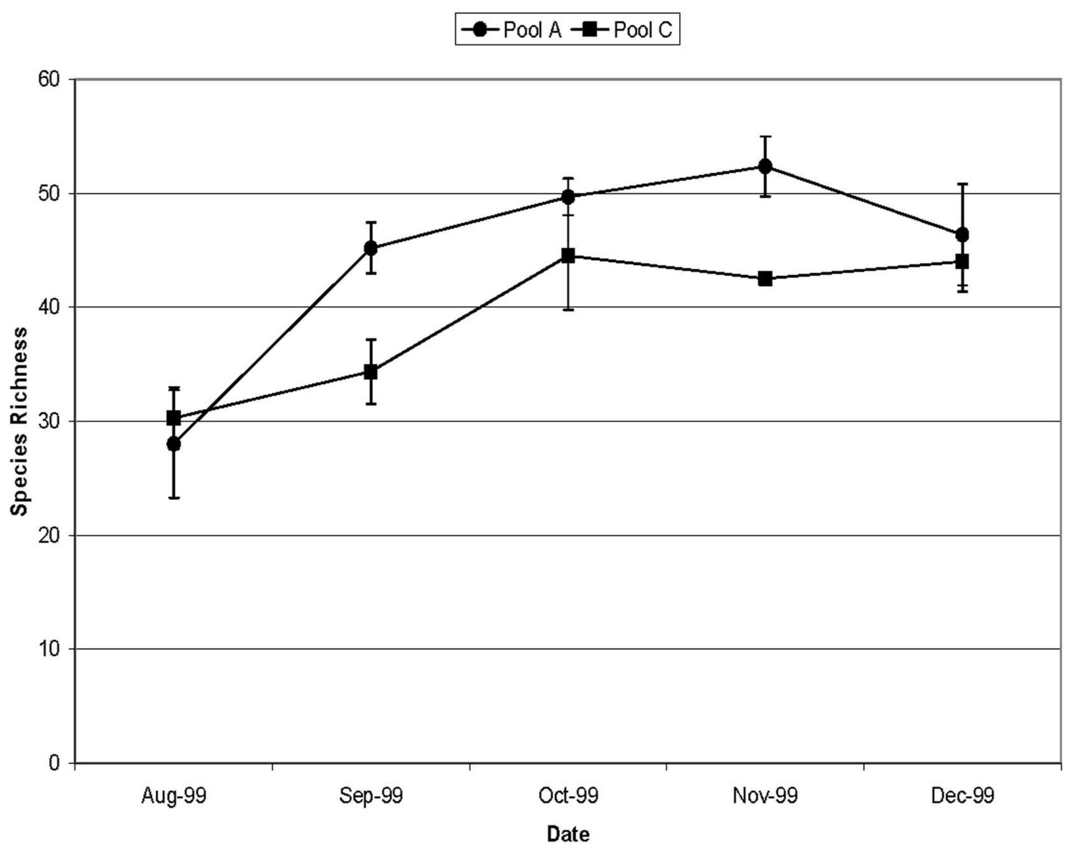

Figure 6-2. Mean monthly periphyton species richness in Pools A and C from August 1999 through December 1999. Error bars represent the standard error of the mean.

Mean species richness of truly planktonic species (water column obligates) per sample in remnant river channels was 22.3 ± 1.8 in Pool A and 22.6 ± 1.7 in Pool C (Table 6-4). Phytoplankton species richness varied considerably for some months and very little for other months (Figure 6-4). Highest species richness values occurred during September and lowest values occurred during October–November.

Mean biovolume of truly planktonic species per sample was 11.76 ± 4.58 and 2.85 ± 1.13 in Pools A and C, respectively (Table 6-4). Biovolume in Pool A varied considerably while variability in Pool C was relatively low (Figure 6-5). There were 11 dominant (>5% of total biovolume) phytoplankton species in Pools A and C (Table 6-5). Identities of species with highest biovolume in Pools A and C were different, with Euglenophytes dominating in Pool A, and Cyanophytes and Chlorophytes dominating in Pool C.

Kirchneriella subsolitaria was the most dominant species in Pool A and *Scenedesmus quadricauda* was the most dominant species in Pool C. The Shannon-Wiener index was 2.7 ± 0.2 in Pools A and C (Table 6-5).

Linear regression analysis showed no significant relationships between biovolume and DIN, SRP, turbidity, water temperature, pH or DOC. Shannon-Wiener index values also were not significantly related to any of the water quality variables.

Table 6-3. Dominant periphyton species (>5% of total biovolume) within remnant channels of the Kissimmee River from August 1999–December 1999.

Pool	Species	Division	% of total biovolume	Rheophilic *
A	*Gomphonema gracilis*	Bacillariophyta	8.3	
A	*Oedogonium* sp. *1*	Chlorophyta	15.8	
A	*Oedogonium* sp. *2*	Chlorophyta	6.1	
A	*Spirogyra* spp. *1*	Chlorophyta	54.2	
C	*Oedogonium* sp. *1*	Chlorophyta	17.4	
C	*Schizomeris leibleinii*	Chlorophyta	5.3	
C	*Spirogyra* sp. *1*	Chlorophyta	32.5	
C	*Spirogyra* sp. *2*	Chlorophyta	6.2	

* Palmer 1977.

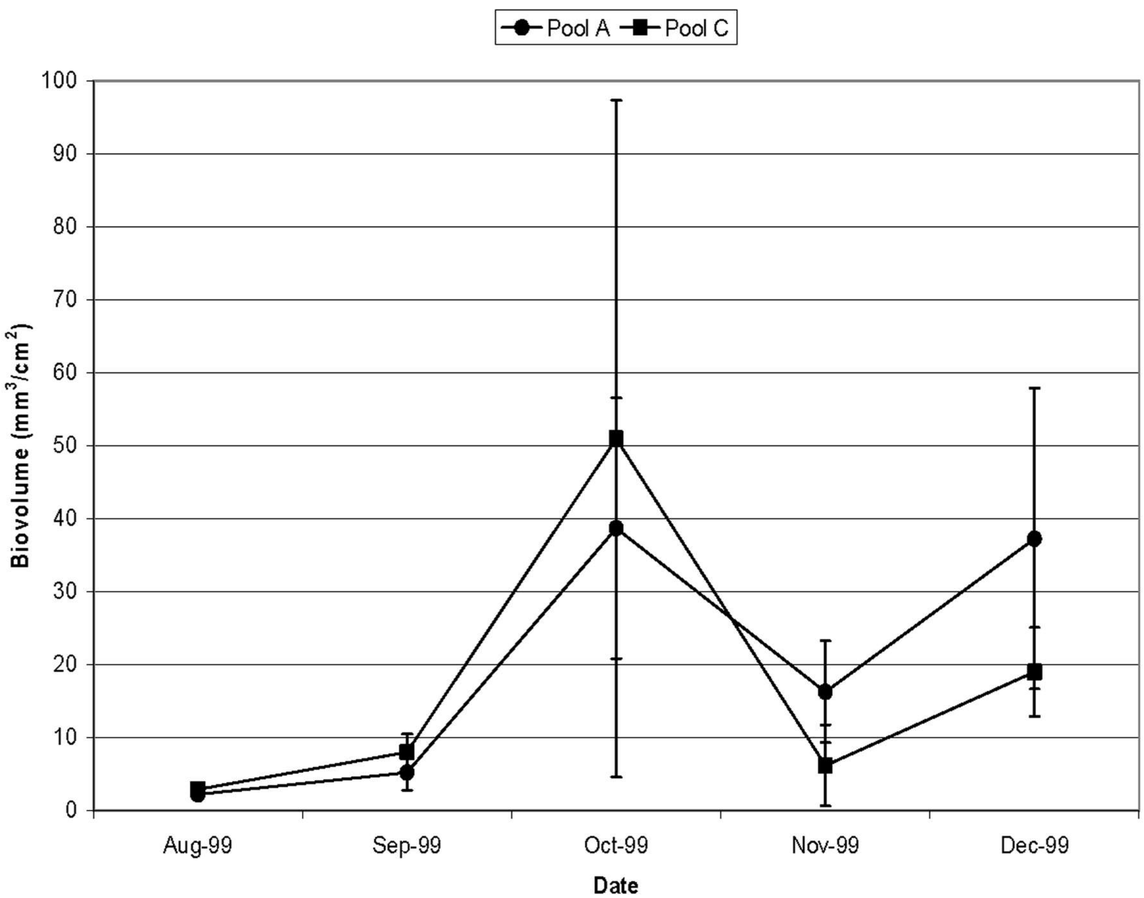

Figure 6-3. Mean monthly periphyton biovolume (mm^3/cm^2) in Pools A and C from August 1999 through December 1999. Error bars represent the standard error of the mean.

Table 6-4. Mean species richness, biovolume, and summary statistics of truly planktonic phytoplankton in Pools A and C from July 1999–December 1999. Values in parentheses represent the standard error of the mean.

Species Richness	Pool A	Pool C
Mean	22.3 ± (1.8)	22.6 ± (1.7)
n	14	13
max	36	35
min	11	14
CV%	30.4	26.6
Biovolume (mm^3/L)		
Mean	11.76 ± (4.58)	2.85 ± (1.13)
n	13	11
max	47.91	10.32
min	0.12	0.08
CV%	140.59	131.28
Shannon-Wiener	2.7 ± (0.2)	2.7 ± (0.2)

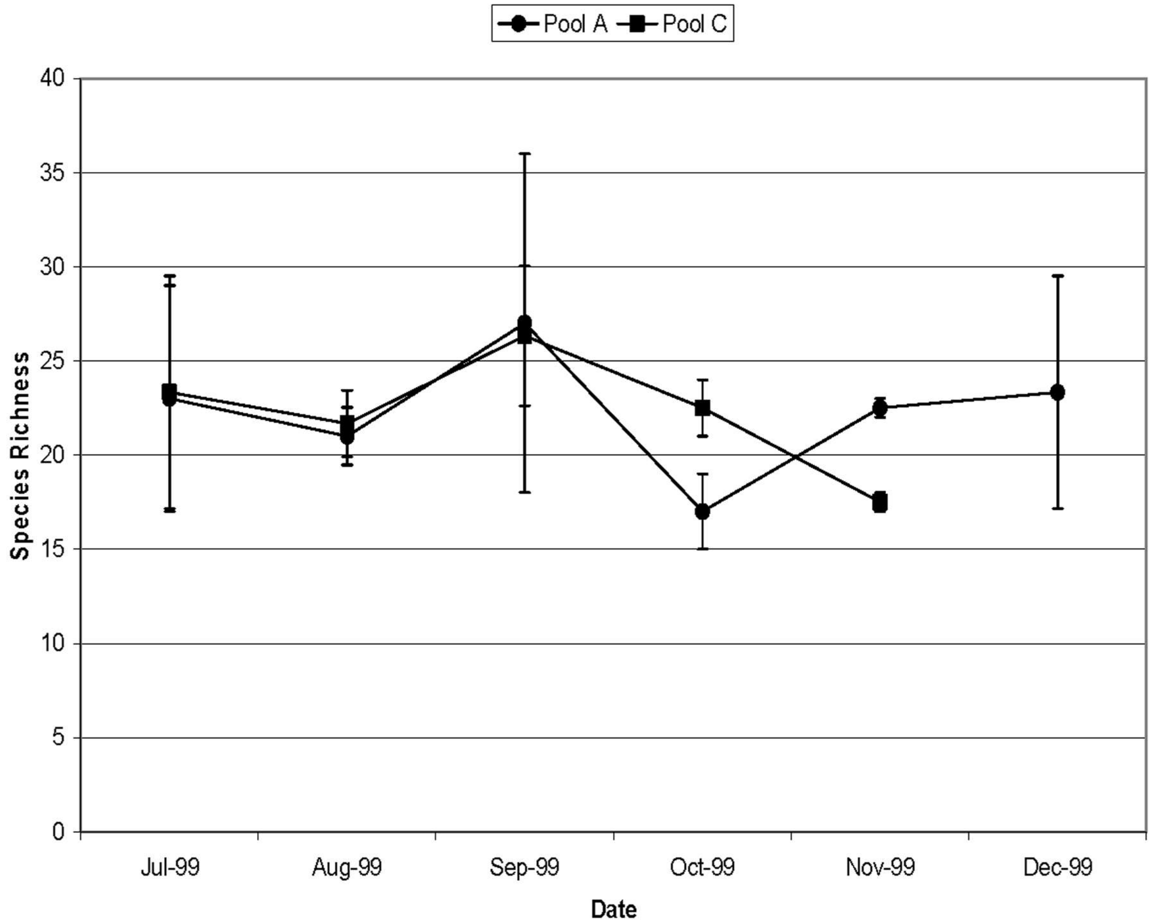

Figure 6-4. Truly planktonic phytoplankton species richness in Pools A and C from July 1999–December 1999. Error bars represent the standard error of the mean.

DISCUSSION

Baseline Conditions

Periphyton and Phytoplankton Species Richness and Biovolume

The periphyton and phytoplankton data presented in this chapter are limited because samples were only collected for five to six months in a single year. Several years of monthly samples would be necessary to capture seasonal and annual variability in algae species richness and biovolume. Additionally, data were collected after the beginning of Phase I construction in Pools B and C. Only data from stations that had not received flow at the time of collection were included in this study. However, with an area of disturbance as large as Phase I backfilling and construction nearby, we cannot entirely rule out the possibility of construction impacts on the periphyton and phytoplankton communities.

None of the dominant periphyton species were rheophilic. However, as many as 36 rheophilic species were present in the population. This source of rheophilic cells is potentially important for reestablishing a community structure dominated by rheophilic species normally found in a lotic system.

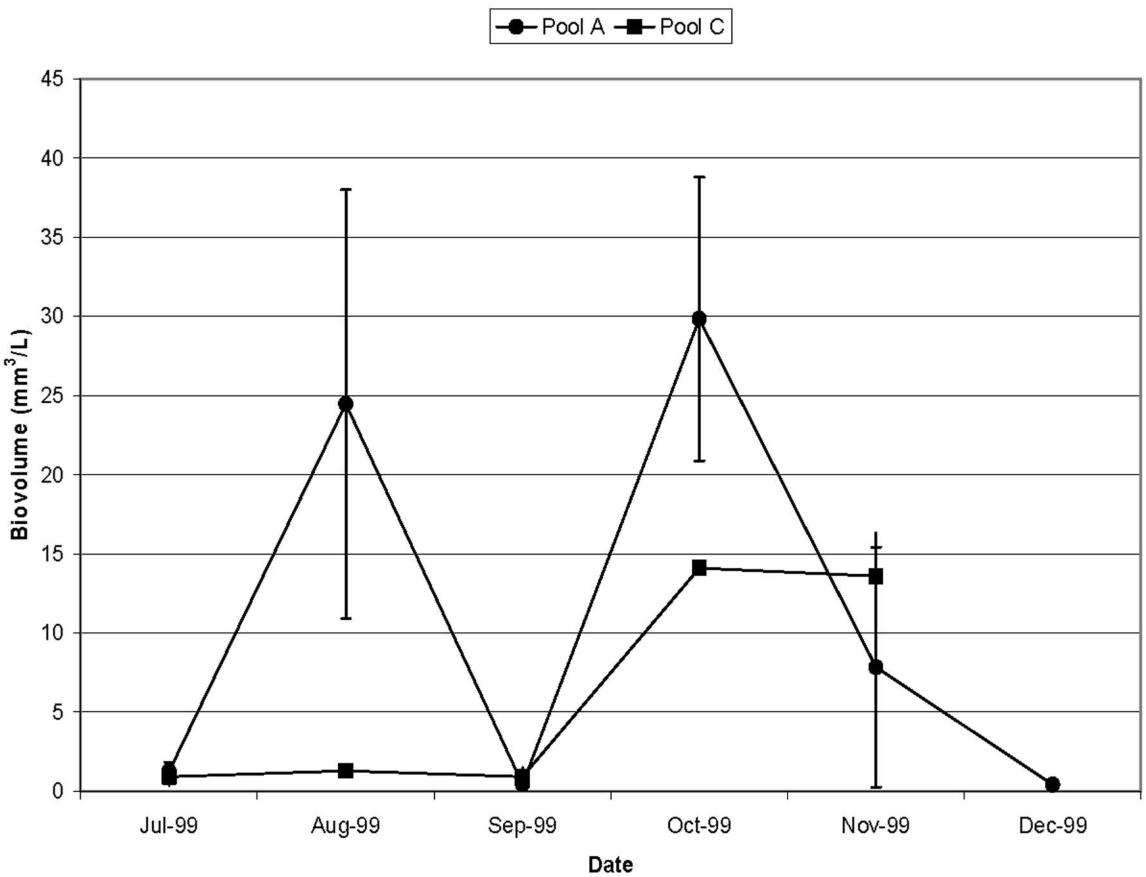

Figure 6-5. Mean monthly phytoplankton biovolume (mm^3/cm^2) in Pools A and C from July 1999 through December 1999. Error bars represent the standard error of the mean.

Table 6-5. Dominant (>5% of total biovolume) truly planktonic phytoplankton species within remnant channels of the Kissimmee River from July 1999–December 1999.

Pool	Species	Taxa	% of Total biovolume
A	*Aphanocapsa* sp.	Cyanophyta	6.2
A	*Euglena* sp.	Euglenophyta	16.5
A	*Euglena acus var. rigida*	Euglenophyta	10.2
A	*Euglena minuta*	Euglenophyta	8.0
A	*Kirchneriella subsolitaria*	Chlorophyta	21.5
A	*Cryptomonas erosa*	Other	7.8
C	*Chroococcus minor*	Cyanophyta	8.9
C	*Schizothrix calcicola*	Cyanophyta	5.4
C	*Euglena minuta*	Euglenophyta	7.8
C	*Scenedesmus armatus*	Chlorophyta	18.4
C	*Scenedesmus quadricauda*	Chlorophyta	20.8

Periphyton biomass is controlled by changes in resources (nutrients, light, and temperature) and disturbance (flooding, suspended sediment), or grazing by invertebrates and fish (Biggs 1996). Of all the water quality variables measured, periphyton biovolume responded significantly only to changes in DIN, suggesting that periphyton in the channelized system may be limited by nitrogen.

Spirogyra spp. and *Oedogonium* spp. were the most dominant periphyton species observed. Both taxa are classified as preferring low disturbance, moderately enriched habitats (Biggs et al. 1998). It is likely that after flow is restored, periphyton species that prefer low disturbance habitats such as low flow velocity canals and lakes, will be reduced in number.

Variability of phytoplankton species richness was low throughout the study area. However, biovolume was over four times greater in Pool A than in Pool C. Extremely high biovolume values were recorded at one station in Pool A (KREA 97, Ice Cream Slough) in August and October. High biovolume values recorded at this station are likely the result of a localized algae bloom (chlorophyll *a* and DIN values also were high at KREA 97 during August and October). However, because no significant relationships were found using regression analyses, it is unclear which environmental variables control changes in phytoplankton biovolume in the channelized Kissimmee River.

ACKNOWLEDGEMENTS

The authors wish to thank South Florida Water Management District Kissimmee Division staff and Riverwoods Field Lab staff for assistance with data collection and sample processing. We also thank Stephen G. Bousquin, Gary E. Williams, Therese East, Scot Hagerthey and Dan Haunert for comments and suggestions that improved this chapter.

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CHAPTER 7

LITTORAL VEGETATION IN THE CHANNELIZED KISSIMMEE RIVER

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ABSTRACT: To provide baseline (channelized-condition) data for assessment of the effects of restored flow on littoral vegetation in remnant channels, measurements of littoral plant communities were made under non-flowing conditions in remnant runs of the channelized Kissimmee River in 1998 and 1999. Variables measured included plant cover by species, the width of vegetation beds, and the vegetated percentage of channel area, among others. Reference data to estimate pre-channelization conditions were collected in an experimental semi-restored run in 1998 following a nine month episode of continuous flow diverted from canal C-38 by weirs installed for this purpose. During the non-flowing baseline period, vegetation beds were substantially wider and had higher cover of floating and mat-forming species relative to cover of emergent species than in the reference data. In the enhanced-flow reference data used to estimate pre-channelization conditions, river channels had narrower mats and plant communities were heavily dominated by emergent species. Comparison of these data sets suggests that as a result of elimination of flow, vegetation bed widths increased and changes in the structure of plant communities occurred, greatly reducing relative cover of emergent species. Channelization of the Kissimmee River likely precipitated a string of effects associated with these changes in littoral vegetation, resulting in interrelated impacts on channel morphology, water quality, and wildlife habitat. Reestablishment of flow in the Kissimmee River is expected to restore littoral vegetation to conditions more typical of lotic, pre-channelization conditions, in which littoral vegetation is limited to relatively narrow littoral zones near the edges of channels and is dominated by emergent species.

INTRODUCTION

In flowing rivers, growth of macrophytes is constrained primarily by channel depth and flow (Dawson 1998). These and other components of the physical habitat of river channel plants were substantially modified by elimination of flow in the Kissimmee River following construction of canal C-38 (Bousquin et al. 2005). Virtually all flow in the former (remnant) river channel was intercepted by the canal. Changes in plant habitat related to elimination of flow included alterations in channel cross-section, substrate characteristics, channel depth, and distribution of point-bars (Anderson et al. 2005); and water chemistry (Colangelo 2005, Jones 2005).

Aquatic plant species that are emergent in growth-form (rooted to the substrate) are well-adapted to a flowing river environment. Rooted beneath water, the shoots and leaves of emergent plants extend above the surface for photosynthesis and gas exchange (Reimer 1984, Dawson 1988). Because they are attached to the substrate, emergent plants can resist translocation by normal flows in rivers and tend to dominate in-channel plant communities under flowing conditions. In contrast, under non-flowing conditions free-floating species tend to increase because they can occupy deeper sections toward the center of channels and can expand in area at the water surface in shallower areas, which may enhance competitive advantages for space and light over established or propagating emergents. Mat-forming species such as *Scirpus cubensis* also may increase under non-flowing conditions; in turn, the surface mats they form can provide boglike substrates that can be colonized by emergents and terrestrial species (Figure 7-1), including shrubs (Milleson et al. 1980, Miller et al. 1990).

Figure 7-1. A remnant channel in the channelized Kissimmee River, ca. 1999, showing "floating mat" expansion toward center of river channel.

Because of the sensitivity of river channel plants to flow regimes, the character of river channel littoral (edge) vegetation can be expected to change following elimination or restoration of flow, both in species composition and the areal extent of vegetation. For a river system, plant communities that are characteristic of flowing conditions are an indicator of an ecologically functional system. For these reasons, the effects of channelization on littoral vegetation in the Kissimmee River are of interest to the restoration evaluation program, both as simple indicators of change following restoration of flow and as indicators of progress toward ecological integrity.

Objectives

The objectives of this study were to:

(a) Estimate baseline (channelized) conditions in littoral vegetation beds and plant communities;

(b) Quantify the impacts of channelization and resulting elimination of flow on littoral plant communities, by using reference data to estimate pre-channelization conditions;

(c) Develop expectations (predictions based on reference data) for responses of littoral vegetation to restoration for the purpose of evaluating the restoration project goal of restoring ecological integrity.

Littoral Plant Communities

Littoral vegetation beds of several recognizable types occur in the channelized Kissimmee River. Which species occur at a particular location is determined by flow and water depth, type of substrate, and channel curvature; founder effects and availability of propagules are likely additional factors. Assemblages vary in species composition and dominance; some common river channel plant communities are defined in Bousquin and Carnal (2005) and are separated in that report by growth forms of typical species, e.g, emergent or floating species.

Shallow areas adjacent to banks tend to be characterized by short-stature emergents such as *Polygonum densiflorum* (smartweed), *Hydrocotyl umbellata* (pennywort), and various aquatic grasses such as *Sacciolepis striata* (cupscale) and *Panicum hemitomon* (maidencane). Under non-flowing conditions, at least where exotics are controlled, deeper areas adjacent to steep banks and at the deep edges of mats are often dominated by *Nuphar lutea* (spatterdock), a native emergent with long petioles connecting fleshy, bottom-rooted rhizomes to floating leaves. It occurs in water as deep as 2–3 m (S. Bousquin, South Florida Water Management District, personal observation).

Semi-buoyant "floating mats" up to 1 m thick, formed of dead plant material and prevalent in non-flowing channels, serve as suitable substrates in deep areas for emergent species, including those listed above, wetland shrubs such as *Ludwigia peruviana* (Peruvian primrosewillow) and *Salix caroliniana* (coastal plain willow), and upland shrub species such as *Myrica cerifera* (waxmyrtle). Pockets of open water in these mats may provide habitat for floating species like *Salvinia minima* (water spangles, a small-fronded (<0.5 in dia) native aquatic fern most often found in low current conditions, e.g., in channelized runs and in flowing runs in backwaters and, under flowing conditions, in areas sheltered by other plants); often in association with another small floating fern, *Azolla caroliniana* (mosquito fern); *Wolfiella gladiata* (mudmidget); and species of *Lemna* (duckweed).

Under non-flowing conditions and lacking weed control, extensive mats composed nearly exclusively of invasive floating species (*P. stratiotes* and *E. crassipes*) may occur. Such large mats, which may be virtually monospecific, are dependant on lack of flow to stay in place and possibly, except in deep areas, to retain a competitive advantage over emergents.

METHODS

Study Areas

Baseline sampling was conducted in remnant river channels that had lacked sustained flow since completion of channelization in 1971. Channel substrates were composed of unconsolidated deposits of organic material averaging 14 cm in thickness in Pool C runs. These deposits overlaid the original pre-channelization sandy riverbed (Anderson et al. 2005). River channel widths ranged from 11.9 m to 69.0 m and averaged 36.1 m. Channel depths to firm sand averaged 1.7 m (Anderson et al. 2005). Reference data were collected in June 1998 from an experimentally semi-restored remnant channel (River Run #1) in lower Pool B. This run had received intermittent flows and stage fluctuations since 1985, and continuous, moderate to high flow for nine months prior to data collection. Weirs placed across the C-38 canal as part of the Demonstration Project (Toth 1991) diverted water through the remnant channel.

In the channelized system prior to the late 1980s, beds of floating exotic vegetation were common, occasionally completely spanning remnant channels (Campbell 1989; R.M. Bodle, L. Toth, South Florida Water Management District, personal communications). An intensive herbicide treatment program was undertaken during 1983–1987 to reduce cover of these species. Substantial reductions in the cover of floating exotics to low "maintenance control" levels had been achieved by 1988 (Grimshaw 2002). Targeted cover in this ongoing program is approximately ≤5% absolute cover (R.M. Bodle, South Florida Water Management District, personal communication). These species have been kept relatively stable at these low levels since 1988 by regular (usually twice annually) herbicide applications (Grimshaw 2002). Weed control has and will continue to maintain low levels of invasive species over the period of data collection in both the reference and baseline study areas. Therefore, the magnitude and effects of herbicide applications are assumed to be similar in the Impact, Control, and Reference areas (defined below), and are not viewed as confounding factors.

Sampling Methods

Sampling Methods-Baseline Data

Baseline sampling was conducted twice annually over a two-year period from 1998–1999 during the winter dry season (usually February–March) and the summer wet season (August–September), except in 1998 when dry season sampling extended into May. Sampling was conducted at fixed transects distributed in non-flowing (remnant) channels of Pools A, B, and C (Map Appendix 1A–8A). Each transect is permanently marked on opposite banks with galvanized steel poles. Transects are located at channel bends and straight reaches to capture variation associated with channel shape or pattern.

Baseline sampling was conducted in one-meter wide belt transects established by sighting between the transect poles and placing 1 m by 2 m quadrats on the upstream side of the sightline, with the long dimension of the quadrat on the transect. Baseline surveys were initiated at the left bank facing downstream and were continued across the channel by adding consecutive quadrats. For each quadrat, we recorded the overall percentage cover of living and dead vegetation to the nearest 5%, and cover of all plant species using a six-level system developed by Daubenmire (Table 7-1) (Daubenmire 1959). Several metrics, described below, were derived from the raw cover, cover class, and dimensional measurements. The midpoints of cover classes were used for calculations involving species cover classes (Table 7-1) (Daubenmire 1959).

Table 7-1. Cover ranges and midpoints of the Daubenmire scale (Daubenmire 1959).

Cover Class	Range (%)	Midpoint (%)
0	0	0
1	0.1 - 5.0	2.5
2	5 - 25	15
3	25 - 50	37.5
4	50 - 75	62.5
5	75 - 95	85
6	95 - 100	97.5

Vegetation data from 91 transects were used in the baseline analyses. Impact area channels, which will receive flow following backfilling as part of Phase I of the restoration project, were sampled at 70 transects distributed among the five remnant channels in Pool C and the southernmost remnant channel in Pool B (Map Appendix 6A–8A). Twenty-one transects were sampled in three channels in Pool A where flow will not be restored (Map Appendix 1A–4A), which will be used as a control area. Data from the Control area will be incorporated in the restoration evaluation to assess the effects of background variation in measured variables using a before-after-control-impact (BACI) approach (Stewart-Oaten et al. 1992).

For data analyses, each transect was subdivided into two transect sections, one on either side of the channel. Transect sections were used to distinguish individual vegetation beds on opposite banks, and are the basic calculation unit for most species and dimensional metrics used in this study (see descriptions of individual metrics, below). Calculations based on transect sections refer only to the vegetation bed from which they were derived. For example, a species' relative cover value refers to the relative cover estimate for the species in one of the two possible vegetation beds (left or right) sampled at a transect. Calculations that combine section metrics are based only on quadrats that intercept an area with ≥5% vegetation; derived values such as means for these metrics do not consider unvegetated quadrats.

Transect sections in both the baseline and reference data sets were categorized as being located at inner margins of curved channel bends (inner), outer margins of curved channel bends (outer), or at the margins of straight reaches (straight) (Figure 7-2) for evaluations of variation in width associated with channel pattern or curvature/position. Grand means of widths presented for the baseline period are the averages of the four baseline sample period means for each pattern category (n=4). Widths and vegetated percentage of channel were averaged over all sampled transect sections in each pattern category for each of the four sample periods. An average of 130 transect sections were measured per sample period in the Impact area; 42 transect sections were measured per sample period in the Control area.

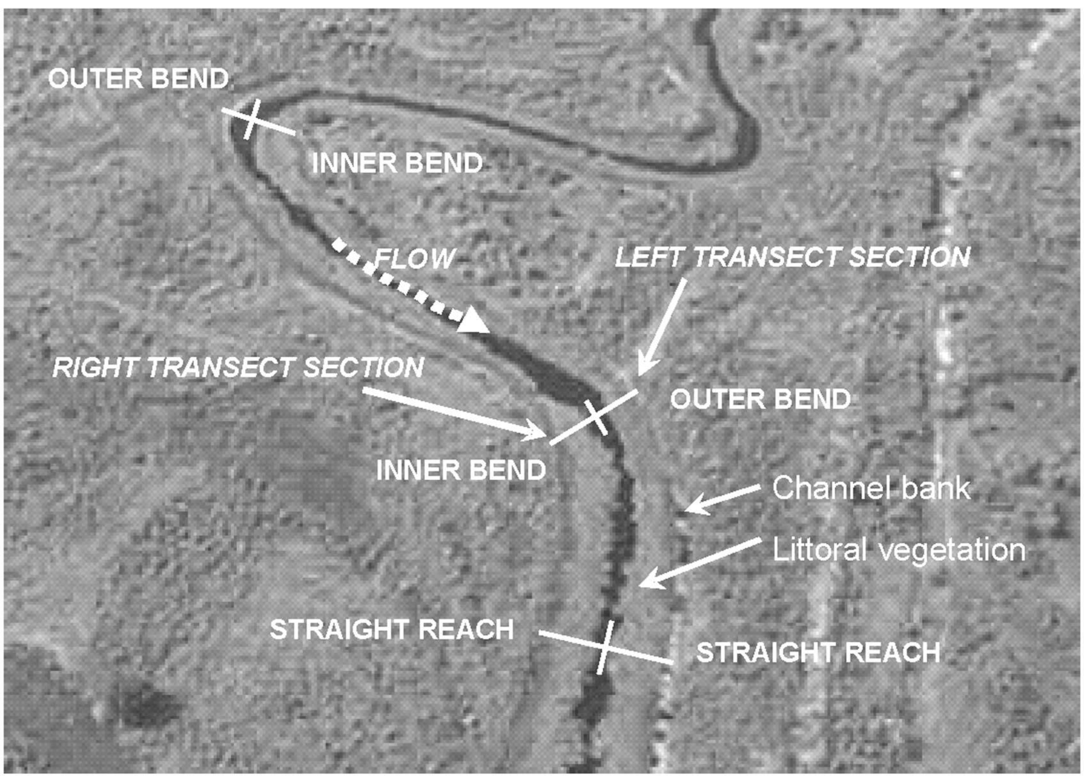

Figure 7-2. Diagram of transect orientation illustrating classification of transect sections at inner bends, outer bends, and straight reaches.

Relative cover and species richness were averaged over all sampled vegetated transect sections for each species or growth-form for each of the four baseline sample periods. Grand means for the baseline period are the averages of the four sample period means for each species or growth-form (n=4). An average of 125 vegetated transect sections occurred per sample period in the Impact area; 42 vegetated transect sections occurred per sample period in the Control area.

Sampling Methods-Reference Data

Reference surveys to estimate pre-channelization conditions used methods similar to those presented above for baseline data. Quantitative reference data to estimate pre-channelization littoral plant community structure and the width of vegetation beds were obtained from the semi-restored run. Cover class (Daubenmire 1959) (Table 7-1) data from a field survey of 13 transects in the semi-restored channel (C. Hovey, unpublished data) and cover estimates from photointerpretation of 1998 aerial photography (C. Hovey, unpublished results) were used to estimate mean relative cover of plant species under flowing conditions. Relative cover means for this survey are the averages of sampled vegetation beds (transect sections, two per transect, n=26) that occurred at the 13 transects.

Reference estimates of vegetation bed widths were derived from data collected in a concurrent but separate survey of vegetation beds in the same channel (C. Hovey, unpublished). Width data were collected at 42 beds at inner channel bends (n=11), outer bends (n=19), and straight reaches (n=12) of river channel. Beds in each of these categories were averaged to derive reference means. Additional qualitative assessment of actual in-channel vegetation cover prior to construction of canal C-38 was based on June 1956 black and white aerial photography (1:12000) (C. Hovey, unpublished).

Methodology for the reference survey data differed in some respects from baseline methods. Of the metrics used in the baseline data, only width, relative cover, relative frequency, and importance could be calculated with confidence from the reference data. Channel widths were not recorded and the precise locations of measurements are unknown, so vegetated percentages of channels could not be calculated. In some other respects, data collection for the reference survey data differed from baseline methodology. Only littoral beds delineated on aerial photographs of the area were measured in the 1998 ground survey, so bends without vegetation were not included in estimates of mean bed width. This methodology probably results in inflated means compared to those calculated from baseline surveys, particularly along outer channel bends, which may lack vegetation under flowing conditions. However, because this difference would result in bias in the direction opposite from that expected (wider beds in the reference period, so less baseline/reference contrast), this is not viewed as a large problem for baseline/reference comparisons. Although River Run #1 was not fully restored by the Demonstration Project, the reference data represent a point on a trajectory toward probable community structure and bed width in a restored system.

Grouping Variables

Both the reference and baseline data sets were organized by categorical grouping variables, including Area (Impact, Control, or Reference), and Season (summer or winter). Species were categorized by Growth Form as emergent, floating and mat-forming, submergent, or N/A (family or genus only, or unidentified species); and by Origin as native, non-native, or unknown.

Metrics

The following metrics were measured or derived for baseline period data. Those marked † were also measured or derived for the reference data. Section metrics were measured for individual vegetation beds within transect sections and refer only to vegetated quadrats. Transect metrics refer to entire transects, including both vegetated and unvegetated (open water) quadrats.

<u>Width (section metric)</u>†. An estimate of the lateral dimension of a littoral vegetation bed from the bank to its waterward edge. Beds were considered to end at the most waterward quadrat where total living plant cover was ≥5%. Width was estimated by multiplying quadrat length (2 m) by the number of contiguous vegetated quadrats containing ≥5% absolute plant cover in the transect section; the last waterward quadrat was estimated to the nearest 1 m if less than the entire quadrat contained ≥5% cover.

<u>Relative Cover (section metric)</u>†. An estimate of the cover of a plant species or group of species (e.g., emergent species or native species) relative to the cover of all species in a transect section. It was calculated by dividing the sum of the cover class midpoints of each species or group of species in all quadrats in the transect section by the sum of the cover class midpoints of all species in all quadrats in the same transect section. Relative cover was calculated for each species, species growth-form (floating and mat-forming species, emergent species, or submergent species), and species origin category (native, non-native).

Relative Frequency (section metric)†. As calculated here, frequency is the number of quadrats in which a species occurs in a transect section. Relative frequency is the frequency of a species divided by the sum of the frequencies of all species in all transect sections.

Section Percentage Frequency (section metric)†. Percentage of the number of transect sections at which a species or group of species was present.

Importance Value (IV) (section metric)†. Cover or frequency alone may be a misleading indicator of the influence of species in communities, given that other measures are possible and may convey different relationships. Relative frequency expresses commonness of species across sampling sites, while cover better-expresses influence within the community. Importance value (IV) (Grieg-Smith 1983) is an index value between 0 and 100, calculated here as the sum of the relative cover and relative frequency values for a species in a transect section. Although this index has problems when used in a community comparison context (Brower et al. 1990), importance value is used here to give a better estimate of the relative influence of individual species in communities than either cover or frequency alone.

Percentage Live Cover (section metric). Average percentage cover of living plants in a vegetation bed. Percent live cover was calculated for each transect section on each sampling date by averaging the percent live cover of vegetated quadrats at each transect section. Two transect section values were derived per transect per sampling period. This is the only metric that is a mean of quadrat values prior to calculation of group means, although the term "mean" is not included in the name of the metric.

Species Richness (section metric)†. Species richness is the number of species found in a transect section.

Percentage Vegetated Area (transect metric). Estimate of the percentage of the river channel covered by ≥5% vegetation cover in a transect. Percent vegetated area was calculated for each transect on each sampling date by dividing the number of vegetated quadrats by the total number of quadrats in a transect. Percent vegetated area is an estimate of vegetated area relative to channel area within the area of the belt transect, and so standardizes mat widths with respect to the width of channels.

Statistical Methods

Except for bed widths, data were seldom normally distributed across sampling periods. Baseline and reference data that did not pass the Kolmogorov-Smirnov test of normality were compared with the Kruskal-Wallis one-way analysis of variance on ranks; reference-baseline comparisons of widths were conducted with two-way ANOVA. All comparisons were one-tailed and were considered significant at p <0.05 or marginal in significance if close to 0.05. All means are reported and graphed with ± one standard error. SAS version 8 (SAS Institute, Cary, NC) was used for statistical analyses.

Baseline Period Control Area Data

When interpreting baseline period (channelized-condition) data, it is important to recognize that the stratification of sampling in the Control and Impact areas is a distinction determined exclusively by planned restoration project activities, i.e., impacts (e.g., restored flow, hydroperiod, and other hydrologic characteristics) that were planned to take place in the Impact area but not in the Control area. At the time of baseline data collection, no impact had yet taken place. The purpose of Control area data is to evaluate Impact area responses using BACI-type approaches (e.g., Stewart-Oaten et al. 1992) with respect to an area where the impact had not occurred. Although Control area data are of intrinsic interest in baseline studies, to avoid confusion in comparisons, Control area data are being reserved for future BACI analyses in which the Impact area data will be compared with future evaluation data from the same location following restoration of flow. Control area data are presented in Table 7-2, but Impact area data alone are used in this report to represent the baseline period in graphs and statistical comparisons.

RESULTS

Vegetated Percentage of Channels and Width of Vegetation Beds

During the baseline period, mean vegetated percentage of river channels was 56.7% ± 5.0%. Inner bend widths averaged 12.4 m ± 0.7 m, outer bends 6.0 m ± 1.0 m, and straight sections 9.3 m ± 0.6 m

(Figure 7-3). In the 1998 reference field survey used to estimate pre-channelization conditions, mean widths were 5.0 m ± 0.4 m on inner bends, 3.8 m ± 0.5 m on outer bends, and 3.6 m ± 0.6 m on straight reaches (Figure 7-3). Pre-channelization (1956) aerial photography (C. Hovey, unpublished) proved problematic to interpret decisively because a distinct littoral zone was difficult to distinguish from marshes and other floodplain edge vegetation. However, the width of littoral vegetation beds appeared greatest on inner channel bends, where beds tended to be approximately twice as wide as on outer bends, which often had little littoral vegetation. Widths of vegetation beds on either side of straight channel reaches appeared approximately equal, but not as wide as on inner bends nor as narrow as on outer bends (C. Hovey, unpublished report). The aerial photography results are consistent with the ground survey results. Reference and baseline means for inner bends and straight reaches were significantly different (P <0.001, two-way analysis of variance on ranks) (Table 7-2). Outer bends were not significantly different (P = 0.081) (Table 7-2).

Table 7-2. Metric means for Control, Impact, and Reference (flowing, using estimates of pre-channelization) area data. Only Impact and Reference data are compared statistically. Nonsignificant results are marked with an asterisk (*).

Metric	Category	Area	Mean	Standard error	n	P
Width (m)	Inner	Control	12.5	0.6	4	N/A
		Impact	12.4	0.7	4	< 0.001
		Reference	5.0	0.4	12	
	Outer	Control	7.9	0.7	4	N/A
		Impact	6.0	1.0	4	0.081*
		Reference	3.8	0.5	20	
	Straight	Control	13.8	0.4	4	N/A
		Impact	9.3	0.6	4	< 0.001
		Reference	3.6	0.6	13	
Relative cover (by growth form) (%)	Emergent	Control	62.2	4.0	4	N/A
		Impact	43.3	3.4	4	<0.01
		Reference	95.5	2.0	13	
	Floating & Mat-forming	Control	34.1	3.9	4	
		Impact	49.6	4.0	4	<0.01
		Reference	4.5	2.0	13	
Relative cover (by origin) (%)	Native	Control	80.5	3.2	4	N/A
		Impact	74.4	4.6	4	<0.01
		Reference	95.5	2.0	13	
	Non-native	Control	18.8	3.3	4	N/A
		Impact	25.2	4.7	4	<0.01
		Reference	4.5	2.0	13	
Richness (n species)		Control	16.2	0.7	4	N/A
		Impact	11.6	0.2	4	<0.01
		Reference	4.9	0.7	13	
Average percentage live plant cover (%)		Control	59.6	4.4	4	N/A
		Impact	43.6	6.0	4	
Vegetated percentage of channel (%)		Control	75.9	3.9	4	N/A
		Impact	56.7	5.0	4	

Relative Cover and Species Richness

Emergent species and floating/mat-forming species had similar mean relative cover in the baseline period. Of living plant cover, 43.3% ± 3.4% was emergent species, 49.6% ± 4.0% was floating and mat-forming species, and the remainder was submergent and other species (e.g., terrestrial species and taxa identified only to family or genus) (Figure 7-4). In contrast, emergent species clearly dominated littoral zones in the reference semi-restored flowing channel. Based on the field survey data, mean combined relative cover of emergents was 95.5% ± 2.0%, and the estimate based on photointerpretation was 97%. Mean combined relative cover of floating and mat-forming species in the field survey was 4.5% ± 1.9%, and 3% in the photointerpretation estimate. Mean relative cover of emergent species and floating and mat-forming species were significantly different between the baseline and reference survey data (P <0.01, Kruskal-Wallis one-way analysis of variance on ranks) (Table 7-2).

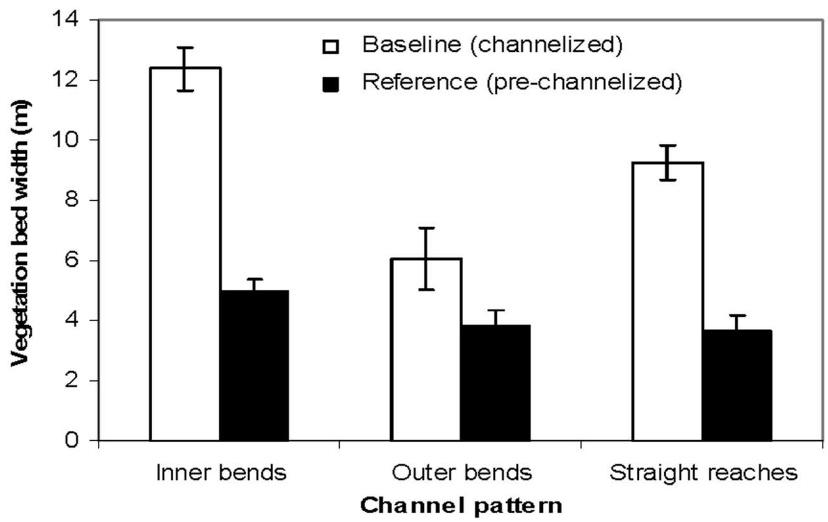

Figure 7-3. Vegetation bed widths on inner and outer bends and straight reaches in the baseline and reference area data. Error bars represent the 95% confidence interval of the mean.

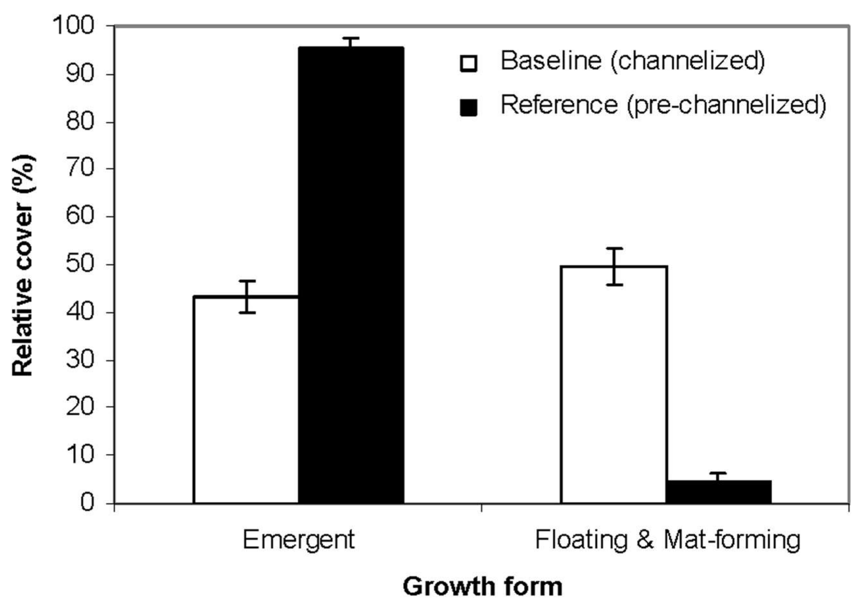

Figure 7-4. Mean relative cover of plant growth forms in baseline (channelized) and reference (flowing) channels. Error bars represent the 95% confidence interval of the mean.

Table 7-3. Mean relative cover, mean relative frequency, and importance values expressed as proportions for all species that occurred with values of ≥5% in any of these metrics in the baseline or reference data. Importance is the sum of relative cover and relative frequency.

Form	Code	Species	Relative cover (%)		Relative frequency (%)		Importance	
			Reference (pre-channelized)	Baseline (channelized)	Reference (pre-channelized)	Baseline (channelized)	Reference (pre-channelized)	Baseline (channelized)
Emergent	AP01	*Alternanthera philoxeroides*	0.0	2.1	0.0	5.2	0.0	7.3
	HU01	*Hydrocotyle umbellata*	12.5	8.8	18.8	6.8	31.3	15.6
	LP01	*Ludwigia peruviana*	0.0	3.2	0.0	4.1	0.0	7.3
	NL01	*Nuphar lutea*	26.4	11.0	20.3	4.2	46.7	15.2
	PD01	*Polygonum densiflorum*	35.2	4.7	25.0	4.1	60.2	8.8
	PH01	*Panicum hemitomon*	5.5	0.6	9.4	1.4	14.9	2.0
	SS01	*Sacciolepis striata*	4.1	8.5	6.3	6.9	10.4	15.4
Floating & Mat-forming	EC01	*Eichhornia crassipes*	2.5	0.5	4.7	0.8	7.2	1.3
	LM99	*Lemna* sp.	0.0	5.5	0.0	7.7	0.0	13.2
	PS01	*Pistia stratiotes*	2.0	7.6	4.7	5.3	6.7	12.9
	SC05	*Scirpus cubensis*	0.0	10.3	0.0	5.5	0.0	15.8
	SM01	*Salvinia minima*	0.0	20.8	0.0	7.7	0.0	28.5
	WG01	*Wolffiella gladiata*	0.0	2.9	0.0	5.2	0.0	8.1

Table 7-3. Mean relative cover, mean relative frequency, and importance values expressed as proportions for all species that occurred with values of ≥5% in any of these metrics in the baseline or reference data. Importance is the sum of relative cover and relative frequency.

Form	Code	Species	Relative cover (%)		Relative frequency (%)		Importance	
			Reference (pre-channelized)	Baseline (channelized)	Reference (pre-channelized)	Baseline (channelized)	Reference (pre-channelized)	Baseline (channelized)
Emergent	AP01	*Alternanthera philoxeroides*	0.0	2.1	0.0	5.2	0.0	7.3
	HU01	*Hydrocotyle umbellata*	12.5	8.8	18.8	6.8	31.3	15.6
	LP01	*Ludwigia peruviana*	0.0	3.2	0.0	4.1	0.0	7.3
	NL01	*Nuphar lutea*	26.4	11.0	20.3	4.2	46.7	15.2
	PD01	*Polygonum densiflorum*	35.2	4.7	25.0	4.1	60.2	8.8
	PH01	*Panicum hemitomon*	5.5	0.6	9.4	1.4	14.9	2.0
	SS01	*Sacciolepis striata*	4.1	8.5	6.3	6.9	10.4	15.4
Floating & Mat-forming	EC01	*Eichhornia crassipes*	2.5	0.5	4.7	0.8	7.2	1.3
	LM99	*Lemna* sp.	0.0	5.5	0.0	7.7	0.0	13.2
	PS01	*Pistia stratiotes*	2.0	7.6	4.7	5.3	6.7	12.9
	SC05	*Scirpus cubensis*	0.0	10.3	0.0	5.5	0.0	15.8
	SM01	*Salvinia minima*	0.0	20.8	0.0	7.7	0.0	28.5
	WG01	*Wolffiella gladiata*	0.0	2.9	0.0	5.2	0.0	8.1

Relative cover, relative frequency, and importance values (IV) for species with values >5 in any of these three metrics in either or both the baseline and reference data are shown in Table 7-3; IVs are graphed alone for the reference and baseline data in Figure 7-5. Six of the species on this list are floating/mat-forming species, including the tiny floating, aquatic fern, *Salvinia minima* (water spangles), which had the highest IV in the baseline period data. Two other small-leaved floating plants, *Wolffiella gladiata* (watersprite), and *Lemna* sp. (duckweed) occurred with lower IV. Also on this list (and present in both data sets) were *Eichhornia crassipes* (water hyacinth) and *Pistia stratiotes* (water lettuce), both floating, invasive exotics, and the only floating species recorded in the reference data. Several floating and mat-forming species, including *Scirpus cubensis*, *S. minima, Lemna* sp., and *W. gladiata* were present in the baseline data, but not in the reference data.

Common emergent species in the baseline and reference data were *Nuphar lutea* (spatterdock), *Polygonum densiflorum* (smartweed), the native grass *Panicum hemitomon* (maidencane), *Alternanthera philoxeroides* (alligatorweed), *Hydrocotyle umbellata* (pennywort), and the shrub *Ludwigia peruviana* (Peruvian primrosewillow).

Mean species richness of transect sections in the Impact area was 12.6 species (Table 7-2). In the Reference data, richness was only 4.5 species. Species richness was significantly different between the reference and baseline data (P <0.01, Kruskal-Wallis one-way analysis of variance on ranks) (Table 7-2). All species encountered in the baseline surveys are listed in Appendix 7-1A.

Non-native Invasive Species

Native species had lower mean relative cover (74.4% ± 4.6%) in the baseline data than in the reference area (95.5% ± 2.0%) (Figure 7-6) (Kruskal-Wallis one-way analysis of variance on ranks, P <0.001) (Table 7-2). Free-floating invasive non-natives (primarily *E. crassipes* and *P. stratiotes)* were present in both data sets but occurred in only about 10% of transect sections in the reference data, and 6% of transect sections in the baseline data. However, combined mean relative cover of invasive non-natives was not high (5–10% in both datasets).

DISCUSSION

Impacts of Channelization on Littoral Vegetation Beds

The substantial differences between flowing and non-flowing channels in the same system suggest that elimination of flow was a factor allowing expansion of littoral vegetation beds toward mid-channel areas, and increases in cover of floating and mat-forming species relative to cover of emergent species.

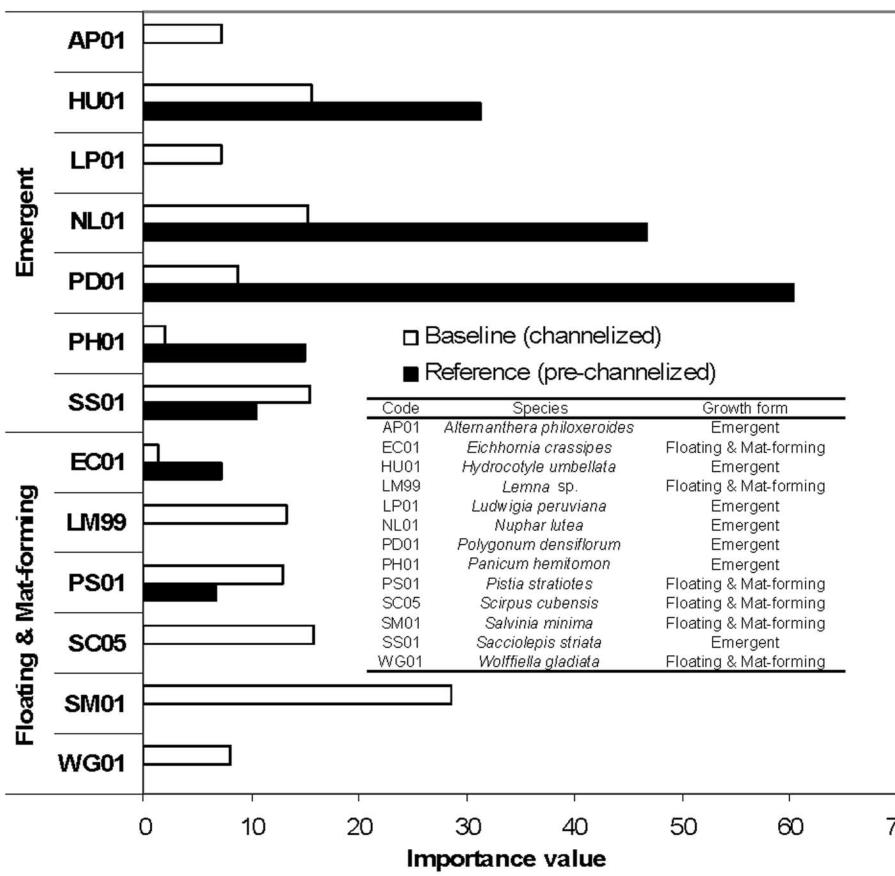

Figure 7-5. Common (IV ≥ 5) species in baseline (channelized) and reference (flowing) area remnant channels. Error bars represent ± one standard error of the mean.

The lower mat widths and cover of floating species observed under flowing conditions are likely the result of combinations of (a) gross removal of free-floating species by flow, particularly in unsheltered areas of the channel; (b) removal of parts of or entire floating mats by flow; and (c) undermining of substrates by flow. In flume experiments, Riis and Biggs (2003) found that removal of emergent macrophytes by high flow was primarily due to uprooting of species resulting from erosion of substrate sediments, rather than stem breakage. Conversion to emergent-dominated communities is likely also partly due to removal of floating species. Because relative cover values are being used, reductions in floating and mat-forming species alone could cause an increase in relative emergent cover without an increase in absolute cover. Changes in species composition can be interpreted as biological responses of species adapted to particular ranges of tolerance in flow.

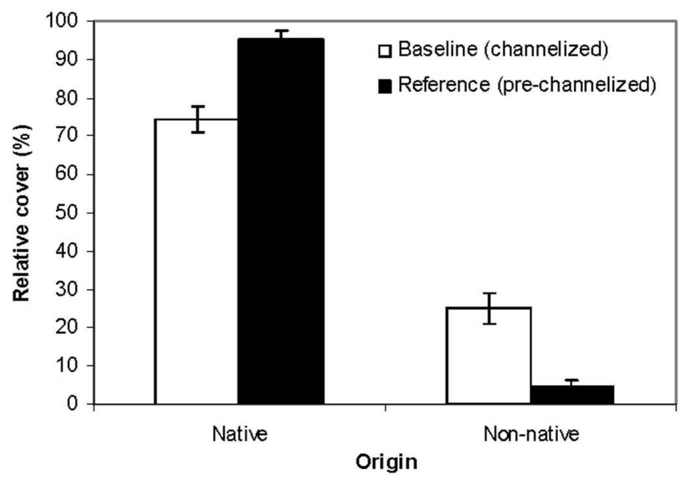

Figure 7-6. Mean relative cover of native and non-native species in the baseline and reference littoral vegetation surveys. Error bars indicate ± one standard error of the mean.

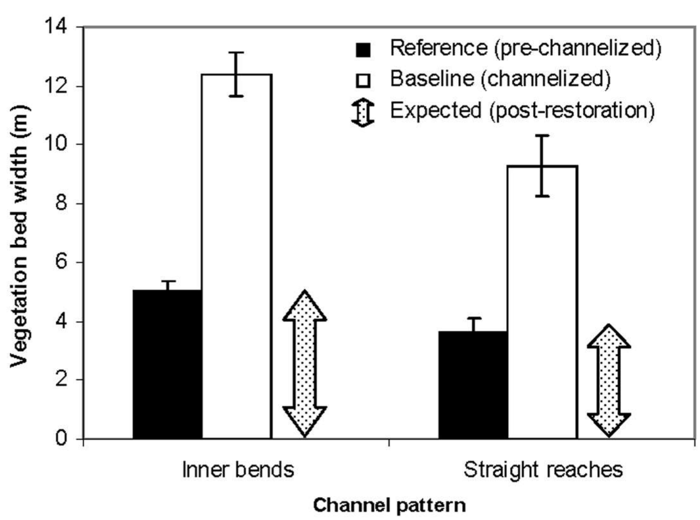

Figure 7-7. Mean littoral bed widths on inner bends and straight reaches of river channel in the baseline and reference littoral vegetation surveys, showing values expected following restoration of flow based on reference data. Error bars indicate ± one standard error of the mean.

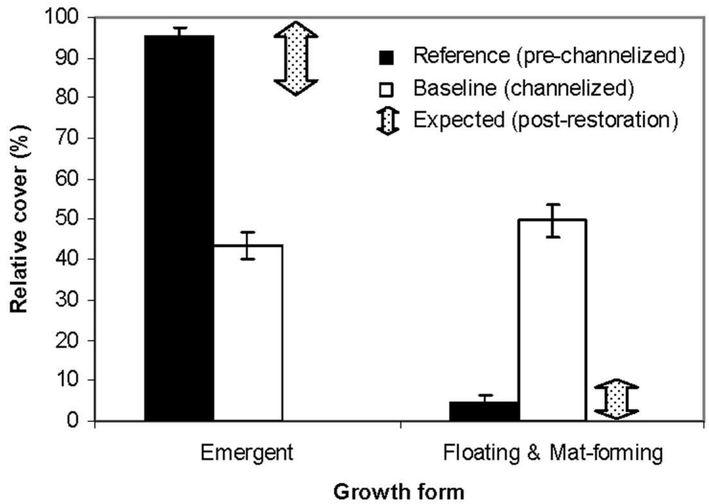

Figure 7-8. Mean relative cover of emergent and floating and mat-forming species in the baseline and reference littoral vegetation surveys, graphed with values expected following restoration of flow. Error bars indicate ± one standard error of the mean.

Expectations

As part of the restoration evaluation program, performance measures called expectations were developed for selected metrics for which (a) good reference data exist or can be extrapolated from remote but similar sites; and (b) are anticipated to show clear, measurable, and ecologically meaningful responses to restoration. Littoral plant community structure and the width of vegetation beds were suggested by the data presented in this report as expectation metrics and have been selected as restoration expectations. More details on expectations than are presented below are available in Bousquin and Hovey 2005a and 2005b.

Vegetation Bed Widths Relative to Channel Pattern

The expectation for vegetation bed widths (Bousquin and Hovey 2005a) was based on reference vegetation bed width data for inner bends and straight reaches of channels; an expectation was not developed for outer bends because widths were not significantly different between the reference and baseline data (Table 7-2). The expectation predicts that, following restored flow, littoral vegetation beds will persist in restored river channels, but that their mean widths will decrease to five meters or less from the bank on inner channel bends, and four meters or less from the bank on straight channel reaches (Figure 7-7).

Littoral Community Structure

The expectation for littoral plant community structure (Bousquin and Hovey 2005b) was also based on the reference data presented in this report. Mean relative cover of both emergent species and floating/mat-forming species differed significantly between the reference and baseline areas (Table 7-2). The expectation predicts that, following restored flow, littoral plant community structure will undergo the following changes: (a) combined mean relative cover of emergent species will increase to >80%, and (b) combined mean relative cover of floating and mat-forming species will decrease to <10% (Figure 7-8).

Non-native Species

As discussed in the Methods section, invasive exotics, primarily *Pistia stratiotes* and *Eichhornia crassipes*, have been maintained at low levels since 1988. This information is consistent with the baseline and reference results, which reveal low relative cover and relative frequency of invasive exotics overall (Figure 7-5). Substantial mats of these species were nonexistent in both the baseline and reference data sets, likely because of weed control efforts, although infestations prior to vegetation management efforts were reportedly common (see Study Areas in the Methods section, above). Because management efforts have maintained low constant levels of these target species, the effects of weed control are best viewed as a constant background factor that is not likely to affect future data collection or evaluation analyses.

It is probable that, in the absence of weed control, these free-floating invasives would have proliferated under non-flowing conditions, and anecdotal pre-weed control information cited in the Methods section supports this idea. Had weed proliferation gone unchecked, reintroduction of flow would likely have resulted in even more pronounced contrasts between flowing and non-flowing channels than those demonstrated in this report.

Both *P. stratiotes* and *E. crassipes* were present prior to channelization, but extensive mats were likely limited to backwaters, abandoned meanders, and edges of active channels. Although *Scirpus cubensis* is not native to Florida, and could be considered both a nuisance and invasive in the Kissimmee system because of its presumed role in floating mat formation (Milleson et al. 1980), it is not generally regarded as invasive (Florida Exotic Pest Plant Council 2003).

Conclusions

The results indicate that the distribution and species composition of littoral plant communities in the Kissimmee River were different in flowing channels compared to non-flowing channels. In the channelized system, aquatic vegetation was likely limited by flow to relatively narrow littoral zones dominated on average by emergent species. With the dredging of C-38 and diversion of flow to the canal, disconnected river channels became non-flowing pools in which cover of vegetation in channels, and relative cover of floating and mat-forming species, probably increased as a result of channelization.

ACKNOWLEDGEMENTS

The author extends grateful acknowledgement to the following individuals who have made significant contributions to the littoral vegetation project since its inception in 1998: Teresa Fretwell, Caroline Hovey, William Patrick Davis, Louis Toth, David J. Colangelo, Kimberly Kanter, Renee Hartless, R. Harper Carroll IV, Don Frei, and Julie Lynch.

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CHAPTER 8

CHANNELIZED KISSIMMEE RIVER FLOODPLAIN VEGETATION

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ABSTRACT: Prior to channelization over 80% of vegetation in what is now Pools A–D of the Kissimmee River occurred in wetlands. By 1974, three years after channelization was completed, over 60% of pre-channelization wetlands had disappeared and upland vegetation covered more than half of the original floodplain's area. The major components of pre-channelization floodplain wetlands were Broadleaf Marshes, Wet Prairie, and Wetland Shrub communities. This chapter describes the composition of these important plant communities; explains the methods used to collect baseline-period species data for future comparison with post-restoration data; and provides estimates of community change that occurred as a result of channelization.

INTRODUCTION

Measurable landscape- and community-level responses to changes in plant habitat, such as alterations in hydroperiod, make plant communities sensitive indicators of environmental change. After channelization of the Kissimmee River was completed in 1971, canal C-38 intercepted and contained virtually all flow formerly carried by the river channel and floodplain. Large areas of floodplain were no longer inundated seasonally, and within a few years a dramatic conversion to a system dominated by upland vegetation had taken place. Prior to channelization, over 80% of vegetation in Pools A–D occurred in wetlands. By 1974, over 60% of pre-channelization wetlands were gone and upland vegetation covered more than half of the original floodplain's area (Carnal and Bousquin 2005). Restoration of the hydrology of the Kissimmee River is expected to restore these wetland plant communities to their original distribution and areal coverage.

As plants respond to changes in habitat, changes in the distribution and composition of plant communities can provide notice of the recovery trajectory and status of restoration (Smart 2000, de Boer 1982). Vegetation is also a mediating factor between hydrology and animal trophic levels. As the distributions and composition of plant communities change, the species of animals that utilize them also change. Declines in animal taxa that depended on pre-channelization vegetation (or other properties of their habitat to which vegetation also responded) are documented in other chapters of this volume (Glenn 2005, Koebel et al. 2005a, b, Williams and Melvin 2005). Vegetation is therefore a direct and powerful indicator of bioic change at other trophic levels in response to hydrologic restoration.

The Kissimmee River Restoration Evaluation Program includes three primary vegetation monitoring components encompassing both large-scale mapping and ground-based studies of floodplain and river channel plant communities. The floodplain vegetation study is an ongoing effort designed to monitor community change as plant community succession takes place in response to restoration of pre-channelization hydrologic characteristics of the river and floodplain. This study is intended to capture species-level information not available from vegetation mapping, which is based primarily on remotely-sensed data (Bousquin and Carnal 2005, Carnal and Bousquin 2005, Shuman and Ambrose 2003). Baseline-period wet season data produced by this study are presented here.

Pre-Channelization Hydrology and Vegetation

Prior to channelization, the Kissimmee River underwent a seasonal cycle of wet and dry periods; however, it is likely that only peripheral areas of the floodplain underwent consistent annual seasonal drying (Koebel 1995). Substantial portions of the floodplain were probably inundated for long periods most years (Toth et al. 1995, Anderson 2005) with maximum water depths ranging from 0.3–0.7 meters (Koebel 1995).

The major components of pre-channelization floodplain wetlands were Broadleaf Marshes (BLM, Bousquin and Carnal 2005) (7060.8 ha), Wet Prairies (WP) (3203.9 ha), and Wetland Shrub communities (WS) (1976.3 ha), which together accounted for over 98% of floodplain wetlands prior to channelization (Carnal and Bousquin 2005).

The following descriptions are based on photointerpretation of pre-channelization aerial photography by Pierce et al. (1982); detailed species data are not available for the pre-channelization period. Broadleaf Marshes dominated by *Sagittaria lancifolia* (arrowhead) and *Pontederia cordata* (pickerelweed) occurred in portions of the floodplain closest to the river, which had the longest and deepest hydroperiods. Broadleaf Marsh communities graded upslope into Wet Prairies as average water depth and hydroperiod decreased. Where they overlapped with Broadleaf Marsh, Wet Prairies on the Kissimmee were dominated by *Panicum hemitomon* (maidencane) (Bousquin and Carnal 2005), which decreased in abundance toward the edges of the floodplain, where various wetland grasses, sedges, and forbs dominated Wet Prairie. Wetland Shrub communities of two major types occurred in areas of long-duration hydroperiod prior to channelization: *Cephalanthus occidentalis* (buttonbush) communities, which form an open canopy in some Broadleaf Marshes, and *Salix caroliniana*- (willow) communities, which occurred primarily in riparian areas.

Objectives

The objectives of this chapter are:

(a) To describe the baseline period species composition and structure of the major wetland plant communities of the Kissimmee River floodplain: Broadleaf Marsh, Wet Prairie, and Wetland Shrub.

(b) To establish baseline conditions for monitoring evaluation of successional change following restoration.

(c) To describe probable impacts of channelization and restoration on Broadleaf Marsh and Wet Prairie plant communities.

METHODS

Field Methods

Baseline sampling was conducted by L. Toth in July–October (wet season) 1998 at 87 5 m x 20 m plots in Pools A and C where Broadleaf Marsh, Wet Prairie, and Wetland Shrub communities had occurred prior to channelization. Plot locations were originally selected in replicate clusters of three, stratified by elevation and pre-channelization vegetation (using the pre-channelization vegetation map of Pierce et al. 1982). Ten of these plots, located on the site of a former sod farm, an unvegetated levee, and a site that was improved pasture prior to channelization, were not used in the analyses presented here. Plots were permanently marked with PVC corner poles. The plots were sampled three times (summer 1998, winter 1998–1999, and spring 1999) prior to backfilling of C-38, which began in June 1999. The 1998 wet season data were used in the analyses presented here because the season of this sample corresponds to that of the available wet season reference data described below (see *Reference Data Methods*).

At each plot, the cover classes of understory plant species were recorded using a six-level system of Daubenmire (Daubenmire 1959) (Table 8-1); canopy cover (>2 m) was recorded using a modified four-class system (Table 8-1). Water depth at plot center and corners was recorded. Plot corners were surveyed to provide accurate ground elevations, which range from 46.3–50.1 feet in Pool A and 31.8–41.4 feet in Pool C.

Table 8-1. Modified Daubenmire scale used for recording cover of plant species within vegetation plots.

Scale	Cover Class	Cover range	Midpoint
Overstory	1	1 - 5	2.5
	2	6 - 50	22.5
	3	51 - 90	70.0
	4	91 - 100	95.0
Understory	0	0	0.0
	1	1 - 5	2.5
	2	6 - 25	15.0
	3	26 - 50	37.5
	4	51 - 75	62.5
	5	76 - 95	85.0
	6	96 - 100	97.5

Data Summary Methods

Community Classification

Baseline plant species cover data from the 77 plots were classified following the decision rules in the Kissimmee River Restoration Evaluation Program (KRREP) Vegetation Classification System (Bousquin and Carnal 2005), using visual examination of plot data and the cluster analysis presented in Bousquin and Carnal (2005) to group plots by compositional similarity in species cover. Communities were keyed to the Bcode Group level of the KRREP Vegetation Classification System, which defines plant communities by the presence of one or more indicator species. The classification defines 73 Community Types (abbreviated as Bcodes), most of which are defined by dominant species or groups of species. For mapping and areal estimation purposes, Community Types are generally grouped into Bcode Groups, which is the second-finest level in the classification. Bcode Groups cluster similar Community Types, which may have different dominant species or different abundances of dominant species, but are similar in their habitat requirements and physiognomy (appearance and shape). Although use of cover classes with large ranges inhibits data classification using cover-based decision rules (because actual species cover values could be at the upper or lower end of the range), most classification decisions simply involved determining dominant species. Decisions were checked against the data collector's original classifications, which had been made on-site. Cases in which decisions were unclear involved distinctions between Broadleaf Marsh and Wet Prairie, which clustered together in the cluster analysis and are similar in species composition.

In most cases, because precise criteria of the classification were not always met for Broadleaf Marsh and Wet Prairie, some choices between these types were judgment-based, primarily using presence of indicator species. Full species lists for all plots are presented in Appendix 8-1A. During the baseline period, Broadleaf Marsh and Wet Prairie community compositions may be transitional or intermediate between both types of marsh, possibly due to an unsuitable flooding regime that causes repeated setbacks in succession. While both types could have been classified as an intermediate type (e.g., Miscellaneous Wet Prairie Vegetation, MxWP in Bousquin and Carnal 2005), their designations as "transitional" better-clarifies relationships to the more robust broadleaf and wet prairie marshes believed to have occurred prior to channelization (Toth, unpublished).

The midpoints of cover classes (Table 8-1) (Daubenmire 1959) were used for calculation of relative cover. For species with both understory and canopy cover in a plot, the strata were combined by selecting the stratum with the highest midpoint and using that number for relative cover calculations. Although coarse, this method allowed estimation of relative dominance and species composition.

Species Wetland Status Classification

Wetland affinities of plant species were classified using the wetland status categories of the National Wetlands Inventory (NWI) (U. S. Fish and Wildlife Service 1988) (Table 8-2). Relative frequencies for these categories were calculated as the number of occurrences in a plot of species in a status category divided by the number of species in the plot (expressed as a percentage). Two combined categories were used in analyses: a) relative frequency of obligate + facultative wetland species, and b) relative cover of facultative + facultative upland + upland species.

Table 8-2. Wetland status categories (adapted from U. S. Fish and Wildlife Service 1988). The last category was defined for this study.

OBL	Obligate wetland. Occurs almost always (estimated probability 99%) under natural conditions in wetlands.
FACW	Facultative wetland. Usually occurs in wetlands (estimated probability 67%-99%), but occasionally found in non-wetlands.
FAC	Facultative. Equally likely to occur in wetlands or non-wetlands (estimated probability 34%-66%).
FACU	Facultative upland. Usually occurs in non-wetlands (estimated probability 67%-99%), but occasionally found in wetlands (estimated probability 1%-33%).
UPL	Obligate upland. Occurs almost always in uplands under natural conditions.

Reference Data Methods

A digitized version of the Pierce et al. (1982) pre-channelization vegetation map was used to determine the pre-channelization vegetation types of the floodplain vegetation plots. Pierce et al. (1982) photointerpreted 1952–1954 1:8000 black-and-white aerial photographs for their maps of early post-channelization vegetation, applying categories that had been established by Milleson et al. (1980) where possible and defining new categories as needed. Milleson et al. (1980) had mapped early post-channelization Kissimmee floodplain vegetation based on 1973–1974 photography. Because the vegetation mosaic on which Pierce et al.'s map was based no longer existed at the time of mapping, direct ground-truthing was not possible. However, the authors had the benefit of Milleson et al.'s (1980) recent photointerpretation and ground-truthing of post-channelization vegetation. The original categories of Pierce et al. (1982) and Milleson et al. (1980) were converted to the KRREP Vegetation Classification System for compatibility with Kissimmee River Restoration baseline data. The entire classification, details of conversion decisions, and a crosswalk among the three classification systems are presented in Bousquin and Carnal (2005). Species abundance data are not available in either classification.

Estimates of pre-channelization community structure in Broadleaf Marsh and Wet Prairie are from L. Toth (unpublished reports and data), who used species data collected in July–November (wet season) of 1984–1994 in 1 m^2 quadrats placed at 7.6 m intervals on transects in inundated sections of Pools A and B to estimate pre-channelization species composition and frequency of NWI categories for Wet Prairie and Broadleaf Marsh. Broadleaf Marsh reference data were collected at "remnant broadleaf marsh" transects (Deer Run North, Deer Run South, and Turkey Trail) in the impounded lower portion of Pool B, where floodplain elevations had received long (usually $\geq$250 d) annual hydroperiods since channelization. Wet Prairie reference data were collected at transects in reestablished Wet Prairie communities in Pool B (Pine Island Slough and Duck Slough) and an impoundment in Pool A (Rattlesnake Hammock). Percentage of species in NWI categories was calculated for each quadrat in each transect, then averaged for each transect (Toth, unpublished).

RESULTS

Baseline and Pre-channelization Conditions

Community Change Following Channelization

By 1998, the plant communities that had occurred at the vegetation plots prior to channelization had developed into one of three general vegetation types: (1) Upland Herbaceous communities, primarily pastures (Bcode Group UP, Bousquin and Carnal 2005); (2) upland or mesophytic shrub communities (Upland Shrub, US); or (3) transitional Broadleaf Marsh /Wet Prairie (BLM/WP or MxWP) communities. Of the 28 locations that were classified by Pierce et al. (1982) as Broadleaf Marsh prior to channelization in the 1950s, only six plots at lower elevations (Table 8-3) persisted as wetland vegetation. These plots were at various stages of succession and all seemed intermediate between Broadleaf Marsh-Wet Prairie communities in the channelized-period baseline data (Appendix 8-1A). The remaining 22 formerly Broadleaf Marsh plots had become either Upland Herbaceous or Upland Shrub communities by 1998 (Table 8-3). All of the 28 pre-channelization Wet Prairie sites had developed upland vegetation, either herbaceous (primarily improved pastures) or shrub-dominated (Table 8-3). Of the 21 locations classified by Pierce et al (1982) as wetland shrub vegetation prior to channelization, five plots persisted as wetlands (either Wet Prairie or Broadleaf Marsh) after channelization and one was classified as a Miscellaneous Wetland (MW, the Bcode Group in which fern-dominated communities are grouped — essentially similar to a *Myrica cerifera* floating mat community (S.MCF, Bousquin and Carnal 2005) but with lower cover of shrubs and higher cover of *Osmunda regalis*). However, most (15) of the formerly wetland shrub plots had developed either herbaceous or shrub-dominated upland vegetation (Table 8-3).

Table 8-3. Baseline (channelized condition) vegetation plots, arranged by pre-channelization vegetation.

Pre-channelization vegetation	Baseline vegetation	Pool	Mean elevation (ft)	SE (ft)	n	Elevation range (ft)
Broadleaf Marsh	Broadleaf Marsh	A	46.3	0.0	3	46.3 - 46.4
		C	33.9	0.1	2	33.8 - 33.9
	Upland Herbaceous (pasture)	A	48.7	0.2	6	48 - 49.3
		C	36.8	0.7	7	35.4 - 41.1
	Upland Shrub	A	47.3	0.3	3	46.6 - 47.7
		C	34.1	0.2	6	33.2 - 34.6
	Wet Prairie	C	33.6	0.0	1	33.6 - 33.6
Wet Prairie	Upland Herbaceous (pasture)	A	49.7	0.1	6	49.3 - 50.1
		C	38.2	0.4	17	35.7 - 41.4
	Upland Shrub	C	34.7	0.3	5	34.3 - 35.4
Wetland Shrub	Broadleaf Marsh	C	35.5	0.1	2	35.3 - 35.6
	Miscellaneous Wetland	C	31.8	0.0	1	31.8 - 31.8
	Upland Herbaceous (pasture)	A	48.2	0.1	3	48.1 - 48.3
		C	35.6	0.0	1	35.6 - 35.6
	Upland Shrub	A	47.6	0.0	3	47.3 - 48
		C	35.5	0.8	8	31.8 - 37
	Wet Prairie	C	35.3	0.2	3	34.9 - 35.5

Baseline Community Structure and Species Composition

Obligate and facultative wetland species in baseline-period examples of Broadleaf Marsh in Pools A and C had combined relative cover of 93.6% ± 1.8%, and relative frequency of 84.9% ± 4.2% of species. Except by presence of Broadleaf Marsh indicator species (*Sagittaria lancifolia* and *Pontederia cordata*), distinctions between baseline examples of Broadleaf Marsh and Wet Prairie were difficult; Wet Prairie had relative cover of obligate and facultative wetland species of 93.9% ± 1.5% and relative frequency of 85.5% ± 2.3% (Figure 8-1). Percentage of obligate and facultative wetland species in Wet Prairie and Broadleaf Marsh communities were not significantly different in the baseline data (t-test, $P = 0.453$).

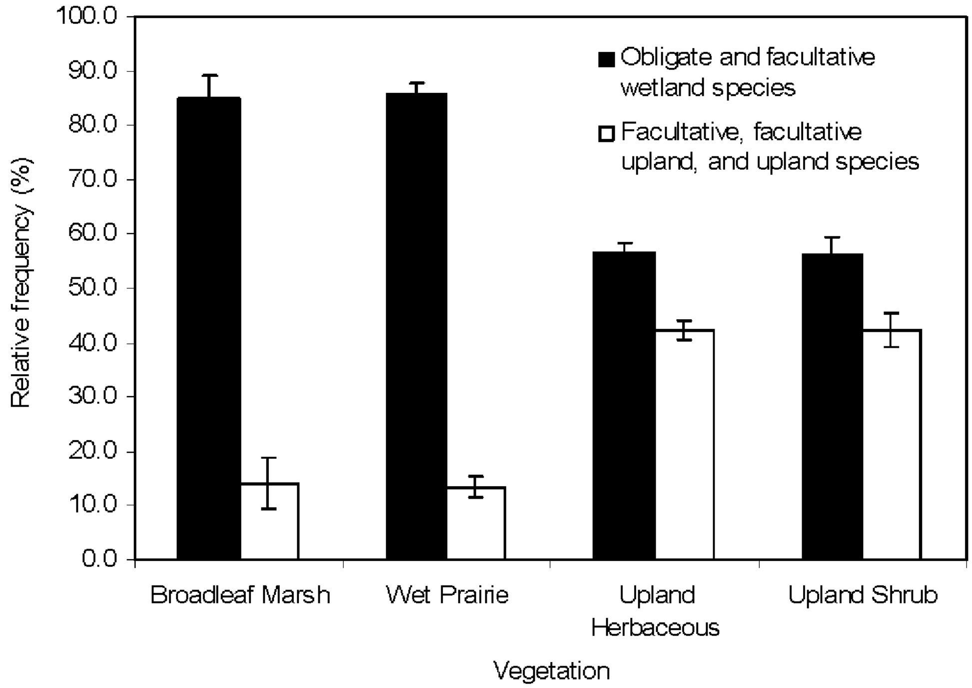

Figure 8-1. Relative frequency of species wetland status categories in the baseline vegetation plots, summer 1998 data.

Commonly associated species in Broadleaf Marshes, in addition to *S. lancifolia* and *P. cordata*, included *P. hemitomon, Leersia hexandra* (cutgrass), *Luziola fluitans* (water grass), *Diodia virginiana* (buttonweed), *Bacopa caroliniana* (bacopa), *Polygonum punctatum* (dotted smartweed), *Cyperus haspan* (sharp-edge sedge), and *Hydrocotyle umbellata* (pennywort). Composition of Wet Prairies was similar and included *L. fluitans, L. hexandra, D. virginiana, P. hemitomon, Eleocharis vivipara* (vivparous spikerush), *Ludwigia peruviana* (Peruvian primrosewillow), *S. lancifolia*, and *Paspalum dissectum* (mudbank crowngrass). Species richness in Broadleaf Marshes and Wet Prairies were 22.0 ± 2.8 species and 22.8 ± 0.9 species, respectively (not significantly different, t-test $P = 0.8$).

The Upland Herbaceous (UP) plots were dominated by combinations of pasture grasses including *Paspalum notatum* (bahiagrass), *Cynodon dactylon* (Bermudagrass), and *Axonopus fissifolius* (carpetgrass). Associated species in some plots included *Sesbania vesicaria* (bladderpod) and *Eupatorium capillifolium* (dogfennel) (Appendix 8-1A), two forbs that tend to increase in ungrazed pastures. On average, obligate and facultative wetland species accounted for 33.9% ± 3.2% of species cover in Upland Herbaceous communities in Pools A and C combined, and made up 56.5% ± 1.9% of the species composition (Figure 8-1).

The Upland Shrub sites had canopies dominated by *Myrica cerifera* (waxmyrtle), *Baccharis halimifolia* (saltbush), or *Rubus cuneifolius* (blackberry); several of each subtype had substantial cover of *Vitis rotundifolia* (grape), a climbing vine (Appendix 8-1A). Understories often included ferns

(*Woodwardia virginica, Blechnum serrulatum, Osmunda regalis, Thelypterus interrupta*) and shrubs (*B. halimifolia, L. peruviana, M. cerifera*). Ten of the 25 shrub sites contained *S. lancifolia* but none contained *P. cordata.* Obligate and facultative wetland species comprised a mean of 37.1% ± 4.0% of plant cover and had relative frequency of 56.3% ± 3.3% of species composition in the Upland Shrub sites (Figure 8-1). Although all of the shrubs listed in Figure 8-1 are upland or mesophytic species, most of the *M. cerifera* sites are located on semi-buoyant floating mats. These mats, formed of dead *Scirpus cubensis* (Cuban bullrush) and other debris, create boglike substrates that tend to occur in the impounded lower sections of pools (Pierce et al. 1982).

Vegetation and Elevation

In Table 8-4, the vegetation plots are ordered by mean elevation of baseline vegetation groups (stratified by pool). Plots are sorted in the order of (from lowest to highest elevations) (1) Broadleaf Marsh, (2) Wet Prairie, (3) Upland Shrub, and (4) Upland Herbaceous communities.

Table 8-4. Baseline vegetation plots in ascending order of mean elevation (feet, NGVD29), (stratified by pool).

Pool	Baseline vegetation	Mean elevation	SE	Range	n
A	Broadleaf Marsh	46.3	0.03	46.3 - 46.4	3
	Upland Shrub	47.4	0.19	46.6 - 48.0	6
	Upland Herbaceous (pasture)	49.0	0.18	48 - 50.1	15
C	Miscellaneous Wetland	31.8	0.00	31.8 - 31.8	1
	Broadleaf Marsh	34.7	0.47	33.8 - 35.6	4
	Wet Prairie	34.9	0.44	33.6 - 35.5	4
	Upland Shrub	34.9	0.37	31.8 - 37.0	19
	Upland Herbaceous (pasture)	37.7	0.34	35.4 - 41.4	26

Reference Data Results

Pre-channelization Community Structure and Composition

Toth's (unpublished) estimates of pre-channelization reference conditions included mean quadrat percentage composition of obligate and facultative wetland species for eight Broadleaf Marsh and eight Wet Prairie transect samples collected in reference locations in Pools A and B from 1988–1997. Sample means of obligate and facultative wetland species were 98.4% ± 0.3% for Broadleaf Marsh and 93.0% ± 1.7% for Wet Prairie. The mean percentage of obligate and facultative wetland species in reference Broadleaf Marshes was significantly higher than in reference Wet Prairie (t-test, $P = 0.008$).

Common species from Toth's data are similar to those reported for the baseline period. Broadleaf Marshes in the reference data included *S. lancifolia, P. cordata, P. hemitomon, L. hexandra, Sacciolepis striata, Alternanthera philoxeroides, Nuphar lutea* (spatterdock), *Polygonum punctatum, B. caroliniana, H. umbellata, C. occidentalis* and *Ludwigia peruviana.* For Wet Prairie, Toth listed *P. hemitomon, L. hexandra, L. fluitans, A. philoxeroides, B. caroliniana, B. monnieri, Centella asiatica, Diodea virginiana, Hydrocotyle umbellata, Polygonum punctatum,* and several species of Cyperaceae (*Carex, Cyperus, Eleocharis, Fimbristylis, Juncus, Rhynchospora* and *Scleria* species). Species richness in Broadleaf Marshes and Wet Prairies in the reference data were 39.9 ± 4.1 species and 61.5 ± 4.8 species, respectively. Richness in Wet Prairie was significantly higher than in Broadleaf Marsh in the reference data (t-test $P = 0.002$).

Comparisons with Baseline Conditions

Mean percentage of obligate and facultative wetland species was significantly higher in the reference data than in the baseline data both for Broadleaf Marshes (t-test, P = 0.010) and Wet Prairies (t-test, P = 0.020) (Figure 8-2). Species richness in both Broadleaf Marsh and Wet Prairie were significantly higher in the reference data than in the baseline data (t-tests, P = 0.002 and P <0.001, respectively) (Figure 8-3). However, direct comparisons between the baseline and reference data may be misleading because of differences in baseline and reference data collection methods.

DISCUSSION

Baseline period sample sizes were small for Broadleaf Marsh and Wet Prairie, but Table 8-4 suggests that community elevational distributions in the baseline data correspond with known relationships and responses of wetland plant communities to inundation. Vegetation distributions on river floodplains are a function of spatial and temporal variation in the distribution, depth, and duration of water (Blom et al. 1990, Lowe 1986). In floodplain habitats, stage, floodplain topography and slope, and temporal/seasonal variation in water levels all affect which species can establish and persist (Blom et al. 1990, Lowe 1986, Welcomme 1979). Broadleaf Marsh (or flag marsh, Kushlan 1990) requires extended, near-permanent inundation. Marshes of this kind are classified in Anderson et al. (1998) as semipermanently flooded ("surface water persists throughout growing season in most years, except during periods of drought; soil surface is normally saturated when water level drops below soil surface"). The Anderson et al. (1998) category includes Cowardin et al.'s (1979) water regime modifiers "Intermittently Exposed" and "Semipermanently Flooded". According to Kushlan (1990), broadleaf marshes require hydroperiods greater than 200 d·yr^{-1} and wet season water depths between 0.3 m and 1 m; however, despite this dependence on flooding, they also require seasonal drying (Kushlan 1990). "Flag" species (*P. cordata* and *S. lancifolia*) communities and *P. hemitomon* marshes (considered to grade into Wet Prairie by Bousquin and Carnal 2005) tend to become less common where seasonal drawdowns are eliminated (Kushlan 1990).

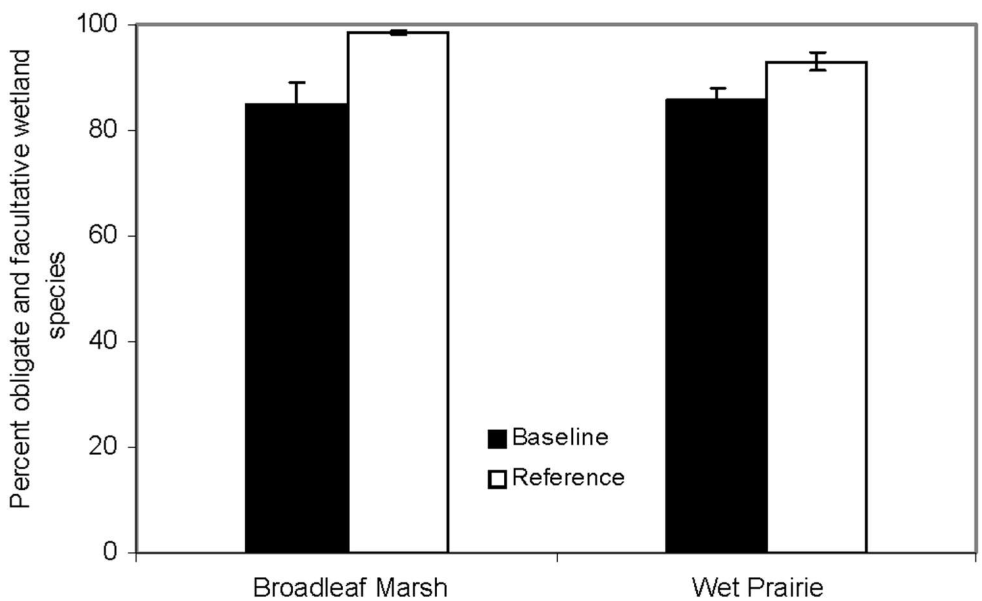

Figure 8-2. Relative frequency of obligate wetland and facultative wetland species in the baseline and reference data.

Wet Prairies have shorter hydroperiods than Broadleaf Marshes, with estimates ranging from 50–100 d·yr^{-1} year (Kushlan 1990) to 180–330 d·yr^{-1} for *P. hemitomon*-dominated marshes (Anderson et al. 1998) and tend to occur between the elevations of deeper marshes and surrounding uplands. Wet Prairies are adapted to fire and may be dependant on burning to inhibit invasion by shrub species (Wade et al. 1980).

Although precise information on the spatial and temporal distribution of water depths prior to or following channelization of the Kissimmee is not available, there is no doubt that water was present at greater depths over larger areas, and for longer periods of time, prior to channelization. Anderson (2005, this volume) shows that stages had a greater range and that high stages occurred with greater frequency prior to channelization than after channelization. Toth (1995) compared elevations in two 1 mi^2 areas of floodplain with pre-channelization stage data to estimate frequencies of areal inundation (Toth 1995); these estimates suggest that most of the area studied was flooded most of the time prior to channelization, with maximum water depths ranging from 0.3–0.7 meters (Koebel 1995). Finally, changes in the distribution of plant communities (Carnal and Bousquin 2005) considered with these changes in hydrology are a strong indication that alterations in community distributions after channelization were in response to changed floodplain inundation patterns. This change in inundation involved less extensive inundation, but also less variation in the levels and distribution of water (Anderson 2005).

As mentioned above, baseline period Wet Prairie and Broadleaf Marshes had not separated well in the cluster analysis (Figure 8-1 in Bousquin and Carnal 2005, this volume) due to overlap in species composition, often with respect to *Panicum hemitomon* (maidencane) (Appendix 8-1A), which is characteristic of both marsh types and is used in Bousquin and Carnal (2005) to distinguish Broadleaf Marsh from Wet Prairie. Percentages of obligate and facultative wetland species were similar in baseline Broadleaf Marsh and Wet Prairie communities, contrasted with significantly higher percentages of obligate and facultative wetland species in reference Broadleaf Marshes than in reference Wet Prairies. That is, baseline-data Wet Prairies were more similar to baseline Broadleaf Marsh than Wet Prairies were to Broadleaf Marsh in the reference data, and had fewer upland-adapted or faculatative species relative to baseline Broadleaf Marsh (Figure 8-2), likely as a result of less varied inundation regimes. This conclusion is consistent with Anderson’s (2005) results of comparisons of baseline and pre-channelization hydrology.

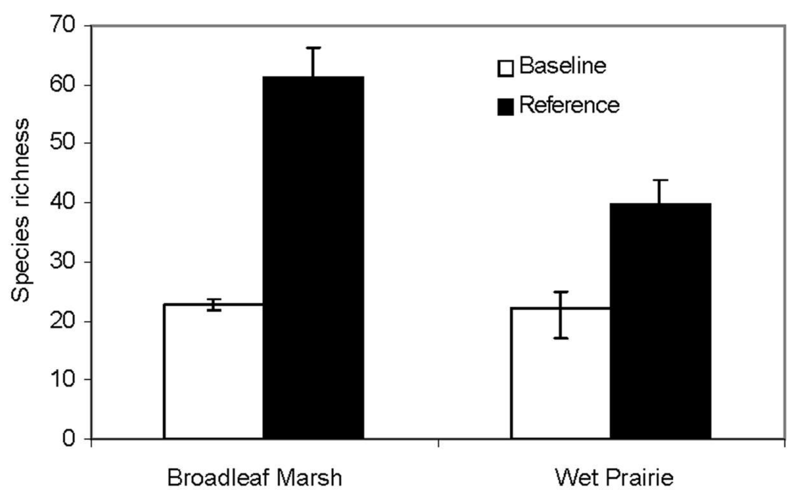

Figure 8-3. Species richness in the baseline and reference data.

The distribution of wetland plant communities has clearly changed since channelization (Carnal and Bousquin 2005). The results of the cluster analysis of the baseline data in Bousquin and Carnal (2005), and the difficulty of discriminating Broadleaf Marsh and Wet Prairie manually, suggest that, following channelization, the composition of communities dominated by Wet Prairie indicator species also changed — and to some extent converged with Broadleaf Marsh — making these types less distinct as identifiable assemblages of species than they were prior to channelization. Intermediate communities like these have likely always existed where these marsh types overlap, but the fact that few clear examples of either type occurred in the baseline data is notable. Lower species richness in baseline communities than in reference communities, particularly in Wet Prairie (Figure 8-3), also suggests that a complement of species was lost. These results are consistent with findings that areas with less varied flooding regimes tend to have lower plant species richness than wetland habitats exposed to periodic drying (Gerritsen and Greening 1989, Mitsch and Gosselink 1986, Keddy and Reznicek 1986, Conner et al. 1981).

ACKNOWLEDGEMENTS

The author gratefully acknowledges the work of Lou Toth, who designed and initiated this study, collected the data presented here, and produced the internal documents on which much of this chapter was based.

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CHAPTER 9

CLASSIFICATION OF THE VEGETATION OF THE KISSIMMEE RIVER AND FLOODPLAIN

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ABSTRACT: The Kissimmee River Restoration Evaluation Program (KRREP) Baseline Vegetation Classification was developed to characterize plant assemblages that occur in the Kissimmee River, its floodplain, and included uplands. The classification provides definitions and decision rules to facilitate consistent and repeatable description of vegetation at several scales of resolution by photointerpreters and field data collectors. The classification was developed using photointerpretation of 1996 Pool C aerial photography, two previous classifications of the Kissimmee River area that were developed prior to and immediately following channelization, and field data. This document focuses on the background of the classification, definitions of vegetation categories, methods used to derive the categories, decision rules for their application to vegetation data, and linkages with previous classifications of the Kissimmee River area. A partial assessment of the classification as applied to field vegetation data also is presented.

INTRODUCTION

The Kissimmee River Restoration Evaluation program (KRREP) Vegetation Classification System provides definitions and decision rules to facilitate consistent, repeatable descriptions of river and floodplain plant assemblages for use by photointerpreters and field data collectors during the course of the KRREP. Because groups of species with similar habitat requirements tend to co-occur and recur across the landscape, an appropriate level of detail for classification categories can be determined for the specific objectives of a management, monitoring, or ecological research project or study. This report will focus on descriptions and definitions of vegetation types, methods that were used to derive the categories, decision rules for their application to vegetation data, and linkage of the classification with previous classifications that have been used in the KRREP. An assessment of the classification as applied to field data also will be presented.

Although previous classification systems existed to describe the Kissimmee River and its floodplain, the channilization of the river by the construction of the C-38 canal and the initiation of a major restoration project on the river necessitated a new vegetation classification better-suited to post-channelization conditions than the previous systems. This need was based on three primary considerations. First, much of the gradient and seral vegetation found in post-channelization conditions was not defined in the previous classifications, either because it was not present under pre-channelization and early post-channelization conditions, or was not captured at the scale of delineation used by these mapping projects. Second, KRREP staff wanted to classify photointerpretation data at a more detailed level than would be possible using the previous classifications, to retain the option of tracking vegetation change at as fine a scale as was possible using photointerpretation. Third, aquatic vegetation was not well-represented in the previous classifications.

Objectives

The objective of the classification is to characterize vegetation that occurs in the Kissimmee River, its floodplain, and included uplands during baseline (post-channelization, pre-restoration) conditions. This is accomplished by synthesis of species-level data into ecological categories at several levels of resolution intended to meet the needs of KRREP vegetation mapping, data analysis, and other data collection and analysis efforts.

Previous Vegetation Classifications of the Kissimmee River and Floodplain

Pre-existing Classifications

Pierce et al. (1982) used 1952-1954 aerial photography to map pre-channelization vegetation on the Kissimmee River floodplain. Their mapping categories were based on a previous classification developed for early post-channelization conditions by Milleson et al. (1980). Categories of these classifications are shown in Tables 9-1 and 9-2.

Table 9-1. Categories of the Milleson et al. (1980) classification.

Agriculture and Urban	Improved Pasture	
	Unimproved Pasture	
	Citrus	
	Urban	
Terrestrial Forested	Oak and Cabbage Palm	
	Wax Myrtle	
	Woody Shrub	
Wetland Forested	Willows (In Floodplain)	
	Willows (In Spoil Areas)	
	Hardwood Trees	
	Cypress	
Marsh	Broadleaf Marsh	
	Maidencane Wet Prairie	
	Rhynchospora Wet Prairie	
	Aquatic Grasses	
	Buttonbush	
	Primrose Willow	
	Floating Tussocks	
	Switchgrass	
	Soft Rush Ponds	
	Sawgrass	
	St. Johns Wort	
Spoil and Barren	Spoil	
	Vegetated Spoil	
	Levees	
	Open Water	C-38
		Kissimmee River

Both classifications used dominant physiognomy (tree, shrub, or herbaceous) and descriptions of habitat of communities (e.g., wetland) to delineate upper hierarchical categories. Both also used dominant and associated species to define and distinguish vegetation categories. Although the term "dominance" was not explicitly defined in either document, the term was used in community descriptions, and Pierce et al. established a cutoff of 30% overstory cover to distinguish shrub and forest from herbaceous categories; however, Milleson et al. did not do so explicitly. Other than this physiognomic-level rule in Pierce et al., quantitative decision rules to distinguish vegetation categories were not stated in either classification.

Vegetation categories in both classifications were usually named for scientific or common names of dominant or prominent species, e.g., "buttonbush" for *Cephalanthus occidentalis*-dominated marshes, although some categories were named with descriptive, vernacular terms for general types of vegetation (e.g., "Broadleaf Marsh") or for topographic features that tend to contain distinctive vegetation (e.g., "wet depression").

Table 9-2. Categories of the Pierce et al. (1982) classification.

Category		Code
Human Influenced	Improved Pasture	PI
	Unimproved Pasure	PU
	Cultivated (in use)	CU
	Cultivated (abandoned)	CA
	Artificial Pond	AP
	Spoil	SP
	Canal	CN
	State Road	SR
Native Upland	Oak/Cabbage Palm	OK
	Pine Forest	PP
	Palmetto Prairie	PM
	Woody Shrub	WD
Forested Wetland	Cypress Forest	CY
	Wetland Hardwood	MP
Wetland Shrub	St. Johns Wort	SJ
	Willow	WI
	Buttonbush	BB
Emergent Wetland	Broadleaf Marsh	PS
	Wet Prairie	WP
	Maidencane Wet Prairie	MC
	Rhynchospora Wet Prairie	RH
	Sawgrass	CL
	Switchgrass	SW
	Mixed Aquatic Grass	TG
	Wet Depression	DW
Aquatic	Floating Mat	FM
	Floating Tussock	TS
	Kissimmee River	KR
	Oxbow	OX
	Open Water	OW
Miscellaneous	Natural Levee	LR
	Unknown-Submerged	US
	Unknown-Poor Quality Photograph	UN

The Initial KRREP Baseline Vegetation Classification

During 1996-1999, Pool C vegetation was mapped using photointerpretation of 1996 aerial photography of Pool C. Categories applied to map polygons were from an initial classification based on species lists. The species lists included species present with greater than 10% cover in photointerpreted polygons. These lists were structured as a 2-level vegetation classification system (Table 9-3) in which the species lists and their assigned codes (the finest level of the system) were grouped under categories that defined various combinations of species life forms, e.g., the category "Woody Shrub and Trees" (Table 9-3, Category L). Photointerpretation was conducted with a minimum mapping unit (MMU) of 100 m^2 (i.e., for mapping purposes, the smallest areas delineated as polygons and assigned discrete vegetation categories were 100 m^2 or approximately 0.03 acre). Detailed information on the photointerpretation and mapping methods used in the mapping project can be found in Carnal and Bousquin 2005.

The initial classification became unwieldy as species combinations accumulated, ultimately resulting in over 650 discrete categories at its finest level, each of which consisted of a unique combination of species. The system lacked an intermediate level of classification comparable to the vegetation categories used in Pierce et al. (1982), and because its lowest level consisted of species lists with no abundance information, species and overstory dominance could not be determined, preventing linkage with the Pierce et al. classification. The initial classification served the purpose of providing a temporary means to record and summarize photointerpreted species data, but it was retired and replaced in 1999 by the new classification that is the focus of this chapter.

METHODS

Criteria for a New KRREP Baseline Vegetation Classification

The new KRREP baseline vegetation classification was designed to meet the following criteria:

(a) Where possible, it should use categories that could be linked with the pre-channelization classification of Pierce et al. (1982) and the post-channelization classification of Milleson et al (1980).

(b) It should have a hierarchical structure that allows for physiognomic distinctions among plant communities, a basic requirement for linkage with the previous classifications; and allow flexibility for users in choice of resolution.

(c) It should define new vegetation categories where necessary for previously undescribed types of vegetation and should be adaptable to ongoing change.

(d) It should be usable with quantitative decision rules to enhance repeatability among users making classification decisions.

(e) It should retain the level of species detail available from existing KRREP photointerpretation data.

Development of the New Baseline Vegetation Classification

Because the lower level of the initial KRREP baseline classification provided species-presence data for species with greater than 10% cover, we were able to produce presence-absence data tables for most Pool C polygons. Photointerpretation was then used to estimate cover of the previously-recorded species, which was needed to determine dominant physiognomy and dominant species for linkage with the previous classifications.

A set of provisional vegetation categories and decision rules was extrapolated from the classifications of Milleson et al. (1980) and Pierce et al. (1982). The provisional classification initiated our use of decision rules based on overstory cover as a means of distinguishing physiognomic categories, and explicit use of dominance (in some cases cover ranges) of characteristic species to distinguish vegetation categories. Although new names were attached to the provisional categories, all were either best-judgment replications of Pierce et al. or Milleson et al. categories, or new categories that we believed would be needed for representation of baseline plant communities.

As photointerpreted cover data were collected for Pool C, provisional community types were assigned to map polygons. In some cases, existing vegetation was adequately described by the provisional categories and decision rules; in these cases, decisions were straightforward and the provisional categories and decision rules were accepted. Category definitions and distinctions were refined in vegetation team meetings, which addressed problems associated with use of the classification, including those encountered by the photointerpreter in applying the provisional decision rules to current vegetation data. Team classification discussions focused on desired levels of classification detail for mapping, potential for evaluation of vegetation change, description of previously undescribed vegetation, and linkage issues with the previous classifications.

Decisions to adjust the level of detail in the new classification occasionally involved splitting provisional categories to describe gradient, seral, or transitional vegetation. For example, the provisional Broadleaf Marsh category was split into several community types to allow description of vegetation intermediate between Broadleaf Marsh and *Panicum hemitomon* (maidencane) Wet Prairie (*Panicum hemitomon* Herbaceous Vegetation, H.PH), between Broadleaf Marsh and *Cephalanthus occidentalis* (buttonbush) communities (*Cephalanthus occidentalis* shrubland, S.CO), and among these three types. That split resulted in the community types H.PS-PH, H.PS-CO, and H.PS-PH-CO (Appendix 9-1A). New community types were defined as necessary to accommodate baseline and existing floodplain vegetation. For example, a new category was needed to describe *Myrica cerifera* (wax myrtle)-dominated shrublands that occur on floating mat vegetation, because of their prominence in the lower areas of pools in the baseline-period floodplain (*Myrica cerifera* Floating Mat Shrubland, S.MCF).

Linkage with Previous Classifications

A basic goal of the baseline classification is to achieve accurate linkage with the pre-channelization vegetation map of Pierce et al. (1982), which was used to estimate reference vegetation coverage and distribution in the pre-channelization floodplain. The need to characterize vegetation change since early post-channelization conditions also made linkage with the Milleson et al. map important.

Linkage with the previous classifications was not straightforward for two reasons. First, current vegetation patterns have resulted from modification of seasonal flooding over the floodplain and elimination of flow through river channels. Channelization resulted in establishment of communities that were not present in the pre-channelization floodplain, including large areas of previously undescribed transitional and possibly successional communities. Second, because the previous classifications were designed for descriptive use, not for tracking vegetation change, vegetation types were not always clearly defined in either classification, objective decision rules were not provided, and aerial photography signatures were not cataloged; thus, procedures to enhance repeatability among users had not been established.

Although these limitations made linkage of current vegetation with previous classifications difficult, we believe that accurate linkage with the Pierce et al. categories has been maintained, particularly for the most important community types (those that cover large areas and those being used as indicators of restoration success; see Carnal and Bousquin 2005 for communities of particular interest and their expected responses to restoration). Linkage was achieved by careful review of the information available in both previous classifications, personal communication with one author (Gary Pierce), and in some cases by judicious use of assumptions based on Kissimmee staff knowledge of floodplain vegetation.

Although linkage with the previous classifications was difficult, it is unlikely that direct use of either of the previous classifications would have ameliorated problems. Because of their definition vagaries and lack of decision rules, use of the previous classifications would probably be no more accurate than linking them with the new classification developed specifically for baseline conditions.

Assessment Methods: Classification of the Vegplots Data Set

Classification Assessment

Vegetation classification is a task in which discrete categories are imposed on a frequently continuous gradient of species composition; it is uncommon to encounter communities that are unique in species composition. When one or few distinguishing characteristics of communities, such as dominance of a selected species, are used in classification decision rules, questions may arise as to whether distinct communities have been defined, or if some rules merely make unnecessary distinctions between communities that are very similar in species composition but have different dominant species.

Because the goals and applications of classifications vary, whether these issues are important relates more to the purpose of the classification than to its internal validity. Some degree of overlap among vegetation categories will occur in any classification of complex, real-world vegetation data. Although overlap among categories does not intrinsically detract from the utility of a classification, excessive overlap may suggest arbitrarily defined categories that have little ecological meaning. Use of overlapping categories with large data sets can result in poor classification decisions among categories, resulting in "blurring" of categories. When estimates of quantitative values are to be derived from a map based on the classification (e.g., estimates of the area of wetland types in Carnal and Bousquin 2005), results can be affected. Developers and users of the classification should be cognizant of the extent of overlap among categories when a classification is applied to quantitative vegetation data. To address this issue, we used multivariate analysis (cluster analysis) on *a priori*-classified field data set to assess the degree to which distinct groupings of similar communities were predicted by the classification.

For assessment of the classification, a cluster analysis was applied to vegetation data collected in 1998 in Pools A and C by South Florida Water Management District staff (Toth, unpublished summer 1998 "vegplot" data). The data set consists of cover class (Daubenmire 1959) data for all species present in eighty-four 5 m x 20 m plots located in floodplain communities ranging from wetland (Wet Prairie, Broadleaf Marsh) to upland types (pastures, shrub communities).

Each plot was classified using Appendix 9-1A and coded accordingly (Table 9-4). A cluster analysis (using a Euclidian distance measure with Ward's group linkage method) was performed on the data set. Cluster analysis can separate sample data based on measured variables, and is used here to define groups of sample units based on similarities in species composition. Cluster analysis constructs hierarchical groupings based on a similarity matrix by successively grouping similar sample units and resulting clusters. The result is a tree diagram (dendrogram) that shows clusters of plots within which more similarity exists among members than to members of other clusters.

Regional and National Classifications

The hierarchical structure and naming conventions of the classification presented here are modeled loosely on the National Vegetation Classification System (NVCS) (Anderson et al. 1998, Grossman et al. 1998).

The NVCS is used for vegetation mapping by the National Park Service and The Nature Conservancy (TNC), among others. Although most Kissimmee vegetation was poorly described by existing categories of the NVCS, it is anticipated that linkage with this national effort ultimately will be achieved by providing our vegetation data and classification documents to TNC for delineation of new types in the NVCS.

RESULTS

The Classification System

The baseline classification is presented in Appendix 9-1A, the *Key to Bcode Groups and Community Types*, which consists of decision rules in the form of a dichotomous key for determination of community types from species data. The classification and key are arranged hierarchically (Table 9-5).

The status categories group communities by characteristic habitat, e.g., upland, wetland, or aquatic. Physiognomic categories describe the general appearance of communities as forest, shrubland, or herbaceous. Community types (abbreviated as Bcodes), are the finest level of the classification, capturing particular communities as distinguished by dominant species. As demonstrated below in the classification assessment, communities dominated by the same species tend to be similar in species composition. Bcode Groups are groupings of community types into ecologically meaningful categories (for example, Wetland Forest or Broadleaf Marsh), at a hierarchical level between community type and Physiognomy. Bcode Groups will be used for more generalized vegetation mapping products than the community type level.

Community types are usually named for the dominant species. In cases when two or more species are used to define communities, species names are separated by hyphens in the community type name. Hyphens used in this way indicate a community type that is intermediate between two more distinct community types, e.g. *Pontederia cordata-Sagittaria lancifolia-Panicum hemitomon* Herbaceous Vegetation (H.PS-PH). When a second species may or may not be present, the second species is enclosed in parentheses (e.g., *Quercus virginiana* (*-Sabal palmetto*) Forest). Community type abbreviations (Bcodes) are derived from the community type names and are preceded by a physiognomic designator (F. = forest, S. = shrub, H. = herbaceous, V. = vine). For example, the Bcode H.PS is a herbaceous community. "Miscellaneous" community types (indicated by "Mx" in the classification) were initially used for species combinations that occurred infrequently or that did not clearly fall into previously established community types. Although some of these (e.g., Miscellaneous Upland Shrubland, S.MxUS) have emerged as distinct community types, these designations initially provided holding categories for mapping data until final classification decisions could be made.

Table 9-3. Excerpted portions of the prior (initial) Kissimmee River Restoration Evaluation Program vegetation classification showing species lists associated with each code. Codes shown are not sequential because rows are sorted by the first and second species listed in order to clarify relationships among codes. Species codes are referenced in Appendix 9-4A.

Category	Code	Species1	Species2	Species3	Species4	Species5	Species6	Species7
	C11	SC05	HU01	MA01	AP01			
C. Floating Aquatics, Submergents, Emergents	C12	SC05	HU01	MA01	AP01	PL99		
	C13	SC05	HU01	MU01	XFERNS			
	C5	SC05	HU01	CD01				
	J66	PH01	CA05	CP99	CX99	LF01	PR01	
	J26	PH01	CP99	LF01	RN99	PR01		
	J70	PH01	RN99	CP99	EL99			
	J73	PH01	RN99	CP99	PG05			
	J32	PH01	SS01					
	J1	PH01						
	J10	PN01	CP99	SI02				
	J14	PN01	CP99	SI02	AV01			
J. Grasses and Sedges	J16	PN01	CP99	RN99				
	J55	PN01	CP99	ST99	RN99	RN99		
	J60	PN01	CP99	LF01	RN99			
	J7	PN01	CP99					
	J72	PN01	CP99	AF01	RN99			
	J9	PN01	CP99	AF01				
	J17	PN01	JE01					
	J18	PN01	JE01	PH01				
	J19	PN01	JE01	SI02	PH01			
	J33	PN01	JE01	SB01				
	L100	AR01	MC01	DV05	QV01			
	L114	AR01	MC01	SC01	ST01	LP01	SC15	
	L115	AR01	MC01	SC01	ST01			
	L120	AR01	MC01	LP01				
	L125	AR01	MC01	SC01				
	L72	AR01	MC01	DV05				
	L83	AR01	MC01	SP01				
	L99	AR01	MC01	PP04	ST01	SC15	XVINES	SC01
	L121	AR01	MV01	SC01				
	L105	MC01	AR01					
	L111	MC01	AR01	LP01	SC01	SC15		
L. Woody Shrub and Trees	L129	MC01	AR01	TD01				
	L130	MC01	AR01	PP04	XVINES			
	L112	MC01	BH01	CO01	SC01	LP01		
	L34	MC01	BH01					
	L68	MC01	DV05					
	L76	MC01	DV05	SC01	LP01	AR01		
	L104	MC01	LP01	BH01				
	L35	MC01	LP01					
	L75	MC01	LP01	DV05	ST01			
	L78	MC01	PG01					
	L90	MC01	PG01	SC01				
	L93	MC01	PG01	ST01	LP01	XVINES		
	L1	MC01						

Quantitative statements of cover criteria are used as a means to facilitate repeatability in classification decisions among different data collectors working at different stages of the restoration project. Cover estimates are used in the Key in three ways. First, distinctions between physiognomic categories are based on 30% overstory cover. Second, most community types are defined by dominant species (defined here as the species with the highest cover, either overall or with respect to its physiognomic group if the group is dominant). Finally, some community types are defined by specified cover values or ranges of cover values.

Table 9-4. Key to the *a priori* classification of vegetation plot data. Based on the *Key to Bcode Groups and Community Types* (Appendix 9-1A).

bcode	CODE	Community Type	Bcode Group	N
H.AF	1	*Axonopus fissifolius* herbaceous vegetation	Upland Herbaceous Bcode Group	9
H.LF	2	*Luziola fluitans* herbaceous vegetation	Wet Prairie Bcode Group	1
H.LH	3	*Leersia hexandra* herbaceous vegetation	Wet Prairie Bcode Group	1
S.MC	4	*Myrica cerifera* shrubland	Upland Shrub Bcode Group	11
H.CD	5	*Cynodon dactylon* herbaceous vegetation	Upland Herbaceous Bcode Group	1
H.MxFN	6	Miscellaneous fern-dominated herbaceous vegetation	Miscellaneous Wetland Vegetation Bcode Group	1
S.MxUS	7	Miscellaneous upland shrubland	Upland Shrub Bcode Group	9
H.MxWP	9	Miscellaneous transitional herbaceous wetland vegetation	Wet Prairie Bcode Group	1
H.PH	10	*Panicum hemitomon* herbaceous vegetation	Wet Prairie Bcode Group	2
H.PN	11	*Paspalum notatum* herbaceous vegetation	Upland Herbaceous Bcode Group	30
H.PN (sod)	13	*Paspalum notatum* herbaceous vegetation (former sod farm site)	Upland Herbaceous Bcode Group	6
H.PS-PH	14	*Pontederia cordata-Sagittaria lancifolia* herbaceous vegetation	Broadleaf Marsh Bcode Group	5

The *Key to Bcode Groups and Community Types* was developed from photointerpreted cover data. Because aerial photography provides an overhead view of vegetation, understory and other covered vegetation are obscured from the photointerpreter's view, resulting in understory and secondary canopy species cover values that would be biased low compared to their actual absolute cover. This bias would affect classification decisions based on cover values of understory species in communities with an overstory of shrubs or trees. Such cases are rare in the Key, and occur only in some community types that include *Cephalanthus occidentalis* as a dominant. Field-collected species cover data can be used with the *Key* if:

(a) cover estimates disregard overlap among species (i.e., if only the amount of cover that is exposed to the sky is recorded), or

(b) absolute cover is recorded for the dominant species or dominant overstory (or vine) species.

Appendix 9-2A provides descriptions of categories of the previous classifications and discussions of linkage issues with the new classification. Appendix 9-3A is a glossary of special terms used in this chapter. Appendix 9-4A is a list of species used in the classification and associated codes. Appendix 9-5A is a table of linkage with the Milleson et al. (1980) and Pierce et al. (1982) classifications.

Assessment Results

The cluster analysis dendrogram is shown in Figure 9-1. Sample units were coded as pppp-xx where p is the plot code (minus the "VP" used in the data collector's original designation) and x is a numeric code for the assigned community type as given in Table 9-4. Important clusters are labeled A–E. The clustering algorithm used species data exclusively (not the vegetation categories assigned to the site names) for analysis.

The algorithm placed all H.AF sites (*Axonopus fissifolius* Herbaceous Vegetation, code 1) in a single cluster (Figure 9-1, cluster A), indicating that the procedure had little difficultly distinguishing H.AF from other community types. This also was the case for H.PN (*Paspalum notatum* Herbaceous Vegetation, code 11), although C035, an H.AF community, also was placed in this cluster. This site had high cover of *Paspalum notatum* (cover class 3) although the site was dominated by *Axonopus fissifolius* (cover class 4). Plot C180, which was classified as H.CD (*Cynodon dactylon* Herbaceous Vegetation, code 5) also clustered with the *P. notatum* communities. Disregarding dominance, species composition of C180 is similar to the *Paspalum notatum*-dominated pastures. All of these dominant species (*A. fissifolius*, *P. notatum*, and *C. dactylon*) are introduced pasture grasses.

One plot (C396) classified as H.MxFN (Miscellaneous Fern-Dominated communities, code 6) was placed by the clustering algorithm with H.MC (*Myrica cerifera* shrubland, code 4) sites because of similar species composition (Figure 9-1, cluster D). The fern *Osmunda regalis* is a common understory component of *Myrica cerifera* stands, often with high cover. Although species composition of this plot was similar to the *Myrica* stands, shrub cover was insufficient to classify them as S.MC (less than 30% shrub cover, see Appendix 9-2A). The cluster analysis placed all S.MC plots together in a single cluster (cluster D), and all but one S.MxUS (Miscellaneous Upland Shrubland) plot in a separate cluster (cluster C). The S.MxUS plot (C163) was placed in a cluster adjacent to the other S.MxUS plots (cluster D) otherwise composed of S.MC plots and the fern-dominated plot. Near the boundary between these two dendrogram clusters (clusters C and D), *Myrica cerifera* is present in plots in both clusters; keying of C163 as S.MxUS is due to higher cover of *Baccharis halimifolia* than *Myrica cerifera* (Appendix 9-2A). *Myrica cerifera* occurs in a variety of habitats: uplands, floating mats, and wetland-upland transition areas.

The remaining cluster, labeled B in Figure 9-1, is composed of seven H.PS-PH plots (*Pontederia cordata-Sagittaria lancifolia-Panicum hemitomon* Herbaceous Vegetation, code 14); one H.PH plot (*Panicum hemitomon* Herbaceous Vegetation, code 10); one MxWP (Miscellaneous Transitional Wetland Herbaceous Vegetation, code 9); one H.LH (*Leersia hexandra* Herbaceous Vegetation, code 3 - *L. hexandra* is a native obligate wetland grass); and one H.LF (*Luziola fluitans* Herbaceous Vegetation, code 2, another native obligate wetland grass). In this data set there is overlap in species composition among these five community types. However, these community types are all classified as Wet Prairie types (group WP) with the exception of H.PS-PH, which is a gradient community type transitional between Wet Prairie and Broadleaf Marsh (Appendices 9-1A and 9-5A).

The five community types with the most pronounced overlap in the cluster analysis were closely-related Wet Prairie and transitional community types. Poor discrimination among these types is likely an artifact of small sample sizes in this data set for some types (Table 9-4); however, the relative positions of these communities clearly reflects a gradient in habitat from long-hydroperiod wetland sites (marshes) to upland sites (pastures).

Table 9-5. Hiercarchical levels of the Kissimmee River Restoration Evaluation Vegetation Classification System.

Hierarchical level	Function	Example	Example abbreviation	Mapping
Physiognomy	Defines groups of communities based on dominant growth-forms	Herbaceous communities	H	All maps
Status	Defines generalized habitat requirements of groups of plant communities	Wetland habitats	(N/A)	All maps
Bcode Group (Group)	Defines groups of ecologically similar Community Types	Broadleaf Marsh communities	BL M	All maps
Community Type (Bcode)	Defines groups of plant communities with particular species composition. Abbreviation includes physiognomic prefix (H. in example to right)	*Pontederia cordata* and/or *Sagittaria lancifolia* herbaceous communities	H.PS	1996 Pool C baseline map only

Mapping and linkage of pasture types with previous classifications is taking place primarily at the higher classification level of Bcode Group (Carnal and Bousquin 2005), so overlap between types is probably not a problem. A fern-dominated site that contained shrubs had insufficient shrub cover to be keyed as a shrub type under the classification. However, because of its similarity in species composition to *Myrica cerifera*-dominated plots, it tended to group with these communities.

The results reveal overall good discrimination by a quantitative method among sample units for most community types included in this analysis. Further, the results suggest that gradients among the habitats supporting these plant communities are accurately reflected in the classification. Our conclusion is that appropriate levels of classification are being used, although in some cases (pasture types), species composition among community types (defined in the key by dominance of a single species) is very similar. In these cases, mapping and area calculations might be more accurate at the higher (lumped) classification level of Bcode Group.

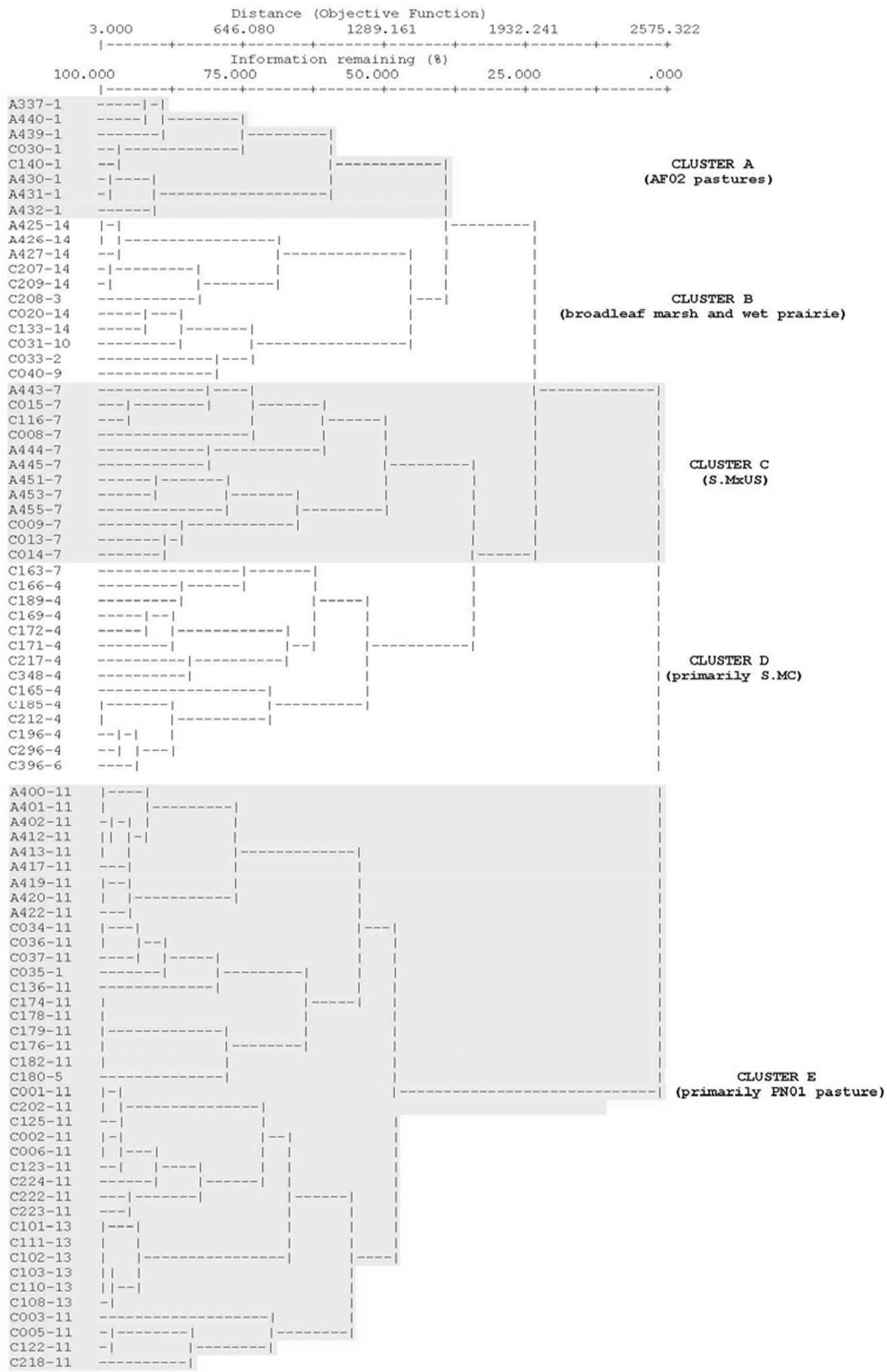

Figure 9-1. Dendrogram for the cluster analysis of 84 floodplain vegetation plots, classified from the *Key to Bcode Groups and Community types* (Appendix 9-1A). The first column gives plot numbers followed by a hyphen and the numeric code (Table 9-4) of the assigned community type. Shaded areas separate natural clusters; however the horizontal extent of shading is a graphical device and has no quantitative meaning.

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CHAPTER 10

AREAL COVERAGE OF FLOODPLAIN PLANT COMMUNITIES IN POOL C OF THE CHANNELIZED KISSIMMEE RIVER

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ABSTRACT: Three aerial photography-based vegetation maps were compared to describe change in the distribution and areal coverage of major plant communities on the Kissimmee River floodplain before and after channelization of the river. The maps describe vegetation prior to channelization, three years after channelization, and 17 years after channelization, which was immediately prior to restoration Phase I of the Kissimmee River Restoration Project. Prior to channelization, the floodplain was dominated by wetland plant communities, primarily Broadleaf Marsh, Wet Prairie, and Wetland Shrub communities. Construction of canal C-38, completed in 1971, and diversion of channel and overbank flow to the canal, resulted in loss of seasonal inundation of the floodplain and precipitated dramatic reductions in the areal extent of wetland vegetation. Phased restoration of approximately one-third of the Kissimmee River, starting with completion of Phase I restoration in 2001, will backfill sections of C-38 and modify water regulation from the river's headwaters. Restoration is expected to reestablish seasonal inundation of the floodplain and the approximate areal extent of wetland plant communities that occurred prior to channelization.

INTRODUCTION

The pre-channelized Kissimmee River floodplain was dominated by wetland vegetation (over 80% of floodplain vegetation) of three main types: (1) Broadleaf Marshes dominated by *Sagittaria lancifolia* (arrowhead) and *Pontederia cordata* (pickerelweed); (2) Wet Prairie communities dominated by *Panicum hemitomon* (maidencane) or *Rynchospora inundata* (beakrush), or composed of mixtures of wetland grasses, sedges, and forbs; and (3) Wetland Shrub communities dominated by *Salix caroliniana* (willow) or *Cephalanthus occidentalis* (buttonbush). These communities occurred in a mosaic on the floodplain, with their distribution determined by site elevation, water depths, and length of inundation (Bousquin 2005, Toth et al. 1998, Toth et al. 1995). Wetland Shrub and Broadleaf Marsh communities dominated lower elevations that were exposed to prolonged, deep hydroperiods. A number of types of Wet Prairie communities occurred at elevations with shorter and shallower hydroperiods, typically at higher elevations along the floodplain periphery.

Excavation of the C-38 canal cut off the river channel meanders, diverting virtually all flow formerly carried by the river channel and floodplain to the canal, and deposited large amounts of dredged spoil material on the floodplain. Implementation in the river's headwaters of a water regulation schedule designed primarily for flood protection prevented seasonal inundation of the floodplain. These changes substantially affected many physical and biological components of the floodplain ecosystem. The overall impact of these changes on the extent and distribution of vegetation, and prediction of the future effects of restoration of floodplain inundation, are the topics of this chapter. Restoration is expected to reestablish the approximate spatial distribution and timing of hydroperiods and result in patterns similar to pre-channelization vegetation.

The Kissimmee River Restoration Evaluation Program (KRREP) includes both landscape-scale and ground-based studies of plant communities to monitor and evaluate progress and success of the restoration. Vegetation mapping by photointerpretation of aerial photography is an integral component of the evaluation of ecological responses to restoration.

Objectives

The objectives of this study were to:

(1) Provide plant community data for Pool C as a baseline for evaluation of future change.

(2) Describe the impacts of channelization on plant communities by comparing reference (pre-channelization) and post-channelization coverages of plant communities.

(3) Predict the effects of restored hydrology on floodplain vegetation using pre-channelization data.

METHODS

Available Vegetation Data

This chapter will make use of three vegetation maps of the Kissimmee River and floodplain (Table 10-1). Two of these maps cover the entire length of the river and floodplain from the mouth of the river at Lake Kissimmee to its outlet at Lake Okeechobee. These two maps were developed during the early feasibility and planning stages of the restoration project and have been digitized by KRREP staff to provide GIS coverage of the entire restoration project area and control areas in Pools A–D. The first of these maps was based on pre-channelization (1952–1954) aerial photography (Pierce et al. 1982) and is referred to as the reference pre-channelization or 1954 vegetation map; the other is based on early-post channelization (1973–1974) photography (Milleson et al. 1980) and is referred to as the early post-channelization or 1974 vegetation map. A third map based on 1996 aerial photography of Pool C was produced by KRREP staff to establish a baseline for comparison with future vegetation mapping. This most recent map is called the Pool C baseline or 1996 vegetation map.

Together, the three maps provide a timeline of vegetation change in Pool C, showing river and floodplain vegetation 17 years prior to channelization, three years after channelization, and 25 years after channelization, just prior to the 1999 restoration activities that restored flow and inundation to much of Pool C.

Classification of Vegetation

Details of the vegetation classification used in this project can be found in Bousquin and Carnal (2005). The classification hierarchically defines plant communities at the finest level as community types (abbreviated as Bcodes), based on dominant species in plant communities. Groups of ecologically similar community types are called Bcode Groups or Groups. The highest levels of the classification are physiognomic and general habitat (status) hierarchical levels that separate groups of Bcode Groups (Table 9-5 in Bousquin and Carnal 2005).

A vegetation polygon is an area of homogeneous vegetation that is outlined on a vegetation map and is assigned a vegetation category (e.g., the name of a community type) to describe the type of vegetation the polygon includes. Polygon classification of the baseline 1996 Pool C vegetation map was performed at the detailed community type level by assigning a community type and Bcode Group to each vegetation polygon. Use of the community type level was possible for Pool C because the photointerpretation data of the 1996 photography included cover estimates for up to eight species in most polygons (Bousquin and

Carnal 2005). The community type level of classification detail was not possible for the 1954 pre- and 1974 early post-channelization vegetation maps because species data were not collected for these projects. Pool C is described at the community type level in the 1996 subsection of the Results section, but comparison with the previous maps was not possible at that level. For comparisons, the original classification categories used in the pre-channelization and early post-channelization maps were converted to the Bcode Group level of the KRREP Vegetation Classification System. Decisions involved in conversion of the previous maps to the KRREP system, and a crosswalk among the classifications are detailed in Bousquin and Carnal (2005).

For these reasons, data are presented in several ways in this chapter:

(1) Areal estimates of change resulting from channelization and predictions of future change expected to result from restoration are presented for the entire restoration area or are subdivided by restoration construction phase (see Bousquin et al. 2005, Chapter 1 of this volume). These comparisons use the 1952 and 1974 data sets only because of the spatial limitations of 1996 mapping.

(2) Comparisons of the 1996 map with previous maps are presented at the Bcode Group level for Pool C only. These comparisons are constrained to Pool C by the spatial extent of 1996 mapping.

(3) Community type (Bcode)-level areal estimates for Pool C are based on the 1996 map only. These results, however, do provide insights into the likely composition of grouped community data in the other maps.

Map Boundaries

The 1996 Pool C baseline map, and the pre-channelization and early post-channelization maps of Pools A–D were digitally overlaid to determine a common boundary for all three coverages, and to establish a fixed boundary for future evaluation program mapping. The three maps were clipped to represent the same spatial area of 15,461 ha in Pools A–D (covered only in the pre-1996 maps) and 4582 ha in Pool C (covered in all three maps) (means of map areas were used because of small discrepancies in polygon boundary matching; standard errors were ± 0.2 ha for Pools A–D, ± 1.3 ha for Pool C). This common area represents the revised study area boundary for baseline and future areal estimates and comparisons. Baseline mapping of the remaining pools will be based on aerial photography taken in 2003 overflights of the river and floodplain. All three maps were also overlaid with the boundaries of areas that will be affected by the four successive phases of restoration construction (Phases I–IV), slated for completion in 2010.

Reference Pre-channelization Mapping Methods (1952–1954 Vegetation Map)

Reference conditions for pre-channelization floodplain vegetation were available in the form of vegetation maps based on 1950s aerial photography (Pierce et al. 1982). Pierce et al. (1982) mapped 33 plant communities or land uses from two sets of 1952 and 1954 1:8000 black and white aerial photography. Their classification was based on that of Milleson et al. (1980) (see below), but several vegetation categories were added or combined with other categories. Because floodplain vegetation had changed considerably by the time of Pierce et al.'s (1982) work, ground-truthing was not possible. The manually-drawn polygon delineations in Pools A–D were digitized by Kissimmee Division staff and reduced to a scale of 1:24000. Species data are not available for this map; generalized species composition of mapped categories are available as descriptions in the text of Pierce et al. (1982) and are summarized in Bousquin and Carnal (2005).

Early Post-channelization Mapping Methods (1973–1974 Vegetation Map)

Earlier post-channelization data for Pools A–D were obtained from vegetation maps produced from photointerpretation of 1974 (1:4800) and 1973 (1:24000) aerial photography by Milleson et al. (1980) (Figure 10-1, Table 10-1). Milleson's classification was developed by delineating plant communities discernible on the photographs and verifying their signatures in the field during 1978. Ground surveys were conducted using low altitude inspection from a helicopter and supplemented by numerous observations on the ground in selected areas (Milleson et al. 1980). Milleson's classification category descriptions were based on predominant and associated species which comprised discrete communities. Vegetation delineations were digitized into Computervision Corporation's Automated Mapping System using a highly accurate digitizer. The resulting data were used to produce maps and calculate area of each community category (Milleson et al. 1980). Like the pre-channelization map, species data are not available except as generalized text descriptions of vegetation categories.

Baseline Mapping Methods (1996 Vegetation Map)

Aerial Photography

The baseline (channelized period) vegetation map was based on color infrared (CIR) aerial photography of Pools A–D of the Kissimmee River and floodplain, acquired in June 1996 at a scale of 1:6000.

The extent of the overflight was based on the U. S. Army Corp of Engineers (USACE 1991) project area boundary for the 1994 overflight of Pools A, B, C, and D. Color infrared film was chosen because of its sensitivity to light energy (Sabins 1987) and its ability to reduce the effects of atmospheric haze (Greer et al. 1990). Color infrared film is especially useful for differentiating wetland vegetation (Owens and Laustrap 1990, Owens 1990) because it is sensitive to the multiple reflectance and scattering of light in the spongy mesophyll structure of plants (Fouche 1993), which is greater when water content is high. In this kind of image, greater water storage within plant tissue causes wetland vegetation to appear darker pink or red relative to upland terrestrial species. Because the amount of infrared light reflected from vegetation is related to contained moisture and the structure of leaves (Greer et al. 1990), species can be distinguished on CIR images with a high degree of confidence.

Vegetation Mapping

Study Area Boundary. Ground elevation and historic hydrology were the determining factors for the original study area boundary for baseline vegetation mapping. The mapping boundary was later adjusted to make the final Pool C baseline map compatible with the pre-channelization and early post-channelization maps (see above).

Scale and Minimum Mapping Unit. The large scale (1:6000) of photograph acquisition was chosen to ensure that species could be differentiated, and that subtle features such as narrow bands of river channel vegetation could be delineated. A small minimum mapping (MMU) unit of 100 m^2 accommodated inclusion of these and other small patches of vegetation (Digital Map Appendix 13A and 14A). The MMU defines the smallest allowable polygon delineations, although it was not intended as a rule. Within a forest community, for example, it was not necessary to delineate each small opening in the canopy to capture understory herbaceous vegetation coverage. This would not change the overall physiognomy of the community and would result in an overestimate of the coverage of the herbaceous community.

Photo Preparation and Polygon Delineation. A vegetation polygon is a drawn boundary on an aerial photograph that delineates a contiguous area of homogenous vegetation. Clear Mylar overlays were attached with drafting tape to each positive transparency of the aerial photography to provide a medium for polygon delineations. The Mylar was then registered to the photography by marking the fiducials or cross hairs on each side of the transparency with drafting pens. Each transparency was labeled in the upper right corner with date, photo number, and pool for reference. Work areas (area of photointerpretation) were determined by the extent of overlap between consecutive photos within a flight line and by the amount of sidelap of photos in adjacent flight lines. This established the work area in the center of the photography where distortion is at a minimum. Vegetation polygons, work areas, boundaries, and labels were delineated on the Mylar overlays.

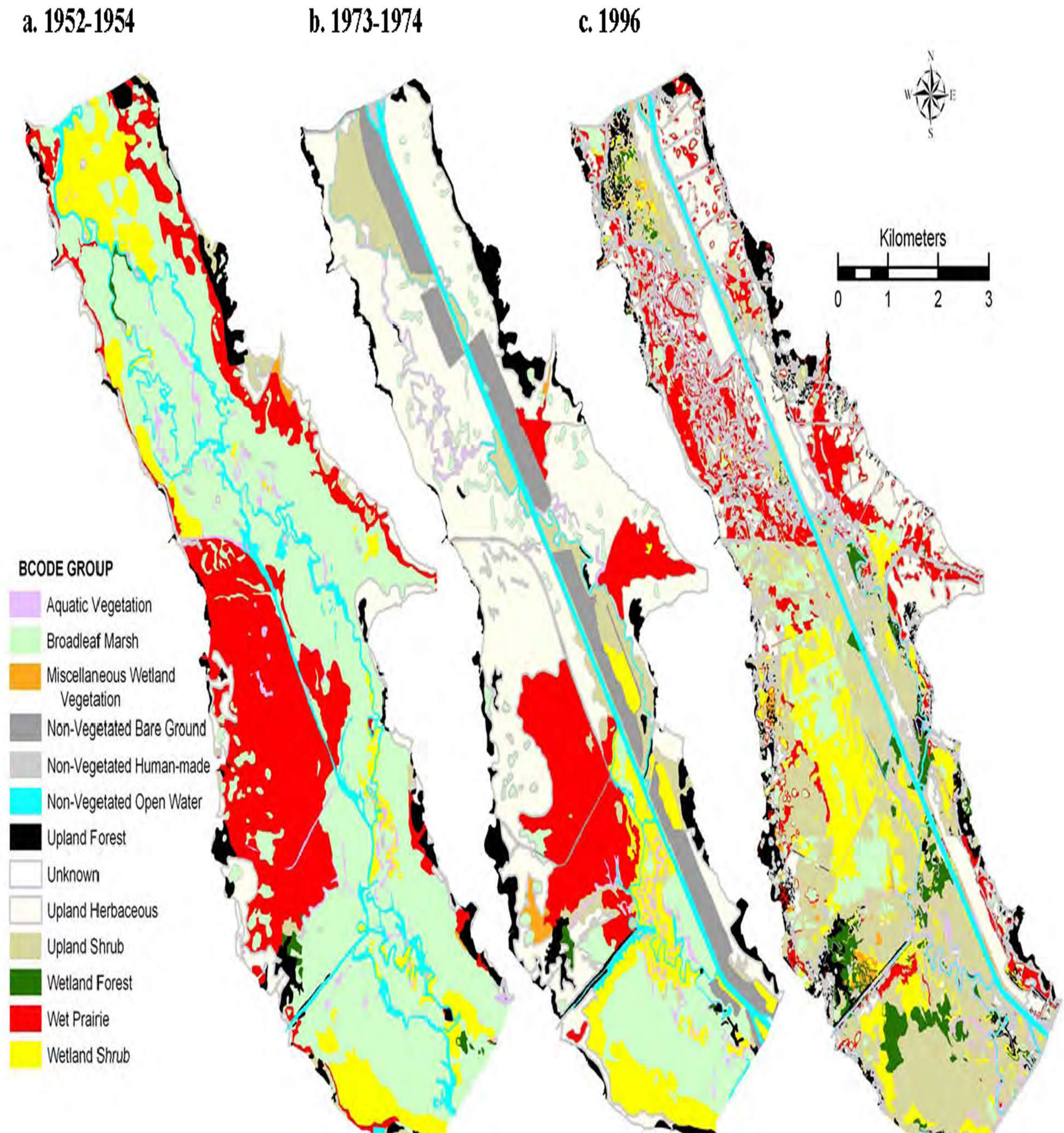

Figure 10-1. Vegetation maps of Pool C: (a) 1952–1954 (pre-channelization, reference period) (data from Pierce et al. 1982); (b) 1973–1974 (early post-channelization) (data from Milleson et al. 1980); and (c) 1996 (post-channelization, baseline period).

Table 10-1. Vegetation maps used in restoration evaluation of vegetation change, 1952–1996.

Date of photography	Map Source	Digitized coverage	Period	Vegetation classification level	Application in this chapter
1952-54	Pierce et al. 1982	Pools A-D	Pre-channelization reference	Bcode Group	Reference data for predictions of restored areal abundance of plant communities
1973-74	Milleson et al. 1980	Pools A-D	Early channelized	Bcode Group	Early responses to channelization
1996	Carnal and Bousquin 2005	Pool C	Channelized baseline	Bcode Group	Baseline data for comparison with pre-channelization data in Pool C and future restored-condition data
				Community Type (Bcode)	Detailed descriptions of Pool C baseline condition

Vegetation Automation. All delineations were originally automated by transforming and projecting each individual work area to a base map. This georeferencing technique was abandoned because distortion in the photography resulted in gaps between work area delineations. Subsequently, a method for simultaneous two-dimensional projective coordinate transformation was applied (Dewitt 1999). This method transforms multiple overlapping images to a ground control system. Application of the methodology produced an accuracy of 3 to 5 meters where there was sufficient control and low topographic variability. Using this methodology, work area delineations were aggregated into a single, seamless GIS coverage and added to the KRREP spatial database.

Photointerpretation and Verification. Photointerpretation was performed using a Baush and Lomb 240-zoom stereoscope mounted on a Richards light table. During photointerpretation, vegetation signatures were identified, delineated, and labeled using three decision methods. First, field reconnaissance of all remnant river channels and portions of the floodplain was conducted before and continued after the photography was obtained. Notes about species present in 1996 were recorded on Mylar sleeves over prints of 1994 black and white aerial photography. Similar reconnaissance was conducted using the 1996 baseline photography (e.g. Map Appendix 11A and 12A) and included areas of the floodplain that were previously unvisited. Second, field reconnaissance was conducted during photointerpretation and classification to calibrate polygons with unknown signatures. Locations of field observations were captured with a Global Positioning System (GPS) unit, species information was recorded, and ground photos were obtained for archive and use in making classification decisions during interpretation. Approximately two thirds of all polygons were interpreted with ground verification by one of these two methods. Third, polygon classification was interpreted without ground truth data based on knowledge of flora in the area and skill in recognizing associated signatures developed through ground truth methods.

Landscape Zone Modifier. A landscape zone attribute (Appendix 10-1A) was attached to each polygon to describe its context within the study area boundary. A landscape zone is a geographic feature of the Kissimmee River basin, such as a spoil mound, C-38, floodplain, or the upland ecotone. The landscape zone modifier was needed because some vegetation required additional qualification to accurately describe where the community existed in the river/floodplain system. The landscape zone attribute can provide data independent of vegetation cover, for example an area of dredged spoil material on the floodplain or an area of open water on the floodplain vs. the river channel.

RESULTS

Upland Communities

Vegetation Change in Pools A–D, 1954–1974

Wetland vegetation dominated the floodplain prior to channelization, occurring on over 80% of the floodplain's total area of 15,461 ha in the 1954 reference vegetation map. Pre-channelization wetlands were dominated by herbaceous marshes, including Broadleaf Marsh (BLM) which occurred on 46% of the floodplain, and Wet Prairies (WP) on 21%. Wetland Shrub communities (WS) covered 13% of the floodplain in the 1954 map, and Wetland Forests (WF) about 0.5% (Table 10-2, Figures 10-1 and 10-2).

Following channelization, the 1974 early post-channelization aerial photography indicated that wetlands had declined to about 29% of the floodplain, including Broadleaf Marsh (BLM) on 7% of the floodplain and Wet Prairie (WP) on 13%. Coverage of Wetland Shrub communities (WS) declined to about 8% of the floodplain in the 1974 map, and cover of Wetland Forest (WF) remained approximately constant at <1% (Table 10-2, Figures 10-1 and 10-2).

Upland vegetation occurred on only 8% (1,285 ha) of the floodplain prior to channelization, but increased to 52% in the 1974 early post-channelization photography. Upland Herbaceous (UP), primarily pastures, which occurred on only 3% of the floodplain in 1954, increased dramatically to 37% by 1974. Upland Shrub (US) communities increased slightly from 2% to 9%, and Upland Forest (UF) from 3% to 5%. Occurring in the remaining small areas of both vegetation maps were small areas of Non-Vegetated Open Water (NVOW), Bare Ground (NVBG), and Problematic Signatures (XUNK) (Table 10-2, Figures 10-1 and 10-2).

Vegetation Change in Pool C, 1954–1996

Pool C is treated at the Bcode Group level in this section independently of Pools A, B, and D. This enables comparison between the 1996 baseline mapping effort, which was conducted only for Pool C, and previous map data. Pool C areal coverage of Bcode Groups was very similar to coverage in the whole system. In the 1954 map, wetlands covered approximately 83% of Pool C's total area of 4582 ha, and were dominated by Broadleaf Marshes (BLM) (50% of Pool C) and Wet Prairie (WP) communities (23%) (Table 10-2, Figures 10-1 and 10-2).

Table 10-2. Percent total cover of Bcode Groups within the Pool C boundary, 1952–1996, and the entire restoration project and control area, Pools A–D, 1952–1974. Prior to channelization, wetlands comprised over 80% of the floodplain area in both Pool C and in Pools A–D combined. Following channelization in 1974, upland vegetation covered over 50% of Pool C and Pools A–D combined, and by 1996 covered over 60% of Pool C.

			Area in Pool C (ha)			Percent of Pool C			Area in Pools A-D (ha)		Percent of pools A-D	
STATUS	Bcode Group	Code	1952	1974	1996	1952	1974	1996	1952	1974	1952	1974
Wetland	Broadleaf Marsh	BLM	2269	479	342	50	10	7	7061	1084	46	7
	Miscellaneous Wetland Vegetation	MW	8	23	41	0	1	1	136	182	1	1
	Wet Prairie	WP	1054	552	490	23	12	11	3204	1939	21	13
	Wetland Forest	WF	19	9	164	0	0	4	75	61	0	0
	Wetland Shrub	WS	431	335	413	9	7	9	1976	1235	13	8
Upland	Upland Forest	UF	185	314	314	4	7	7	494	830	3	5
	Upland Herbaceous	UP	205	1832	1219	4	40	27	512	5752	3	37
	Upland Shrub	US	64	329	1274	1	7	28	279	1450	2	9
Wetland/Upland	Vine	VN	0	0	3	0	0	0	0	0	0	0
Aquatic	Aquatic Vegetation	AQ	76	69	151	2	1	3	349	205	2	1
Unknown	Unknown	UN	23	0	6	1	0	0	208	1	1	0
Non-vegetated	Non-Vegetated: Bare Ground	NVBG	0	430	1	0	9	0	0	1683	0	11
	Non-Vegetated: Human Influenced	NVH	2	6	3	0	0	0	31	46	0	0
	Non-Vegetated: Open Water	NVOW	247	201	162	5	4	4	1134	993	7	6
	Totals		4583	4579	4583	100	100	100	15461	15461	100	100

After channelization, wetland vegetation coverage in the 1974 and 1996 Pool C maps was 31% and 32%, respectively. As in Pools A–D, wetland vegetation was composed primarily of Broadleaf Marsh (BLM) (10% in 1974 and 7% in 1996), Wet Prairies (WP) (12% in 1974 and 11% in 1996), and Wetland Shrub (WS) communities (7% in 1974 and 9% in 1996), with a small area of Miscellaneous Wetland (MW) (2% in 1974 and 1% in 1996) (Table 10-2, Figures 10-1 and 10-2).

Uplands comprised only 10% of Pool C prior to channelization and had increased to 54% in 1974, and to 61% in 1996. Upland Herbaceous (UP) communities occurred on 40% of Pool C's area in the 1974 map and 27% in 1996; Upland Shrub (US) communities covered 7% in 1974 and 28% in 1996 (Table 10-2, Figures 10-1 and 10-2).

Small percentages of Pool C were covered by Upland Forest (UF) (~ 7% in both 1974 and 1996), Aquatic Vegetation (AQ) (1% in 1974 and 3% in 1996), and open water (NVOW) (5% and 4%); and less than 2% of bare ground (NVBG) and Problematic Signatures (XUNK) (Table 10-2, Figures 10-1 and 10-2).

Community Type Composition of Pool C

Pool C vegetation is described in this section at the finest level of the KRREP classification, community type or Bcode, which defines communities by dominant species (Bousquin and Carnal 2005). This level of classification is not available for the previous vegetation maps. It is presented here to provide insight into the likely community type composition of the Group levels of classification applied to the other maps.

Upland Communities

Upland Forests. The Upland Forest Group (UF) covered 7% of the total mapped area in Pool C. Upland Forests occurred primarily at elevations higher than Wet Prairie, on natural riparian berms along river channels, and in abandoned river channels. Some upland tree and shrub species colonized areas of spoil and pasture after channelization. The *Quercus virginiana* (live oak) Forest (F.QS) community type comprised 94% of this Group, occurring typically in dense hammocks along the periphery of the floodplain (Figure 10-1), a zone that roughly approximates the study area boundary and 100-year floodline. However, *Q. virginiana* also colonized areas that were drained by channelization.

Forests dominated by *Sabal palmetto* (cabbage palm) (F.SP) covered 5% of the Upland Forest area in Pool C. *Sabal palmetto* (cabbage palm) was commonly found in association with live oak hammocks, although isolated stands of cabbage palms were frequent in pastures and on spoil areas, where they became established after channelization. A Miscellaneous forest (F.MxF) category, which was used for other combinations of upland trees, was less common (2%) (Table 10-3, Figure 10-3).

Upland Shrub Communities. The Upland Shrub Group (US) was the largest component of Pool C in 1996, (Figure 10-3, Table 10-2), occurring in almost every natural habitat on the floodplain and in portions of the former river channel. Upland Shrub communities covered approximately 28% (Table 10-2) of the total mapped area in the 1996 map, and were absent only from heavily grazed pastures and the lowest floodplain elevations. The Upland Shrub Group includes community types composed of both exotic and native Upland Shrub species, including almost pure stands of the invasive native *Myrica cerifera* (wax myrtle) (S.MC) which was the largest component of this Group in baseline Pool C, comprising 77% of the Group and 21% of the total Pool C study area (Table 10-3, Figure 10-3). Because *M. cerifera* was rare in the pre-channelized system, no comparable category was defined in the pre- or early post-channelization classifications. However, by 1996, the community type level of mapping reveals that a myrtle community had invaded a remnant wetland community in the southwest section of Pool C (Figure 10-1) on elevated "floating" substrate composed of roots and vegetative debris. Because of this above-water condition, these *M. cerifera* stands are capable of persisting in the permanently inundated lower portion of the pool. *Myrica cerifera* stands also occurred on higher elevations of former marsh areas.

The Miscellaneous Upland Shrub (S.MxUS) community type encompasses mixed communities of *Schinus terebinthifolious* (brazilian pepper), *Sambucus canadensis* (elderberry), *Psidium guajava* (guava), *Rubus cuneifolious* (sand blackberry), *Baccharis halimifolia* (groundsel bush), and *M. cerifera*. The type is often located on spoil mounds and occurs throughout the drained floodplain. S.MxUS made up 16% of the Upland Shrub Group, while communities dominated by the exotic nuisance species *Schinus terebinthifolious* (brazilian pepper) (S.ST) comprised 6% of the Upland Shrub Group (Table 10-3, Figure 10-3). *Schinus terebinthifolius* was not observed prior to channelization but has invaded a number of habitats since channelization, including the banks of C-38 and riparian zones along remnant river channels, where it has replaced *Salix caroliniana* as the dominant species.

Upland Herbaceous Communities. The second-largest component of the river-floodplain system in the 1996 Pool C data was the Upland Herbaceous Bcode Group (UP) (Table 10-2, Figure 10-2), which comprised 27% of the total mapped area in Pool C (Table 10-2). This Group includes pastures and other communities composed of exotic and native grasses, and communities of exotic weeds, which have became established in areas that were historically Broadleaf Marsh, Wet Prairie, and in some areas, Wetland Shrub. Seeding of *Paspalum notatum* and other forage grasses, consistent grazing, and the lack of water retention on the floodplain promoted development of economically valuable cattle pasture for surrounding land owners. Spoil areas also were covered primarily by Upland Herbaceous communities and comprise large areas of the channelized system.

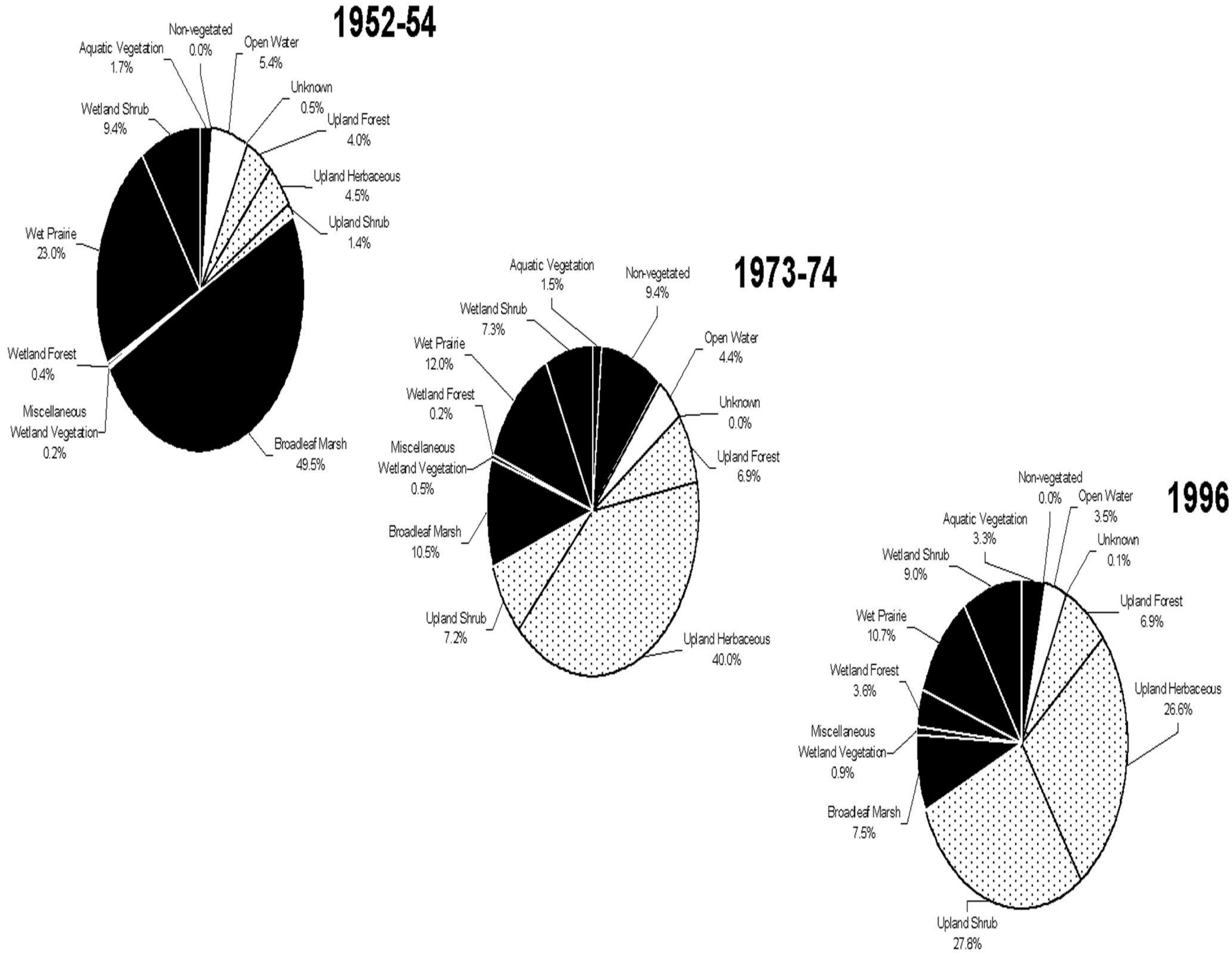

Figure 10-2. Percentages of floodplain cover of Bcode Groups in Pool C in the pre-channelization (1952–1954), early post-channelization (1973–1974), and 1996 vegetation maps.

Table 10-3. Areal extent of Community types (Bcodes) in the Kissimmee River Pool C floodplain, 1996.

Habitat	Bcode Group	Bcode Group name	Bcode	Bcode name	Area (ha)	Percent of Bcode Group	Percent of Pool C
Aquatic	AQ	Aquatic Vegetation	H.EC	Eichhornia crassipes herbaceous aquatic vegetation	2.0	1.4	0.0
			H.EC-PST	Eichhornia crassipes-Pistia stratiotes herbaceous aquatic vegetation	0.1	0.0	0.0
			H.HU	Hydrocotle umbellata herbaceous aquatic vegetation	0.3	0.2	0.0
			H.MFM	Miscellaneous herbaceous floating mat vegetation	4.4	2.9	0.1
			H.MxFA	Miscellaneous aquatic vegetation dominated by floating species	1.9	1.3	0.0
			H.MxM	Miscellaneous littoral marsh vegetation	12.5	8.3	0.3
			H.MxSV	Miscellaneous submergent aquatic vegetation	2.2	1.5	0.0
			H.NL	Nuphar lutea herbaceous aquatic vegetation	16.7	11.1	0.4
			H.PD	Polygonum densiflorum herbaceous aquatic vegetation	27.4	18.2	0.6
			H.PST	Pistia stratiotes herbaceous aquatic vegetation	0.2	0.2	0.0
			H.SCF	Scirpus cubensis herbaceous floating mat vegetation	32.7	21.7	0.7
			H.SS	Sacciolepis striata herbaceous aquatic vegetation	5.5	3.7	0.1
			S.LSF	Ludwigia spp. floating mat shrubland	29.0	19.3	0.6
			S.MCF	Myrica cerifera floating mat shrubland	6.9	4.6	0.2
			S.MxFS	Miscellaneous floating mat shrubland	8.7	5.8	0.2
Non vegetated	NVBG	Non Vegetated, Bare Ground	NVBG	No vegetation - bare ground	1.2	0.7	0.0
			NVH	No vegetation - human-made structures, roads, etc.	3.2	1.9	0.1
			NVOW	No visible vegetation - open water	161.9	97.4	3.5
Unknown	UN		XUNCL	Unknown - Unclassified	4.9	78.4	0.1
			XUNK	Unknown - Uninterpretable signature	1.3	21.6	0.0
Upland	UF	Upland Forest	F.MxF	Miscellaneous upland forest	1.7	0.5	0.0
			F.QS	Quercus virginiana (-Sabal palmetto) forest	295.7	94.2	6.5
			F.SP	Sabal palmetto forest	16.6	5.3	0.4
	UP	Upland Herbaceous	H.AF	Axonopus fissifolius herbaceous vegetation	64.4	5.3	1.4
			H.MxE	Miscellaneous exotic herbaceous vegetation	74.7	6.1	1.6
			H.MxN	Miscellaneous native herbaceous vegetation	132.1	10.8	2.9
			H.MxW	Miscellaneous invasive herbaceous vegetation	109.5	9.0	2.4
			H.PN	Paspalum notatum herbaceous vegetation	838.0	68.8	18.3
	US	Upland Shrub	S.MC	Myrica cerifera shrubland	981.4	77.0	21.4
			S.MxUS	Miscellaneous upland shrubland	203.3	16.0	4.4
			S.PG	Psidium guajava shrubland	7.1	0.6	0.2
			S.SR	Serenoa repens shrubland	8.3	0.7	0.2
			S.ST	Schinus terebinthifolius shrubland	73.7	5.8	1.6
Wetland	BLM	Broadleaf Marsh	H.PS	Pontederia cordata-Sagittaria lancifolia herbaceous vegetation	159.6	46.7	3.5
			H.PS-CO	Pontederia cordata-Sagittaria lancifolia-Cephalanthus occidentalis herbaceous vegetation	25.2	7.4	0.6
			H.PS-HG	Hibiscus grandiflorus-Pontederia cordata-Sagittaria lancifolia herbaceous vegetation	1.0	0.3	0.0
			H.PS-PH	Pontederia cordata-Sagittaria lancifolia-Panicum hemitomon herbaceous vegetation	69.6	20.4	1.5
			H.PS PH CO	Pontederia cordata Sagittaria lancifolia Panicum hemitomon Cephalanthus occidentalis herbaceous vegetation	86.1	25.2	1.9
	MW	Miscellaneous Wetland Vegetation	H.CJ	Cladium jamaicense herbaceous vegetation	0.9	2.3	0.0
			H.HG	Hibiscus grandiflorus herbaceous vegetation	15.3	37.5	0.3
			H.SB	Spartina bakeri herbaceous vegetation	13.4	33.0	0.3
			H.TY	Typha domingensis herbaceous vegetation	11.1	27.3	0.2
	WF	Wetland Forest	F.AR	Acer rubrum (-Nyssa silvatica var. biflora) forest	120.8	73.6	2.6
			F.FC	Fraxinus caroliniana forest	1.9	1.1	0.0
			F.MTF	Mixed transitional forest	22.0	13.4	0.5
			F.MV	Magnolia virginiana forest	3.1	1.9	0.1
			F.TD	Taxodium distichum forest	16.4	10.0	0.4
	WP	Wet Prairie	H.AG	Andropogon glomeratus herbaceous vegetation	14.0	2.9	0.3
			H.CS	Cyperus spp. herbaceous vegetation	4.6	0.9	0.1
			H.ES	Eleocharis spp. herbaceous vegetation	12.2	2.5	0.3
			H.IV	Iris virginica herbaceous vegetation	18.0	3.7	0.4
			H.JEd	Juncus effusus herbaceous vegetation (upland depressions)	12.0	2.5	0.3
			H.JEp	Juncus effusus herbaceous vegetation (wet prairies)	110.8	22.6	2.4
			H.LF	Luziola fluitans herbaceous vegetation	81.6	16.6	1.8
			H.MxWT	Miscellaneous transitional herbaceous wetland vegetation	142.6	29.1	3.1
			H.MxWT	Miscellaneous native wetland graminoid vegetation	39.8	8.1	0.9
			H.PH	Panicum hemitomon herbaceous vegetation	24.5	5.0	0.5
			H.PP	Polygonum punctatum herbaceous vegetation	20.8	4.2	0.5
			H.PR	Panicum repens herbaceous vegetation	0.8	0.2	0.0
			H.RN	Rhynchospora spp. herbaceous vegetation	8.7	1.8	0.2
	WS	Wetland Shrub	S.CO	Cephalanthus occidentalis shrubland	10.3	2.5	0.2
			S.CO-PS	Cephalanthus occidentalis-Pontederia cordata-Sagittaria lancifolia shrubland	28.5	6.9	0.6
			S.CO-PS-PH	Cephalanthus occidentalis-Pontederia cordata-Sagittaria lancifolia-Panicum hemitomon shrubland	156.6	37.9	3.4
			S.HF	Hypericum fasciculatum shrubland	1.0	0.2	0.0
			S.LS	Ludwigia spp. shrubland	126.9	30.7	2.8
			S.SC	Salix caroliniana shrubland	90.2	21.8	2.0
Wetland/Upland	VN	Vine	V.LM	Lygodium microphyllum-dominated communities	0.1	3.8	0.0
			V.MxV	Miscellaneous vine-dominated communities	2.9	96.2	0.1
					4,583.0		100.0

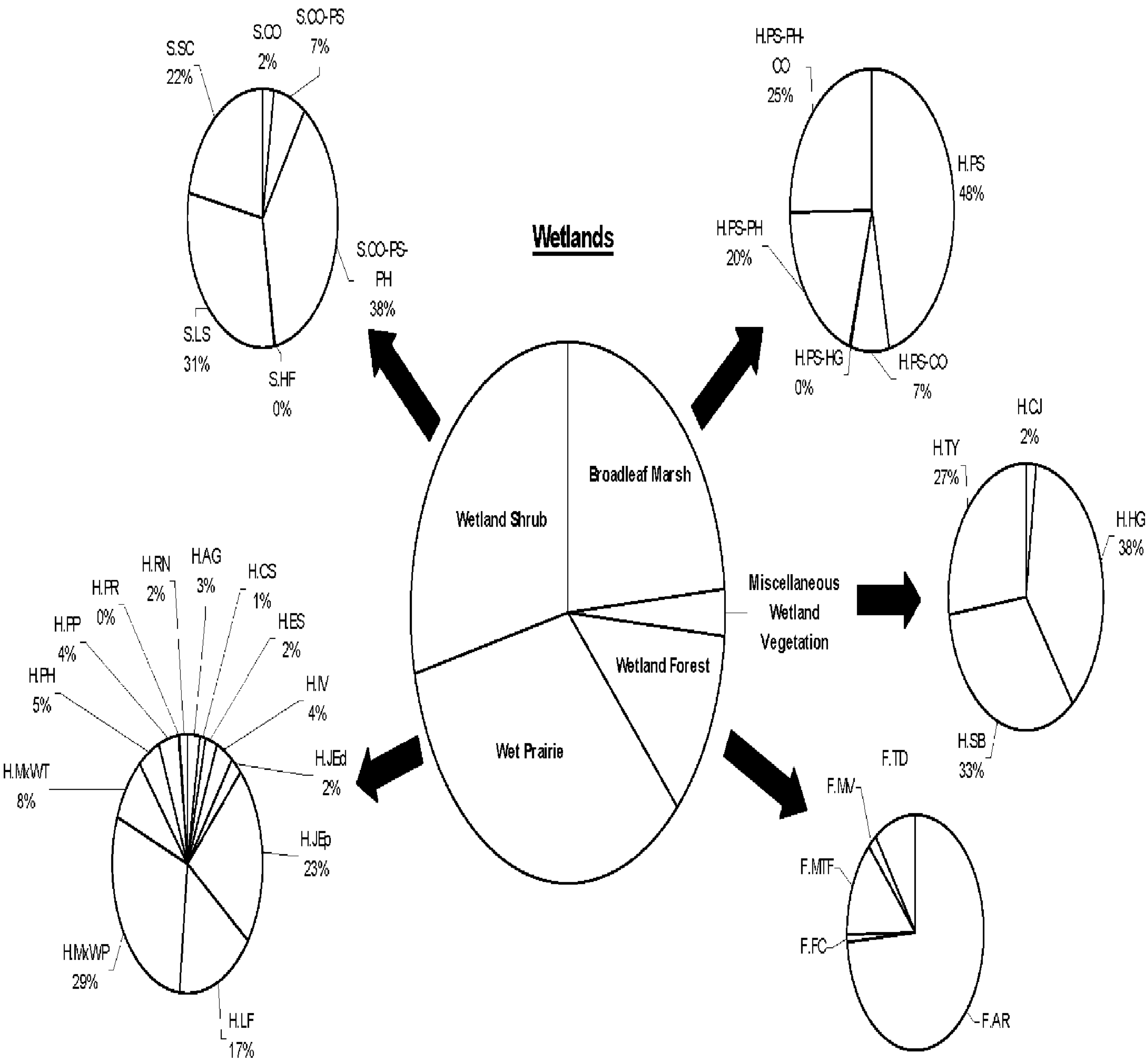

Figure 10-3. Community type percentages (outer circles) of wetland Bcode Groups (central circle) in the 1996 Pool C Vegetation Map. Community type codes are defined in Table 10-3.

The most extensive community type in the UP Group after channelization was pastures dominated by *Paspalum notatum* (bahia grass) (H.PN), which accounted for 69% of this Group and 18% of the total mapped area in Pool C (Table 10-3, Figure 10-3). Second-most common was the Mixed Native Herbacous community type (H.MxN), which covered 11% of Upland Herbaceous areas. This community type has no defined dominant taxa and may include mixtures of native upland graminoids such as *Andropogon virginicus, Panicum angustifolium*, some *Cyperus* spp., non-native and pasture grasses, and other herbaceous communities. Semi-woody annuals and perennials (e.g. *Lantana camara, Sesbania* spp., *Callicarpa Americana)* are often found scattered in these mixed native grasslands. Miscellaneous invasive communities (H.MxW), which are dominated by invasive native species, comprised 9% of the Upland Herbaceous Group in Pool C, and Miscellaneous Exotic Herbaceous communities (H.MxE) occupied 6% of the Group (Table 10-3, Figure 10-3).

Wetland Communities

Wetland Forests. As in other pools during the channelized period, in Pool C, Wetland Forest occurred mostly in the lower third of the pool (Figure 10-1), where hydroperiods were longer due to the backwater effect created by the pool's water control structure and related levees. The largest component of the Wetland Forest Group in channelized Pool C was the *Acer rubrum* (red maple) Forest, which accounted for 74% of the Wetland Forest Group (Table 10-2). *Acer rubrum* communities occurred in dense stands near river channels or mixed with other wetland tree and shrub species. *Taxodium distichum* (cypress) Forest (F.TD) comprised 10% of Wetland Forest, usually in stands in riparian zones of remnant river channels in the lower portion of Pool C (Table 10-3, Figure 10-3). *Magnolia virginiana* (sweetbay) (F.MV) and *Fraxinus caroliniana* (carolina ash) (F.FC) Forests combined, accounted for only 3% of this Group. *Magnolia virginiana* communities typically occurred as domes or "heads" in wet depressions within peripheral Upland Forest. *Fraxinus caroliniana* (carolina ash) communities occurred infrequently in dense clumps within wet depressions. The Miscellaneous Transitional Forested (F.MTF) community made up 13% of Wetland Forest in Pool C (Table 10-3, Figure 10-3). This type is typically composed of combinations of upland (e.g. *Quercus virginiana, Fraxinus caroliniana*) and wetland (e.g. *Persea* spp., *Taxodium distichum, Acer rubrum*) species, occurring in various situations but more often in wetland habitats than upland habitats. The F.MTF community was most abundant along the Istokpoga canal, which was excavated before channelization of the Kissimmee River (Figure 10-1).

Wetland Shrub Communities. The Wetland Shrub Group (WS) accounted for 9% of Pool C area in 1996 (Table 10-2, Figures 10-1 and 10-2). This group includes several community types with at least 30% cover of *Cephalanthus occidentalis* (buttonbush) (S.CO, S.PS-CO, S.PS-PH-CO, Table 10-3), which combined, made up over 47% of the Wetland Shrub Group (Table 10-3). These communities typically occurred with Broadleaf Marsh understories, in several associations differentiated by percent cover of the several dominants (Bousquin and Carnal 2005). These communities have a marsh-like appearance with a thin overstory of shrubs, occurring mainly in the south-central portion of the pool, west of C-38. *Salix caroliniana* (coastal plains willow) communities (S.SC) made up 22% of this Group (Table 10-3, Figure 10-3). *Ludwigia* spp. (primrose willow) communities (S.LS) accounted for 31% of the Wetland Shrub Group in Pool C. The majority of *Ludwigia* species in the river system is *L. peruviana* (Peruvian primrose willow) commonly found in or along abandoned channels, ditches, and at lower elevations. The species has benefited from stabilized hydrology and often occurs where willow has declined. *Ludwigia, Salix,* and *Myrica* shrub communities growing on floating mats formed by *Scirpus cubensis* (cuban bullrush) were differentiated separately and are discussed in the Aquatic Vegetation (AQ) Group, below. *Hypericum fasciculatum* (sandweed) communities (S.HF) typically occurred in the outer rings of upland marsh depressions but were not common (0.2% of the Wetland Shrub Group).

Broadleaf Marsh. The Broadleaf Marsh Bcode Group (BLM) includes five combinations of wetland forb and grass mixtures, although the dominant species of all of these types are *Pontederia cordata* (pickerelweed) and/or *Sagittaria lancifolia* (bulltongue arrowhead) (Bousquin and Carnal 2005) (Table 10-3). The *Pontederia cordata/Sagittaria lancifolia* (H.PS) community type contributed the largest area of the Broadleaf Marsh Group, at 47% of the Group (Table 10-3, Figure 10-3). The other major communities contained lesser coverage of these two species, but all include significant cover of *Panicum hemitomon* or *Cephalanthus occidentalis* (i.e. H.PS-PH, H.PS-CO, H.PS-PH-CO) (Table 10-3) and combined, accounted for over 53% of the Broadleaf Marsh Group. The Broadleaf Marsh community contains five combinations of forbs and wetland grass mixtures, although the dominant species for all of these types is *Pontederia* *cordata* (pickerelweed) and/or *Sagittaria lancifolia* (bulltongue arrowhead). The *Pontederia/Sagittaria* (H.PS) community was the largest constituent of the BLM habitat and must contain at least 50% cover of one or both species of *Pontederia* or *Sagittaria* in a polygon to be classified as such (Table 10-3, Figure 10-3). Other species commonly occurring in Broadleaf Marshes include *Cephalanthus occidentalis, Panicum hemitomon,* and *Hibiscus grandiflorus.*

Wet Prairie. The Wet Prairie Group includes communities with various combinations of graminoid and forb species. *Panicum hemitomon* (maidencane) communities (H.PH) and *Rhynchospora* spp. (beakrushes) communities (H.RN) were common Wet Prairie components in the pre-channelization system (Pierce et al. 1982), but together accounted for only 7% of Pool C Wet Prairie in 1996 (Table 10-3, Figure 10-3). *Panicum hemitomon* dominated a large region of MacArthur Impoundment in a west-central portion of Pool C prior to channelization (Figure 10-2). The ditch and levee system of this impoundment likely shortened hydroperiods and led to the dominance of maidencane in this area, which is surrounded by Broadleaf Marsh. *Polygonum punctatum* (dotted smartweed) (H.PP) and *Juncus effusus* (soft rush) communities (H.JEp and H.JEd) often occurred in wet depressions within pastures. The two *Juncus* community types accounted for 25% of Wet Prairie coverage in Pool C in 1996. The *Polygonum* community type was common in agricultural ditches and accounted for 4% of Pool C Wet Prairie. Like the *Juncus* communities, the *Iris virginica* (Virginia iris) community type (H.IV) was found around pasture depressions, but was less common and seasonal in occurrence. *Iris virginica* accounted for 4% of the Wet Prairie. It was mainly distributed in lower elevation pastures near Oak Creek (Figure 10-1). *Luziola fluitans*- (southern watergrass) dominated communities (H.LF) covered almost 17% of the Wet Prairie habitat in baseline Pool C. Stabilized water levels, pasture grass seeding, and grazing led to replacement of Wet Prairie species by other species including *Paspalum notatum*, *Axonopus* spp., and various species of weeds. In addition, some forbs with low forage value (e.g. *Pontederia cordata, Juncus effusus*) for cattle consumption (Pruitt et al. 1976) established where conditions were favorable. This was evident in pasture depressions containing remnant wetlands with mixed species of broadleaf marsh, and in wet prairies surrounded by heavily grazed pasture grasses.

Several Wet Prairie species that occurred in the channelized system (e.g., *Juncus effusus, Luziola fluitans, Phyla nodiflora, Centella asiatica, Iris virginica, Eleocharis spp., Andropogon glomeratus*) apparently did not occur as dominants in the pre-channelization wet prairies described by Pierce et al. (1982). These plants probably occurred infrequently in the pre-channelized system.

Luziola fluitans (southern watergrass) was common at lower elevations of pastures and depressions, often associated with *Polygonum punctatum* (dotted smartweed), smaller species of *Eleocharis* (spikerush), *Bacopa* spp. (hyssops), *Phyla nodiflora* (turkey tangle frogbit), *Hydrocotyle umbellata* (manyflower marshpennywort), *Centella asiatica* (spadeleaf), and occasionally with *Pontederia cordata, Sagittaria lancifolia,* and *Juncus effusus. Andropogon glomeratus* (bushy bluestem) is a grass preferring moist soils, but was found throughout the channelized system in pastures, floating mats, Upland Shrub communities, and disturbed areas. *Panicum repens* (torpedo grass) and *Leersia hexandra* (southern cutgrass), which often form dense mats in shallow water, also occurred in very small amounts on the floodplain.

Miscellaneous Wetlands. The Miscellaneous Wetland Group (MW) includes communities dominated by *Cladium jamaicense* (sawgrass) (H.CJ), *Typha domingensis* (southern cattail) (H.TY), *Spartina bakerii* (sand cordgrass) (H.SB), *Hibiscus grandiflorus* (swamp rosemallow) (H.HG), and a fern-dominated community (H.MxFN). The MW Group comprised only 0.9% of the total mapped area in Pool C (Table 10-2, Figure 10-2). *Hibiscus* communities were the largest component of this category, comprising about 38% of all MW communities (Table 10-3, Figure 10-3). *Cladium* communities made up 2%, *Typha* communities accounted for 27%, and *Spartina* communities comprised 33% of the MW Group. *Cladium jamaicense* communities occurred mostly in small patches within Broadleaf Marsh communities and was rare on the Pool C floodplain during baseline evaluation. *Typha domingensis* occurred in small areas in often dense clumps across many landscape zones (Appendix 10-1A), particularly in spoil or road ditches. *Spartina bakeri* prefers moist soils, and dominated communities found primarily on the periphery of the floodplain between wetland and upland habitats, where it often occurred in sparse linear expanses. No fern-dominated communities were mapped during baseline evaluation, although ferns are often abundant in the understory of shrub and Wetland Forest communities.

Mixed communities of grass and forb species, which occurred under various hydrologic conditions, and in which dominance is ambiguous or composition does not fit Community type decision rules, were grouped as miscellaneous transitional Wet Prairie (H.MxWP). Trends in species composition within this community may be further evaluated to more clearly define types of transitional Wet Prairies. A

Miscellaneous Wetland grass category (H.MxWT) was used to capture graminoids of mixed dominance, where identification was unclear, or the community is too rare to warrant a separate category, such as *Phragmites australis* (common reed), which occurred in small patches along remnant river channels and C-38.

Aquatic Communities

Aquatic Communities were defined as communities of plants that grow in permanently deep aquatic conditions, as opposed to wetlands which are inundated for only part of the year or that occur in shallow water or wet soil and are dominated by hydrophytic species (Cowardin et al. 1979). An exception to this definition is the communities that develop on floating mats that occur in the lower portions of pools, nonflowing remnant river channels, and abandoned channels under channelized conditions. These communities are difficult to characterize. In some cases, floating mats support normally upland (e.g., *Myrica cerifera*) or wetland species (e.g., various shallow-water rooted emergents), interspersed with fully aquatic species in mat openings of open water (such as the floating species *Pistia stratiotes* or *Salvinina minima*), resulting in recurring communities of species that confound aquatic/wetland/terrestrial distinctions. The Aquatic Vegetation Group (AQ) includes continuous floating mats formed by *Scirpus cubensis* (cuban bullrush) (Pierce et al. 1982), on which occur rooted aquatic vegetation, free floating plants, various marsh species, and shrubs. *Scirpus cubensis*- dominated floating mats (H.SCF, H.MFM) accounted for 25% of the Aquatic Vegetation Group.

Shrub-dominated floating mat community types dominated by *Ludwigia* spp. (S.LSF), *Myrica cerifera* (S.MCF), and occasionally *Salix caroliniana*, which are included in the Aquatic Vegetation Bcode Group, were found in the lower sections of pools, abandoned channels, and occasionally in remnant river channels.

Collectively, floating mat communities made up 3% of the total mapped area of Pool C (Table 10-3, Figure 10-3). The *Ludwigia* spp. type was the most common, accounting for 19% of the Aquatic Community Group and occurring mainly in abandoned channels. The aquatic emergent *Polygonum densiflorum* (denseflower smartweed) Community type (H.PD) made up approximately 18% of the Aquatic Vegetation Group, and *Nuphar lutea* (spatterdock) (H.NL), a rooted floating-leaf emergent, accounted for 11% of the Aquatic Group.

Non-vegetated, Human-Influenced, and Problematic Categories

Approximately 4% of the floodplain was unvegetated open water (Table 10-3, Figure 10-3). Approximately 72% of open water in the baseline 1996 data was located in C-38, while remnant river channels and other natural water habitats (e.g. abandoned river channels and depressions) made up almost 27% of the 1996 open water habitat.

The Non-Vegetated Bare Ground category (NVBG) was used to classify areas of sand or mud and the Non-Vegetated Human-Influenced Group (NVH), was used to represent features that were constructed, such as water control structures, houses and lawns, roads, farm complexes, and rip rap. The combined cover of these categories accounted for only 0.09% of the mapped area in Pool C. Polygons that were uninterpretable, or composed of rarely occurring species that do not fit community type decision rules, were grouped together as Unknowns (UN). This category was needed for only 0.2% of the mapped area in Pool C.

CONCLUSIONS

Floodplain vegetation shifted from dominance by wetland vegetation to dominance by upland communities as early as 1973-74 (Table 10-2, Figures 10-1 and 10-2), two to three years after the C-38 canal was completed. Prior to channelization, wetland vegetation occurred on over 80% of the floodplain's total area. By 1974, three years after completion of channelization, wetlands had declined to about 29% of the floodplain. Pre-channelization wetlands were dominated by herbaceous marshes, primarily Broadleaf Marsh and Wet Prairie, which occurred on 46% and 21% of the floodplain, respectively. Wetland Shrub communities (WS) covered 13% of the floodplain prior to channelization. By 1974, Broadleaf Marsh occurred on only 7% of the floodplain, Wet Prairie on 13%, and Wetland Shrub communities had declined to 8% of the floodplain. Much of the gross-level vegetation change in wetlands that took place in Pool C following channelization had occurred by the time the 1974 aerial photography was taken; little additional change in wetland plant communities had occurred in Pool C by 1996. The similarity of areal vegetation cover in Pool C, compared to the entire floodplain in the 1952 and 1974 maps, suggests that extrapolation of this finding to the entire floodplain is not unreasonable.

Much of the loss of wetlands described in this chapter is accounted for by conversion of marshes to upland pastures. These drained areas were used as improved (human-modified) or unimproved grazing lands. Opportunistic Upland Shrub species increased. *Myrica cerifera* occupied higher elevations of formerly long-hydroperiod marshes and sections of lower pools where dryer substrates of floating vegetation formed in permanently wet areas. *Schinus terebinthifolius* colonized banks of the canal and river channels.

These changes were largely a result of lost seasonal inundation of the floodplain marsh communities that had dominated the floodplain prior to channelization. Less important factors affecting the distribution and extent of vegetation included increases in the elevations of former wetland areas where spoil was dumped, loss of flow in riparian and other river channel habitats, development of "floating" substrates for non-aquatic species, and directly human-mediated factors such as introductions of cattle and forage grass species and suppression of shrubs in drained marshes.

Restoration Expectations

Three expectations were developed to predict vegetation change resulting from restoration (Figure 10-4). The restoration expectations are presented in Carnal (2005a, 2005b, and 2005c) by restoration construction phase. These predictions are based on coverage in the 1954 pre-channelization reference vegetation map, overlaid with restoration phase areas (Table 10-4, Map Appendix 9A). Wetland plant communities are expected to eventually comprise approximately 80% of the area restored in restoration Phases I–IV. Broadleaf Marsh communities are expected to cover 50% or more of the Phase I–IV area, and Wet Prairie communities are expected to cover at least 17% of the Phase I–IV area.

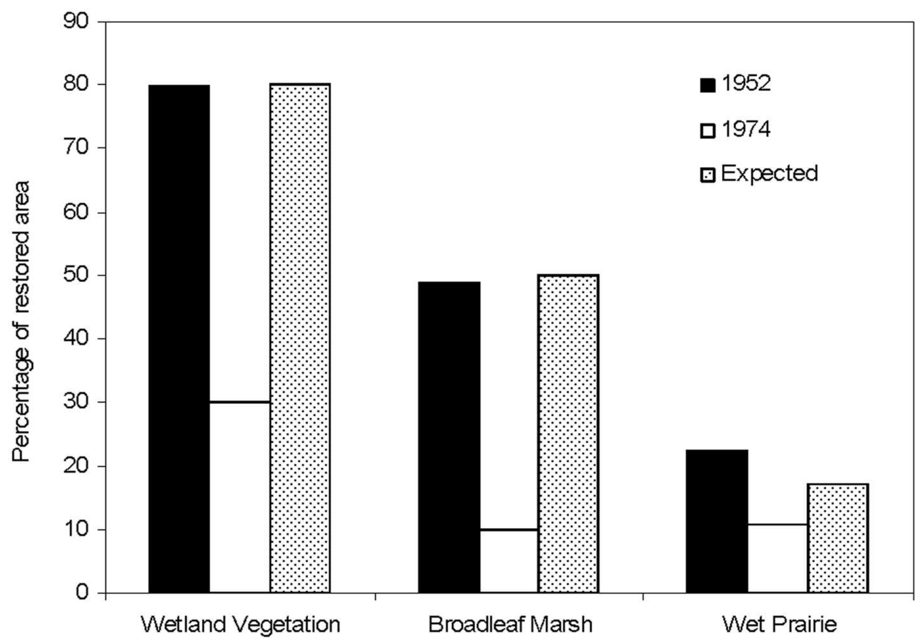

Figure 10-4. Reference, baseline, and predicted area of wetland, Broadleaf Marsh, and Wet Prairie in the restoration project area.

Table 10-4. Areal extent of Bcode Groups by restoration construction phase. The total areas shown are the total area affected by the restoration project. The 1952 pre-channelization reference estimates were used for predictions of restored areal extent of floodplain vegetation.

			Area (hectares)		**Percent of restoration area**	
Restoration phase	**Status**	**Bcode Group**	**1952**	**1974**	**1954**	**1974**
Phase I	Aquatic	Aquatic Vegetation	61.3	35.9	0.6	0.3
	Non-vegetated	Non-Vegetated: Bare Ground		379.3	0.0	3.6
		Non-Vegetated: Human		5.7	0.0	0.1
		Non-Vegetated: Open Water	209.7	176.0	2.0	1.7
	Unknown	Unknown	20.0		0.2	0.0
	Upland	Upland Forest	148.2	269.7	1.4	2.6
		Upland Herbaceous	198.0	1840.6	1.9	17.6
		Upland Shrub	55.6	303.9	0.5	2.9
	Wetland	Broadleaf Marsh	1672.3	174.9	16.0	1.7
		Miscellaneous Wetland	8.6	26.7	0.1	0.3
		Wet Prairie	1185.3	524.7	11.3	5.0
		Wetland Forest	11.6	5.7	0.1	0.1
		Wetland Shrub	276.1	103.5	2.6	1.0
Phase I Total			3846.6	3846.6		
Phase II/III	Aquatic	Aquatic Vegetation	115.5	68.4	1.1	0.7
	Non-vegetated	Non-Vegetated: Bare Ground	0.3	572.9	0.0	5.5
		Non-Vegetated: Human	20.3	34.0	0.2	0.3
		Non-Vegetated: Open Water	440.4	353.9	4.2	3.4
	Unknown	Unknown	16.6	1.3	0.2	0.0
	Upland	Upland Forest	227.6	337.2	2.2	3.2
		Upland Herbaceous	59.2	1384.3	0.6	13.2
		Upland Shrub	102.5	297.5	1.0	2.8
	Wetland	Broadleaf Marsh	2504.2	565.4	23.9	5.4
		Miscellaneous Wetland	32.9	32.4	0.3	0.3
		Wet Prairie	514.5	181.6	4.9	1.7
		Wetland Forest	55.5	35.7	0.5	0.3
		Wetland Shrub	297.5	522.3	2.8	5.0
Phase II/III Total			4386.9	4386.8		
Phase IV	Aquatic	Aquatic Vegetation	25.5	49.9	0.2	0.5
	Non-vegetated	Non-Vegetated: Bare Ground		141.9	0.0	1.4
		Non-Vegetated: Open Water	120.1	77.6	1.1	0.7
		Unknown	6.2		0.1	0.0
	Upland	Upland Forest	66.3	153.1	0.6	1.5
		Upland Herbaceous	64.9	307.0	0.6	2.9
		Upland Shrub	54.3	201.0	0.5	1.9
	Wetland	Broadleaf Marsh	673.5	123.9	6.4	1.2
		Miscellaneous Wetland	16.5	66.0	0.2	0.6
		Wet Prairie	471.3	401.9	4.5	3.8
		Wetland Forest	2.1	7.8	0.0	0.1
		Wetland Shrub	190.3	161.3	1.8	1.5
Phase IV Total			1691.0	1691.4		
Phase IVA	Aquatic	Aquatic Vegetation	7.6	11.8	0.1	0.1
	Non-vegetated	Non-Vegetated: Bare Ground		73.4	0.0	0.7
		Non-Vegetated: Open Water	64.5	48.9	0.6	0.5
		Unknown	2.3		0.0	0.0
	Upland	Upland Forest	25.7	39.2	0.2	0.4
		Upland Herbaceous	3.7	134.9	0.0	1.3
		Upland Shrub	4.0	11.1	0.0	0.1
	Wetland	Broadleaf Marsh	255.6	190.0	2.4	1.8
		Wet Prairie	179.2	16.9	1.7	0.2
		Wetland Forest	2.9	1.9	0.0	0.0
		Wetland Shrub	1.8	19.2	0.0	0.2
Phase IVA Total			547.2	547.2		
Totals of phases			10472	10472	100	100

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CHAPTER 11

AQUATIC INVERTEBRATE COMMUNITY STRUCTURE AND FUNCTIONAL CHARACTERISTICS IN THE KISSIMMEE RIVER-FLOODPLAIN ECOSYSTEM: BASELINE AND REFERENCE CONDITIONS AND EXPECTATIONS FOR RESTORATION

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ABSTRACT: Channelization of the Kissimmee River likely altered aquatic invertebrate community structure and functional characteristics of river channel and floodplain habitats. Remnant river channels are characterized by no flow, low levels of dissolved oxygen, abundant emergent, submergent, and floating vegetation, and thick accumulations of organic matter overlaying pre-channelization sand substrates. The channelized floodplain is characterized primarily by upland pasture, although small areas of remnant, but altered Broadleaf Marsh occur near the southern end of each pool. In order to determine baseline (pre-restoration) conditions, multiple sampling methods were used to determine aquatic invertebrate community structure, functional characteristics, and production in seven river channel and three floodplain habitats. Results indicate that aquatic invertebrate community structure and functional characteristics of the channelized Kissimmee River ecosystem are atypical of unmodified southeastern Coastal Plain blackwater river systems. Aquatic invertebrates of river channel habitats are representative of lentic and depositional habitats rather than flowing water habitats. No flow and isolation of the river channel from the floodplain preclude passive drift and bi-directional exchange of aquatic invertebrates between river channel and floodplain habitats. Floodplain habitats remain dry most of the time, but occasionally support an ephemeral and depauperate aquatic invertebrate community during the wet season. Habitat-specific macroinvertebrate secondary production within the channelized river was highly variable, but generally within the range of values reported for similar habitats in other blackwater river systems. Floodplain macroinvertebrate production was very low, primarily due to sporadic, short-term inundation patterns.

Restoration of the Kissimmee River is expected to alter aquatic invertebrate community structure and secondary production, and reestablish invertebrate drift and food web linkages within and between riverine and floodplain habitats. Shifts in species composition and secondary production, functional feeding and habitat groups, and invertebrate drift will be compared to baseline data and expectations for restoration. Although no historic or baseline data on bi-directional river channel/floodplain exchange exist for the Kissimmee River, post-construction evaluation of this functional attribute will be documented because of its critical role in food web and energy flow dynamics.

INTRODUCTION

Aquatic invertebrates were identified as a critical biological component for assessing restoration of ecological integrity within the Kissimmee River ecosystem (Karr et al. 1991; Harris et al. 1995). Aquatic invertebrates can play an integral role in river ecosystem processes including nutrient cycling (Merritt et al. 1984), decomposition of detritus (Wallace and Webster 1996), and energy flow to higher trophic levels; e.g., amphibians, reptiles, fishes, wading birds, and waterfowl (Weller 1995, Benke et al 2001). Aquatic invertebrates also have a long history of use in biomonitoring (Plafkin et al. 1989, Rosenberg and Resh 1993), and can serve as indicators of biotic integrity and ecological health (Karr 1991).

The pre-channelized Kissimmee River was characterized by a diverse littoral zone composed of submerged, emergent, and floating plants, shifting sand substrate, and minimal amounts of large woody debris (Toth et al. 1995). The river was highly stained with dissolved organic carbon primarily derived from the flanking floodplain and contributing watersheds. Dissolved oxygen levels varied seasonally, but likely ranged from 3–7 mg/L (Colangelo 2005). Discharge exceeded 11 m^3 per second 90–95% of the period of record, with highest discharge generally occurring near the end of the wet season (September–November). Average in-stream velocities ranged from 0.3–0.6 m/second. Pre-channelization stage data indicate that the Kissimmee River experienced a seasonal wet-dry cycle; however, only peripheral areas of the floodplain underwent consistent annual seasonal drying. Most floodplain habitats remained inundated for long periods (e.g., approximately 77% of the floodplain was inundated for 76% of the historical period of record (Toth et al. 1995) with water depths ranging from 0.3–0.7 meters (Koebel 1995). These river channel-floodplain characteristics likely shaped aquatic invertebrate community characteristics and rates of secondary production.

Elimination of flow through remnant channels and conversion of wetlands to pasture likely altered aquatic invertebrate community structure, and disrupted critical food web linkages within and between riverine and floodplain habitats. Under these hydrologic conditions, aquatic invertebrate taxa inhabiting the remnant (non-flowing) river channels are more characteristic of lentic or palustrine systems rather than a flowing river (Vannote 1971, Toth 1993, Warren and Holt 1996). Colonization and production of aquatic invertebrates in remnant Broadleaf Marsh is limited to short periods when summer rains temporarily inundate floodplain habitats, and because exchange of organic matter between the floodplain and the river channel is rare, passive drift by aquatic invertebrates is likely nonexistent.

Restoration of pre-channelization hydrology, including continuous, variable flow and long-term floodplain inundation frequencies, is expected to reestablish historic river channel and floodplain habitats, and aquatic invertebrate community structure characteristics. Specific changes likely will include shifts in functional feeding and functional habitat associations among primary river channel habitats (i.e., large woody debris and sandy benthos), increased macroinvertebrate species richness and diversity among floodplain habitats, and increased passive drift by macroinvertebrates.

Objectives

The objectives of this study are: (1) to assess baseline (pre-restoration) aquatic invertebrate community structure characteristics of the channelized Kissimmee River and floodplain; (2) to estimate rates of aquatic invertebrate secondary production for river channel and floodplain habitats; (3) to document aquatic invertebrate drift within the river channel; (4) to estimate reference conditions for aquatic invertebrate community structure in primary river channel habitats; (5) to estimate reference conditions for floodplain aquatic invertebrate communities; (6) to estimate reference conditions for aquatic invertebrate drift within the river channel; (7) to quantify impacts of channelization by comparing pre-channelization (reference) conditions and baseline conditions; and (8) to define and discuss specific expectations for restoration of aquatic invertebrate community structure and production in primary river channel habitats, aquatic invertebrate community structure in floodplain habitats, and aquatic invertebrate drift within the river channel.

METHODS

Baseline Conditions

Study Site

Aquatic invertebrate community structure characteristics and functional attributes were examined in seven remnant river channel and three floodplain habitats in Pools A, C, and D of the channelized Kissimmee River. Under channelized (baseline) conditions, remnant river channels are characterized by no flow, consistently low levels of dissolved oxygen (generally <2 mg/L) (Colangelo 2005), excessive growth of in-channel vegetation, and large accumulations of organic matter over benthic substrates. Sampled river channel habitats included *Nuphar lutea* (H.NL, *Nuphar lutea* herbaceous aquatic vegetation, Bousquin and Carnal 2005), *Polygonum densiflorum* (H.PD, *Polygonum densiflorum* herbaceous aquatic vegetation, Bousquin and Carnal 2005), *Scirpus cubensis* (H.SCF, *Scirpus cubensis* herbaceous floating mat vegetation, Bousquin and Carnal 2005), *Ceratophyllum/Hydrilla* (H.MxSV, miscellaneous submerged vegetation, Bousquin and Carnal 2005), Mid-channel Benthic (BENT), Mid-channel Water Column (ZOOP), and Woody Debris (SNAG). Snags were defined as any submerged dead wood greater than 1" in diameter. See Bousquin and Carnal (2005) for more detailed vegetation classification scheme.

Sampled floodplain habitats included Broadleaf Marsh (BLM) (Bousquin and Carnal 2005), Woody Shrub (S.MCF, *Myrica cerifera* Floating Mat Shrubland, Bousquin and Carnal 2005), and Woody Debris (FSNAG). Remnant Broadleaf Marsh habitats are spatially homogeneous and dominated by arrowhead (*Sagittaria lancifolia*), pickerelweed (*Pontederia cordata*), and maidencane (*Panicum hemitomon).* Woody Shrub habitats are characterized by dense stands of wax myrtle (*Myrica cerifera*) that exist on a bog-like floating mat. The understory is composed of a diverse mixture of broadleaf marsh, wet prairie, and upland vegetation including broomsedge (*Andropogon glomeratus*), sedges (*Cyperus* spp.), pennywort (*Hydrocotyle umbellata*), spatterdock (*N. lutea*), rushes (*Rynchospora* spp.), and Cuban bulrush (*S. cubensis*). See Bousquin and Carnal (2005) for more detailed vegetation classification scheme.

Aquatic Invertebrate Community Structure

Aquatic invertebrates were sampled quarterly over a two-year period from August 1995–May 1997. Three replicate samples were collected from each river channel and floodplain habitat within Impact and Control sites (when available) on each sample date. Control sites included three remnant river channels (Ice Cream Slough Run, Rattlesnake Hammock Run, and Persimmon Mound Run) and remnant BLM (Latt Maxcy Floodplain) in Pool A. These sites will not be affected by restoration and will serve as long-term Control sites. An additional short-term Control site was established in Pool D Woody Shrub (S.MCF). This site will be impacted by restoration construction during Phase II/III (2008–2010). Impact sites included three remnant river channels (Oxbow 13, Micco Bluff Run, and MacArthur Run), remnant BLM (Pool C Broadleaf Marsh), and Pool C Woody Shrub (S.MCF). These sites will be affected following Phase I construction. Sampling locations within remnant channels were selected by traveling at a constant speed (~ 1000 rpms) for a randomly determined time through the channel, and continuing until the next appropriate habitat type was encountered. Floodplain sample locations were selected by traveling a randomly determined distance (≤400 m) and compass direction from a randomly determined location on the floodplain. All samples were preserved in the field with 5–10% formalin stained with rose bengal. Each sample was located in space and time with a Global Positioning System (GPS) with sub-meter accuracy. For each sample, ancillary data including water temperature, specific conductance, pH, and dissolved oxygen were recorded at a depth of 15 cm below the water surface using a Hydrolab™ or YSI™ multi-probe water quality instrument. In shallow floodplain habitats, water quality parameters were generally recorded within the first 5 cm of the water column. Water depth was recorded at each location with a meter stick or PVC pole calibrated in 5 cm intervals. Current velocity was measured in the river channel with a Marsh-McBirney series 2000 flow meter. A continuous record of river channel surface water temperature at Impact and Control sites was recorded using a HOBO™ temperature logger. Missing values in this record were estimated from a regression developed from this data set, and air temperature records from Archbold Biological Station, Lake Wales, Florida (D. H. Anderson, SFWMD, personal communication).

Preserved samples were sieved into two size classes using 1 mm (coarse fraction) and 125 µm (fine fraction) mesh sieves. All invertebrates were hand-picked from the coarse fraction using a dissecting microscope at 6–12X magnification, and preserved in 70% ethanol. The fine fraction was elutriated to separate organic matter from inorganic matter. The organic matter portion was sub-sampled to a fraction that could be processed in approximately two hours (usually 1/8–1/64). All invertebrates were counted and identified to the lowest taxonomic level using Thorp and Covich (1991), Merritt and Cummins (1996), Epler (1992, 1996), and Thompson (1984). For most taxa, individual biomass was estimated from published length-mass regressions (Benke et al. 1999, Meyer 1989, Culver et al. 1985, Rosen 1981, Anderson and Benke 1994, Anderson et al. 1998a, Lei and Armitage 1980, Fleeger and Palmer 1982). For mites, we used a dry mass of 0.06 mg/individual (D.H. Anderson, unpublished data). For nematodes and leeches, individual mass was estimated volumetrically by assuming a cylindrical shape, a specific density of 1.05, and a dry mass content of 15% (Strayer and Likens 1986). Oligochaetes were dried for four hours at 60°C and weighed to the nearest 0.001 gram.

Abundance and biomass estimates for each taxon in each sample were weighted by sampler area to standardize estimates to numbers/m^2 or grams/m^2, respectively. Mean quarterly density and biomass for each taxon was determined by averaging its sample density and biomass for each replicate on each date. Mean annual density and biomass were determined by averaging the four quarterly estimates of density and biomass. For dates when habitats were not available (e.g., dry floodplain), zeros were averaged to obtain estimates of mean annual density and biomass. Zeros were not included in the calculations when poorly preserved samples were discarded.

Organic matter in the coarse fraction was classified as macrophyte, wood, or detritus, and dried at 60°C for 24 hours. Dried material was weighed to the nearest 0.001 gram, ashed at 450°C for 4 hours, and re-weighed to determine ash-free-dry-mass (AFDM). Ash-free-dry-mass also was determined for organic and inorganic matter from the fine fraction.

Community structure was described by species richness (S = the total number of species present), species diversity (H'), where $H' = -\Sigma(p_i \ln p_i)$ and p_i is the proportion of species belonging to the i^{th} taxa, and community evenness (J'), where $J' = H'/\ln S$ (Price 1984). Taxa were assigned to functional feeding groups according to Merritt and Cummins (1996), Rader (1994), Borror et al. (1989), Merritt et al. (1996) for aquatic insects, and Rader (1994), Gladdon and Smock (1990), and Balcer et al. (1984) for non-insects. Functional feeding group categories included filtering-collectors (FCOLL), gathering-collectors (GCOLL), predators (PRED), scrapers (SC), shredders (including shredders of coarse particulate organic matter and vascular plants) (SHRD), and vascular plant piercers (PRC). Taxa also were classified into four functional habitat groups - LENTIC (only occurring in standing water), LOTIC (only occurring in flowing water), BOTH (occurring in lentic or lotic habitats), and DEP (occurring in lentic or lotic depositional zones). Functional habitat groups were based on the classification in Merritt and Cummins (1996) and supplemented with information from Epler (1996), Tressler (1959), and Thompson (1984).

Analysis of variance (ANOVA; SYSTAT version 8) was used to test for differences in total sample organic matter and mean annual density. These analyses used a randomized block design with site (Impact and Control) as the treatment and habitat blocks. The natural logarithm of total organic matter and total density was used to make the variance independent of the mean. Pairwise comparisons were made using Tukey's HSD test that controls the experiment-wise error rate (Day and Quinn 1989). When sample sizes are uneven, SYSTAT uses the Tukey-Kramer modification that maintains the experiment-wise error rate at or below the nominal level, and is more powerful than most pairwise comparison methods (Day and Quinn 1989). Unless otherwise stated, all statistics are significant at $p < 0.05$.

Secondary Production

Secondary production was estimated using the instantaneous growth rate method, which requires knowledge of individual biomass and growth (Benke 1993). For most taxa, the appropriate length dimension was measured with an ocular micrometer, and individual biomass was estimated from length-mass regressions. Growth rates were estimated from published growth equations (Morin and Dumont 1994, Pickard and Benke 1996, Benke and Jacobi 1994, Hauer and Benke 1991, Anderson and Benke 1994, Anderson et al. 1998a). A growth equation for grass shrimp (*Palaemonetes paludosus*) in the Kissimmee River was developed for this study. A growth equation for crayfish, developed for a congeneric species (*Procambarus alleni*) from wetlands in the Lake Okeechobee basin, also was used. These equations predict daily growth rate from temperature and individual mass.

To estimate annual production, each year of the baseline period was divided into four equal intervals centered on the quarterly sampling date. For each taxon in each sample, secondary production was estimated as the product of biomass, daily growth rate, and number of days in the interval. Production and biomass estimates for each sample were averaged to obtain a mean for each interval. Interval production estimates were summed to obtain annual production, and biomass for each quarter was averaged to obtain mean annual biomass. Annual P/B was obtained by dividing annual production by mean annual biomass.

Aquatic Invertebrate Drift

Aquatic invertebrate drift samples were collected approximately quarterly beginning in January 1998. Paired drift nets (900 cm^2 equipped with 125 µm mesh netting) were placed 15 cm below the water surface and 0.5 m above the substrate at three locations within each of three remnant river channels in Pools A and C. Because there is no flow through remnant channels, there was little risk of nets becoming clogged; therefore, samples were collected at eight-hour intervals ($\pm$ 1 hour) over a 24-hour period. Current velocity at each surface and bottom net opening, wind direction, and wind velocity were measured whenever a net was set or removed. All samples were preserved in the field with 10% buffered formalin stained with rose bengal.

Preserved samples were rinsed through a 125 µm mesh sieve and sub-sampled to a fraction that could be processed in approximately two hours (usually 1/32–1/64). All invertebrates were hand-picked using a dissecting microscope at 12X magnification, and preserved in 70% ethanol.

RESULTS

Habitat Characteristics

Mean annual water temperature in remnant channels was 23°C in year one and 25°C in year two and differed by less than 0.5°C between Control and Impact sites (Figure 11-1). Approximately 90% of all current velocity measurements in all habitats were 0.0 m/s, with only two values >0.2 m/s. Mean annual values for dissolved oxygen, pH, and specific conductance (Figure 11-2) were similar across habitats and sites. Surface dissolved oxygen concentrations were typically low, with a mean baseline value averaged across all habitats and sites, of 2.9 mg/l.

Mean organic matter content of samples (Figure 11-3) was significantly different among habitats (ANOVA, p <0.01) but not between Control and Impact sites (ANOVA, p >0.05). Organic matter composition also varied among habitats, but showed similar patterns at Control and Impact sites (Figure 11-4).

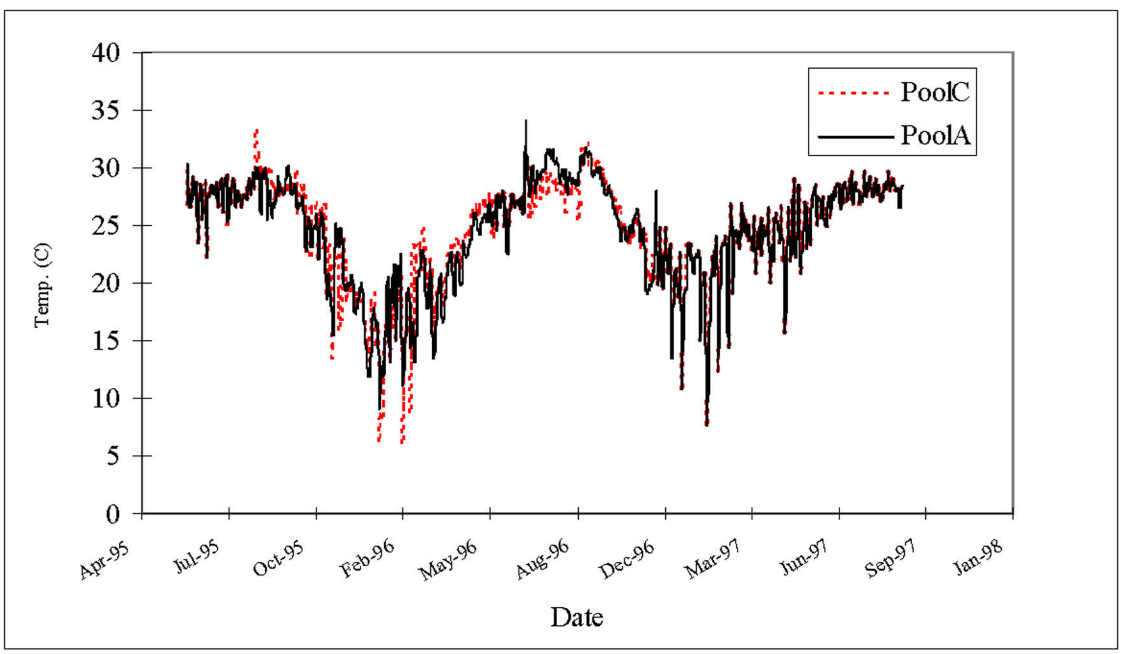


Figure 11-1. Daily water temperature at Impact and Control sites during the baseline study period.

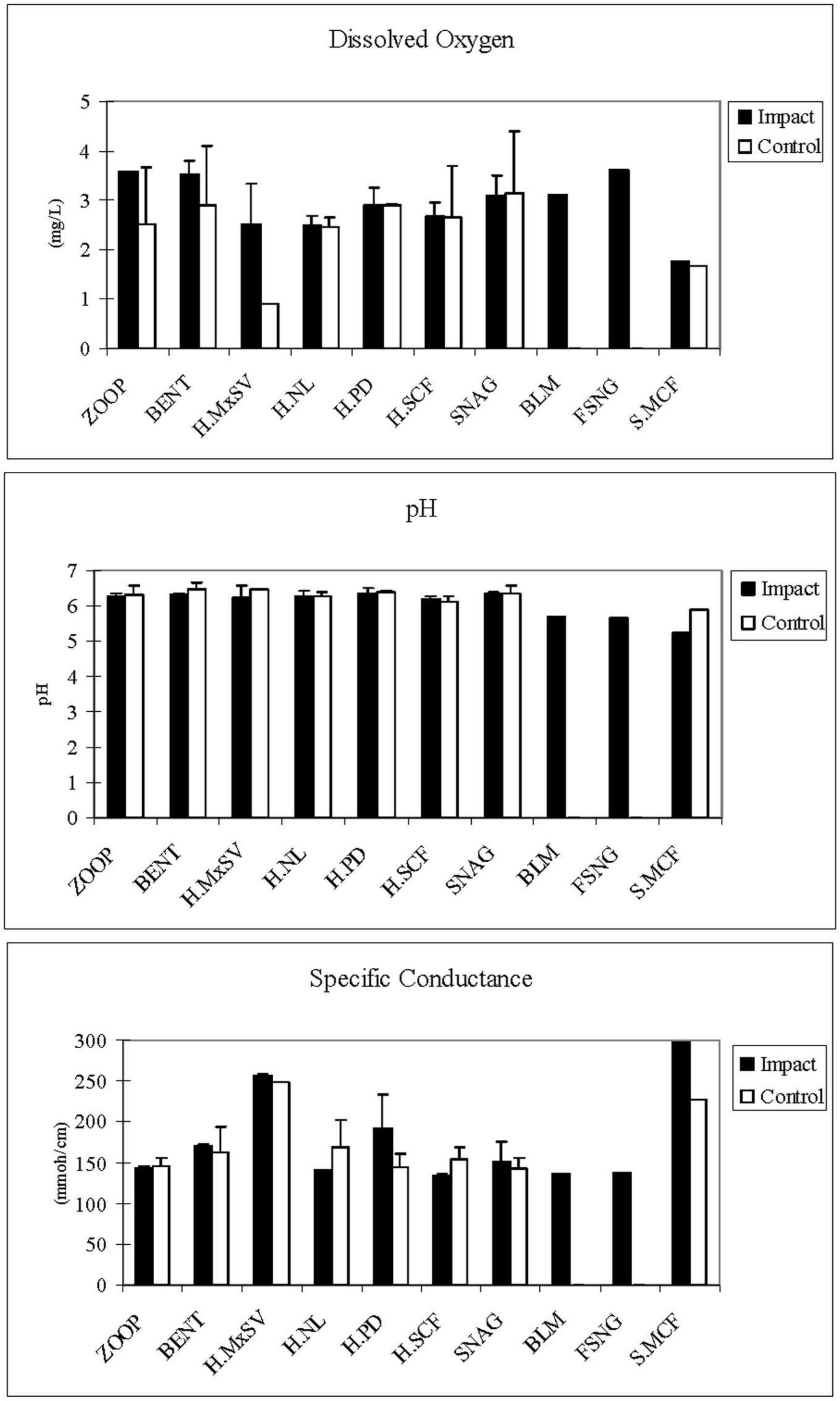


Figure 11-2. Mean annual values for dissolved oxygen (top), pH (middle), and specific conductance (bottom) in Pools A (Control) and C (Impact). ZOOP = Mid-channel Water Column, BENT = Mid-channel Benthic, H.MxSV = Miscellaneous Submerged Vegetation, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, SNAG = River Channel Woody Debris, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, S.MCF = Woody Shrub.

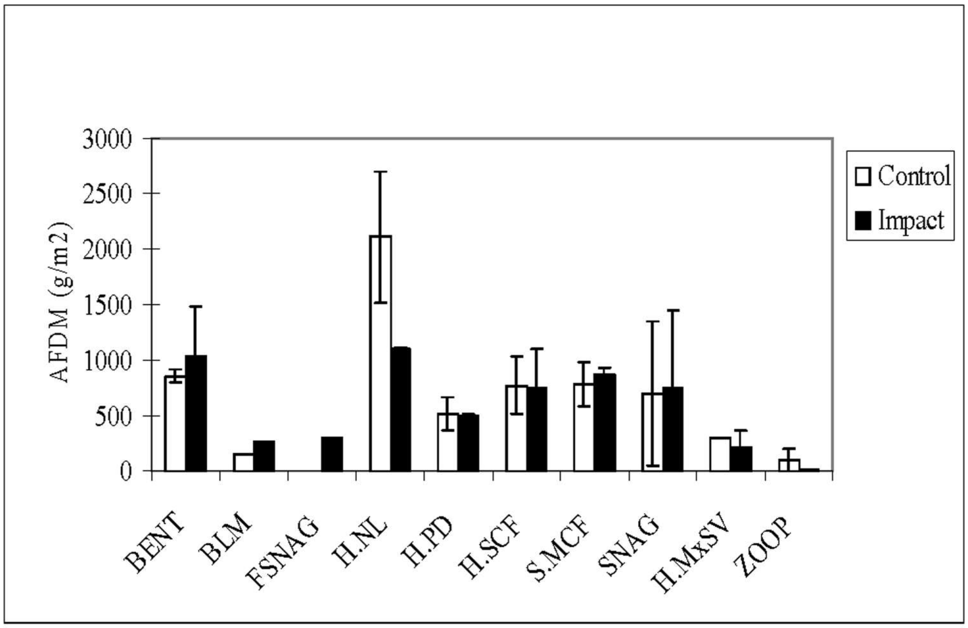


Figure 11-3. Total ash-free-dry-mass (AFDM) of organic matter from replicate samples averaged across dates and years for all habitats. BENT = Mid-channel, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, , S.MCF = Woody Shrub, SNAG = River Channel Woody Debris, H.MxSV = Miscellaneous Submerged Vegetation, ZOOP = Mid-channel Water Column.

Aquatic Invertebrate Community Structure

One hundred and eighty-seven taxa of aquatic invertebrates were collected from remnant river channel and floodplain habitats. Coleoptera (48 genera), chironomids (26 genera), and microcrustaceans (42 genera) accounted for 62% of all taxa. Two additional taxa, *Corbicula fluminea*, and the native unionid mussel, *Elliptio buckleyi*, were not sampled quantitatively; however, qualitative collections of both species occurred at several locations along the river.

Taxa richness, taxa diversity, and community evenness varied among habitats and sites, with higher values generally occurring at the Impact site; however, differences among habitats tended to be greater than those between sites (Figure 11-5). Highest richness and diversity occurred in *Nuphar* (H.NL), *Polygonum* (H.PD), *Scirpus* (H.SCF), snag (SNAG), and Woody Shrub (S.MCF) habitats at Impact and Control sites; however, diversity values were usually <2.0. Community evenness exceeded 0.5 for all habitats except Broadleaf Marsh (BLM), Floodplain Snag (FSNAG), and Submerged Vegetation (H.MxSV). Seasonal patterns were not apparent for taxa richness, taxa diversity or community evenness.

Mean annual density (Figure 11-6) ranged from 6049/m^2 in BLM to 134,871/m^2 in H.SCF at the Control site, and 1732/m^2 in FSNAG to 232,997/m^2 in H.SCF at the Impact site. There were no significant differences in the natural logarithm of total density between Control and Impact sites (ANOVA, $p > 0.05$), but there were significant differences for habitat blocks (ANOVA, $p < 0.01$). Mean annual density in SCIR was significantly higher than all other habitats (Tukey's HSD, $p < 0.05$). Density showed no seasonal pattern at either Control or Impact sites.

Core taxa were identified as those that accounted for at least 5% of mean annual abundance in any habitat at either site. Seventeen core taxa (ten at the Control site, 14 at the Impact site, and seven at both sites) were identified, and accounted for 26–86% of mean annual density in each habitat (Tables 11-1 and 11-2). Most core taxa occurred in most habitats, but their relative abundance varied among habitats.

Gathering-collectors accounted for the largest fraction of individuals sampled in most habitats (Figure 11-7). Microcrustacean filtering-collectors were most abundant in mid-channel open water samples, and were well represented in most habitats, often accounting for 20% of total numbers, and over 40% of total number in mid-channel samples (ZOOP). Macroinvertebrate passive filtering-collectors were absent from most habitats, never accounting for >2% of total numbers on any sampling date. Predators and scrapers accounted for most of the remaining individuals in most habitats.

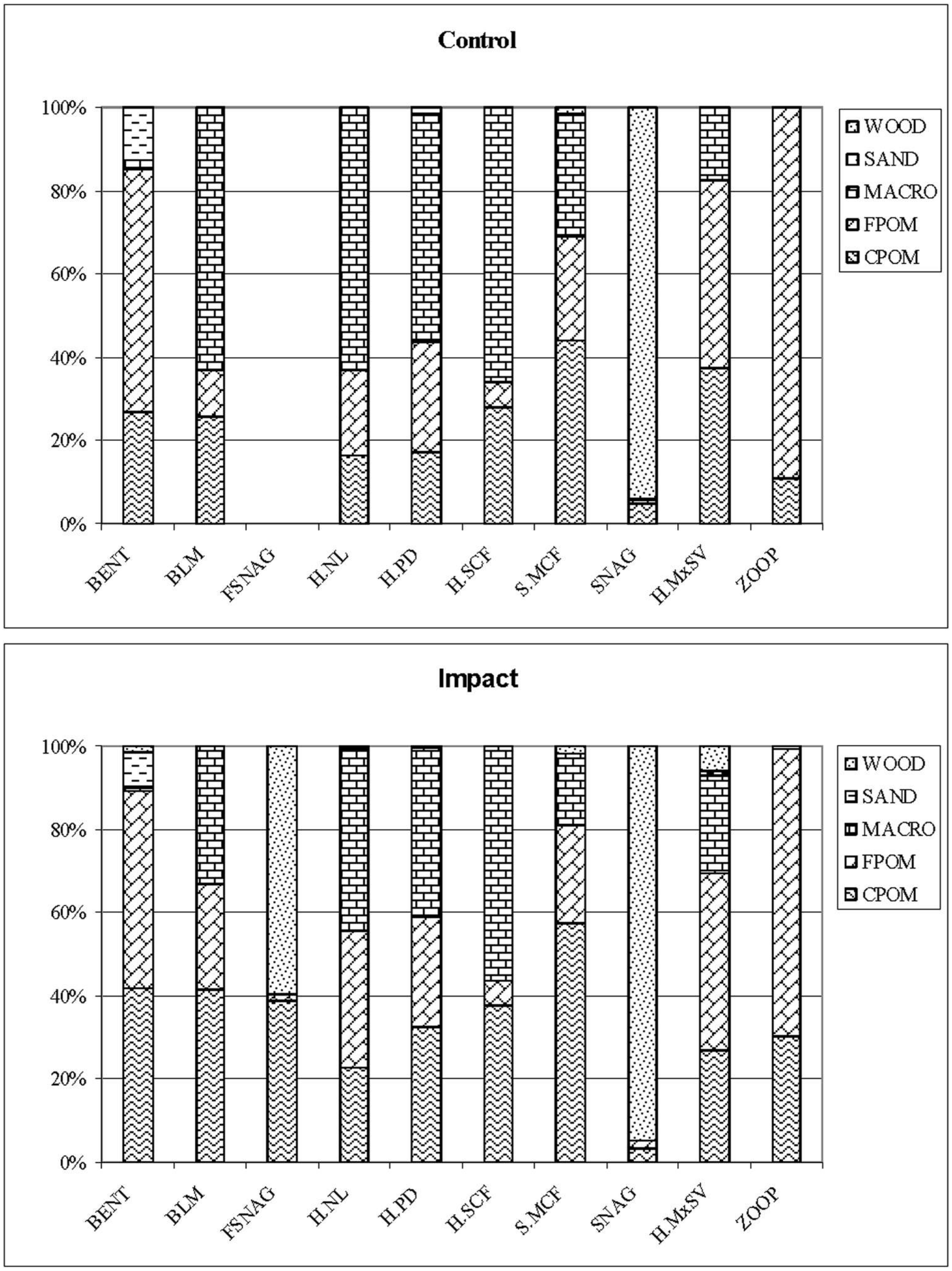


Figure 11-4. Organic matter composition among habitats at Impact and Control sites. BENT = Mid-channel, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, , S.MCF = Woody Shrub, SNAG = River Channel Woody Debris, H.MxSV = Miscellaneous Submerged Vegetation, ZOOP = Mid-channel Water Column.

Functional habitat group composition varied among habitats but was similar among sites (Figure 11-8). Taxa typical of lotic habitats were rare and comprised <3% of mean annual density in each habitat. Taxa typical of lentic habitats accounted for the largest fraction of mean annual abundance, often exceeding 50% in most habitats at both Control and Impact sites. Taxa typical of lentic habitats or lotic depositional areas (BOTH) accounted for the next highest fraction.

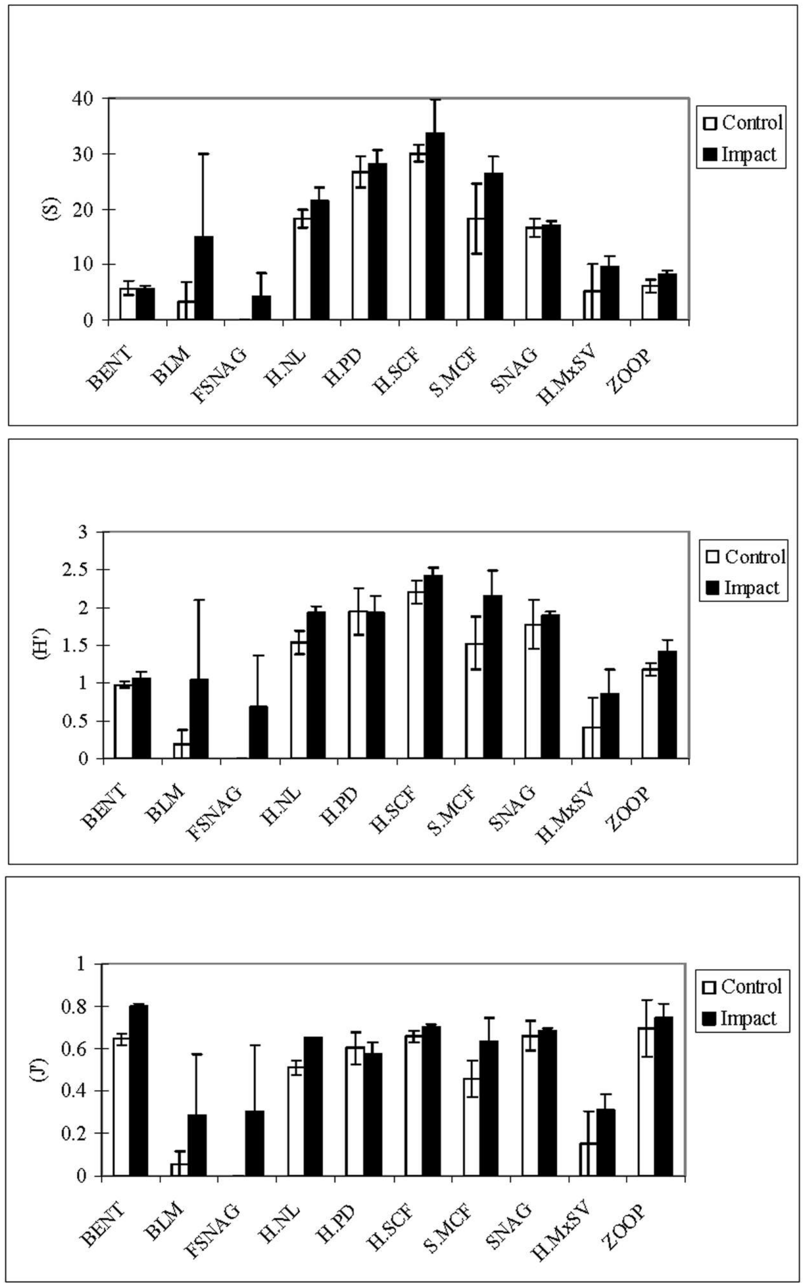


Figure 11-5. Mean taxa richness (top), diversity (middle), and community evenness (bottom) at Control and Impact sites. BENT = Mid-channel, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, , S.MCF = Woody Shrub, SNAG = River Channel Woody Debris, H.MxSV = Miscellaneous Submerged Vegetation, ZOOP = Mid-channel Water Column.

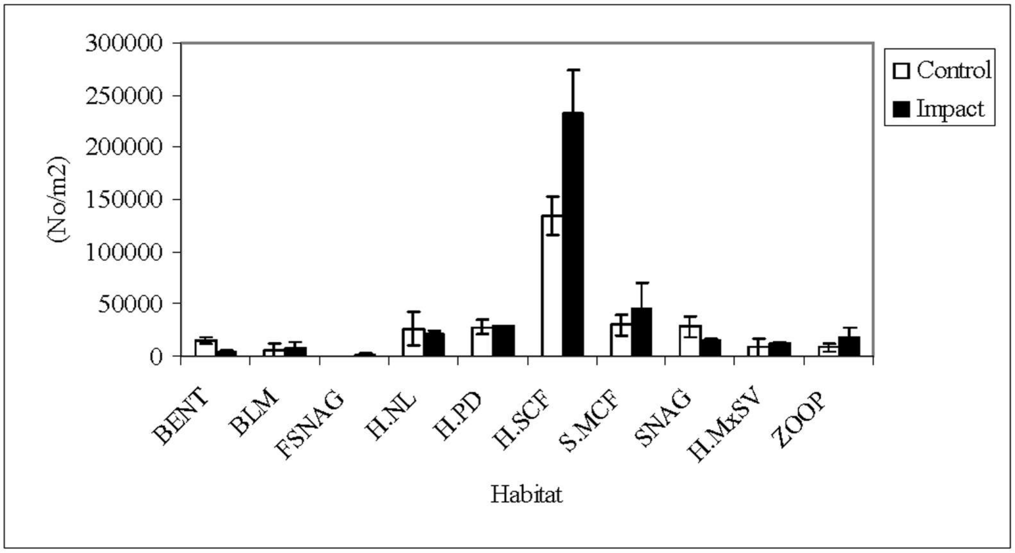


Figure 11-6. Mean annual invertebrate density for each habitat at Control and Impact sites. Bars represent mean ± SE of mean annual density for two baseline years. BENT = Mid-channel, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, , S.MCF = Woody Shrub, SNAG = River Channel Woody Debris, H.MxSV = Miscellaneous Submerged Vegetation, ZOOP = Mid-channel Water Column.

Table 11-1. Mean (SE) baseline density (no/m^2) for core taxa (bold type) at the Control site. Habitats are arranged from mid-channel to the edge of the floodplain. ZOOP = Mid-channel Water Column, BENT = Mid-channel Benthic, H.MxSV = Miscellaneous Submerged Vegetation, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, SNAG = River Channel Woody Debris, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, S.MCF = Woody Shrub.

Taxon	ZOOP	BENT	H.MxSV	H.NL	H.PD	H.SCF	SNAG	BLM	S.MCF
Acari	133	96	284	160	494	2938	654	**3840**	**6366**
	(133)	(93)	(284)	(160)	(185)	(2783)	(154)	(3840)	(546)
Caecidotea	0	0	0	1	88	1409	260	0	**4560**
				(1)	(34)	(1229)	(12)		(2578)
Chironomus/Goeldchironomus	42	245	170	**863**	529	2236	425	0	1801
	(42)	(57)	(170)	(295)	(238)	(1277)	(62)		(466)
Cypria/Physocypria	**2145**	**10322**	**4191**	**6354**	**6122**	**14203**	373	0	38
	(1444)	(2249)	(4191)	(5227)	(1921)	(2791)	(242)		(38)
Dicrotendipes	11	0	362	252	**781**	**8407**	**1476**	0	43
	(11)		(362)	(189)	(362)	(6081)	(502)		(33)
Hyalella azteca	85	19	42	**960**	**2647**	**19836**	**5494**	19	904
	(64)	(19)	(42)	(364)	(841)	(1967)	(3652)	(19)	(347)
Macrocylops	**717**	264	**580**	757	**3844**	**13961**	84	0	**4845**
	(441)	(113)	(580)	(501)	(873)	(4228)	(14)		(1005)
Osphranticum	122	76	1	19	429	415	5	**308**	**2027**
	(80)	(76)	(1)	(19)	(71)	(415)	(3)	(308)	(124)
Paracyclops	106	28	0	179	**666**	3288	11	**307**	**5032**
	(85)	(9)		(179)	(666)	(1870)	(2)	(307)	(2648)
Polypedilum	32	57	40	**1162**	302	3693	339	247	762
	(11)	(57)	(40)	(695)	(46)	(1057)	(196)	(247)	(169)
Others	5181	3663	2954	16037	12139	64484	19300	1327	762
	(2145)	(225)	(2954)	(10242)	(4203)	(15268)	(5038)	(1327)	(169)
Total	8573	14770	8623	26744	28040	134871	28424	6049	29944
	(4432)	(2878)	(8623)	(16555)	(6748)	(18582)	(9417)	(6049)	(10149)

Secondary Production

Annual production and mean annual biomass varied with habitat but showed similar patterns at the Control and Impact sites (Figure 11-9). Differences in production across habitats tended to parallel differences in biomass. Production and biomass were much higher in H.SCF than any other habitat. Estimates of baseline annual P/B tended to be more uniform, and generally exceeded 20 for most habitats (Figure 11-9).

Table 11-2. Mean (SE) baseline density (No/m^2) for core taxa (bold type) at the Impact site. Habitats are arranged from mid-channel to the edge of the floodplain. ZOOP = Mid-channel Water Column, BENT = Mid-channel Benthic, H.MxSV = Miscellaneous Submerged Vegetation, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, SNAG = River Channel Woody Debris, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, S.MCF = Woody Shrub.

Taxon	ZOOP	BENT	H.MxSV	H.NL	H.PD	H.SCF	SNAG	BLM	FSNG	S.MCF
Acari	133	17	89	289	702	3128	**678**	231	**429**	**6797**
	(48)	(2)	(32)	(236)	(0)	(1552)	(461)	(231)	(429)	(5824)
Bosmina	**1566**	106	10	0	0	0	0	0	0	0
	(802)	(83)	(10)							
Cypria/Physocypria	**5112**	**437**	**5139**	**8314**	**7175**	**45734**	**511**	83	42	386
	(4985)	(3)	(4351)	(4332)	(2772)	(30673)	(470)	(83)	(42)	(181)
Daphnia	**1232**	**111**	0	6	0	0	0	0	0	0
	(403)	(2)		6						
Diaptomidae	**2320**	**217**	0	6	0	0	0	0	0	0
	(664)	(47)		(6)						
Dicrotendipes	16	7	**609**	621	**1721**	**16549**	**2526**	70	29	370
	(5)	(7)	(556)	(103)	(1159)	(7726)	(288)	(70)	(29)	(193)
Eucyclops	202	80	100	**2336**	224	6555	42	**570**	2	390
	(180)	(14)	(100)	(1862)	(19)	(4161)	(30)	(570)	(2)	(83)
Glyptotendipes	5	85	73	94	210	1658	**1916**	4	42	0
	(5)	(85)	(29)	(76)	(36)	(260)	(20)	(4)	(42)	
Hyalella azteca	5	73	**1081**	**1227**	**1854**	**9767**	**3028**	44	4	737
	(5)	(21)	(1072)	(794)	(145)	(959)	(575)	(44)	(4)	(198)
Macrocyclops	**1529**	**231**	**545**	**1111**	**3502**	**25377**	85	**773**	33	646
	(446)	(109)	(91)	(220)	(1219)	(7740)	(48)	(773)	(33)	(275)
Osphranticum	**287**	47	**415**	118	**2172**	4221	0	**702**	0	451
	(244)	(47)	(415)	(118)	(811)	(1301)		(702)		(144)
Paracyclops	**1396**	52	20	154	307	7220	0	112	**89**	**2227**
	(483)	(33)	(20)	(102)	(205)	(1248)		(112)	(89)	(435)
Simocephalus	239	**1142**	132	**817**	302	3287	100	22	0	0
	(218)	(1142)	(132)	(364)	(215)	(2431)	(81)	(22)		
Tanytarsini	0	47	**769**	**682**	948	8524	**782**	122	40	272
		(47)	(575)	(574)	(132)	(3618)	(600)	(122)	(40)	(67)
Others	4369	1547	2618	4981	9303	100976	5886	4434	1021	34240
	(3010)	(521)	(4)	(1348)	(1994)	(59483)	(1330)	(4434)	(1021)	(18311)
Total	18409	4226	11598	20758	28421	232997	15552	7168	1732	24379
	(9746)	(1984)	(1636)	(3824)	(935)	(41338)	(553)	(7168)	(1732)	(24379)

Twenty core taxa accounted for at least 5% of the baseline annual production across all habitats at the Control site (Table 11-3); twenty-one core taxa were identified at the Impact site (Table 11-4). Twelve of these were core taxa at both sites, but they were not always core taxa in the same habitats. Approximately 75% of core taxa in both pools are characteristic of lentic or depositional habitats (Anderson et al. 1998).

Functional feeding group contributions to annual production varied with habitat, but tended to show similar trends at both the Control and Impact sites (Figure 11-10). Gathering-collectors generally accounted for the largest fraction of production. Filtering-collectors (active and passive) rarely accounted for >10% of annual production except in mid-channel open water habitats (ZOOP), where they accounted for 27% and 51% at the Control and Impact sites, respectively; however, this guild was dominated by active filtering-collector microcrustaceans.

Functional habitat groups show fairly consistent patterns across habitats at both the Control and Impact sites (Figure 11-11). Taxa typical of lentic habitats (LENTIC) account for about half of annual production in most habitats. Taxa typical of depositional zones (DEP) account for the next largest percentage of annual production. The only departures from this pattern are in floodplain habitats, where taxa that can occur in both lentic and lotic habitats (BOTH) account for a larger fraction. This is primarily due to the production of aquatic mites that are common in both lentic and lotic habitats. Taxa typical of lotic conditions (LOTIC) account for a very small fraction of annual production.

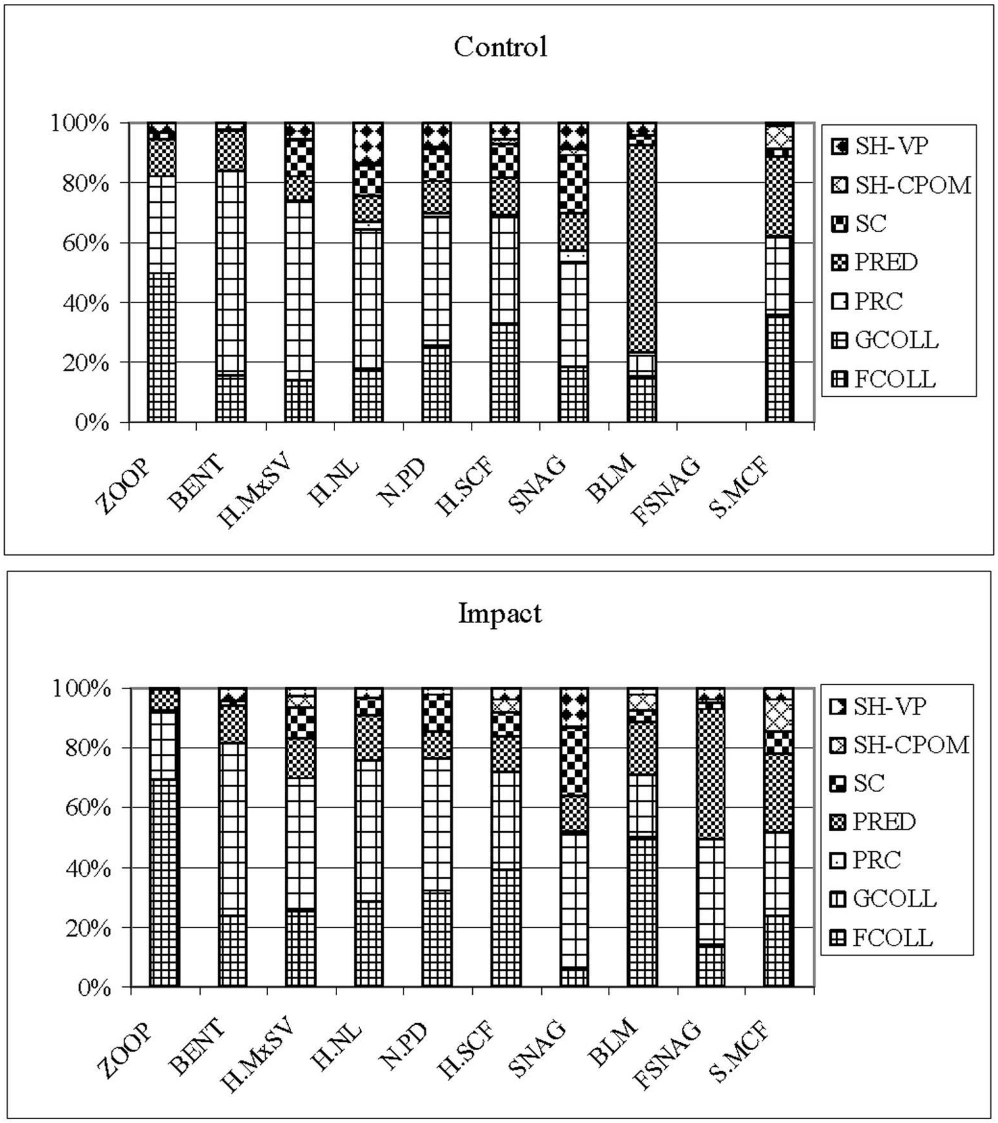


Figure 11-7. Mean functional feeding group composition, based on total abundance, for each habitat at Control and Impact sites. ZOOP = Mid-channel Water Column, BENT = Mid-channel Benthic, H.MxSV = Miscellaneous Submerged Vegetation, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, SNAG = River Channel Woody Debris, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, S.MCF = Woody Shrub.

DISCUSSION

The sampling strategy used in this study was intended to broadly characterize habitat-specific aquatic invertebrate community structure in remnant river channel and floodplain habitats. Because no quantitative invertebrate data exist for the pre-channelized Kissimmee River, our baseline data is compared to data from unregulated southeastern Coastal Plain rivers, with the understanding that any inferences about impacts of channelization must consider other factors (e.g., introduction of exotics, biogeography) that can influence community structure characteristics. For instance, the channelized Kissimmee supports a guild of scraping invertebrates (e.g., snails and *Hyalella azteca*) that is rare in other Coastal Plain rivers. The presence of scrapers is not an obvious consequence of channelization, but may reflect other differences between these rivers, including a greater abundance of macrophytes and associated periphyton, which provide a surface and food source for grazers. Additionally, high water column calcium concentrations (10–20 mg/L) in the Kissimmee (SFWMD unpublished data) may be more favorable for snail growth than water chemistry in other Coastal Plain rivers (Stites et al. 1995).

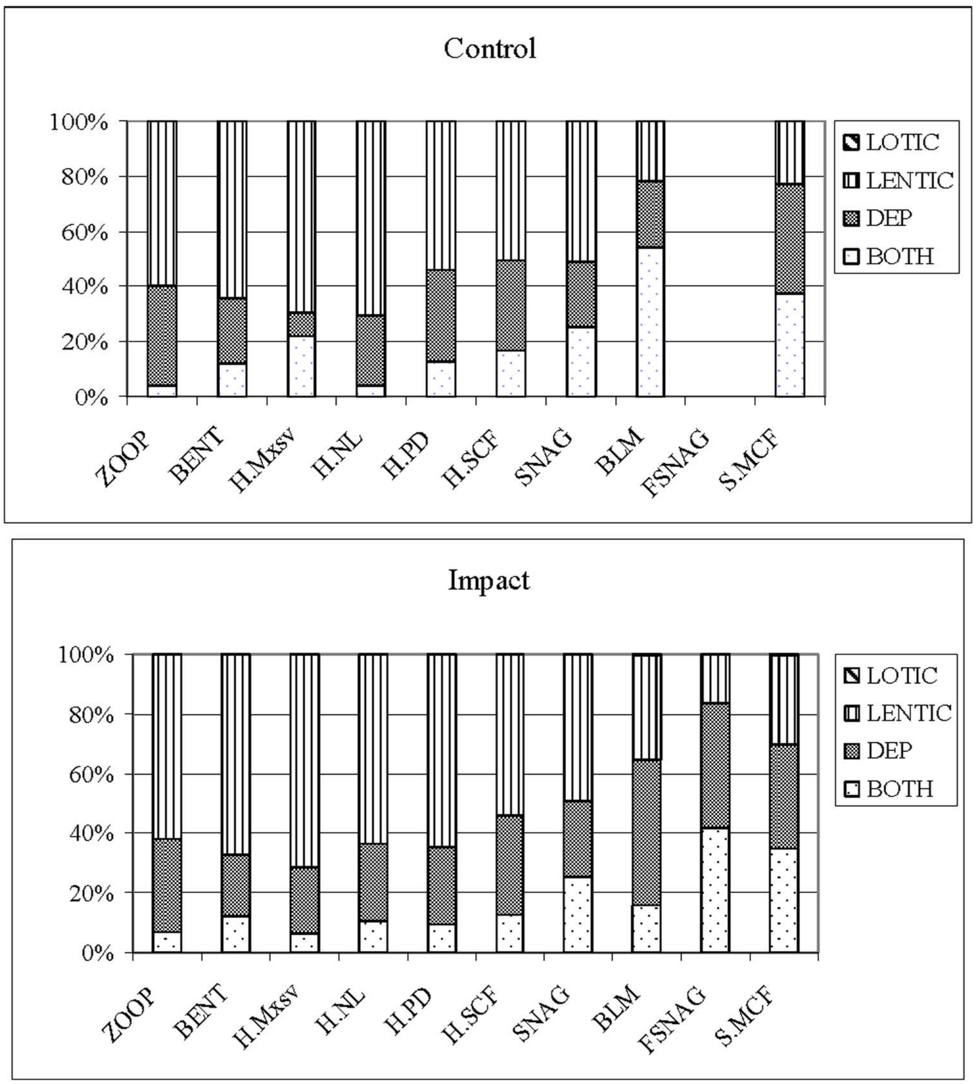


Figure 11-8. Mean annual functional habitat composition, based on total abundance, for each habitat at Control and Impact sites. ZOOP = Mid-channel Water Column, BENT = Mid-channel Benthic, H.MxSV = Miscellaneous Submerged Vegetation, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, SNAG = River Channel Woody Debris, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, S.MCF = Woody Shrub.

Low sample replication (three) and frequency of collection (quarterly) was necessitated by manpower constraints. Although data collected in this manner may not be optimal for addressing temporal or seasonal patterns of abundance or biomass, we believe it was sufficient for documenting structural characteristics of the invertebrate community that are likely to change as a result of restoration (e.g., shifts in functional feeding and functional habitat groups).

Aquatic Invertebrate Community Structure

Invertebrate density in remnant channels of the Kissimmee River is generally within the range reported for three unimpacted Coastal Plain blackwater rivers (Benke et al. 1984, Smock et al. 1985, Benke and Meyer 1988).

The highest estimates of mean density in the channelized Kissimmee were found in floating mats of H.SCF, which had densities nearly four times greater (130,000–230,000/m^2) than those reported for any habitat in Coastal Plain river systems. Floating H.SCF mats consist of a dense web of highly branched roots located just below the water surface. The roots accumulate large amount of fine particulate organic matter, and provide a highly heterogeneous habitat that supports large numbers of microcrustaceans, *Hyalella azteca*, and several chironomids.

Invertebrate taxa diversity was low in all habitats and rarely exceeded 2.0. These values are in the range for moderately polluted streams, with values <1 typical of heavy pollution (Wilhm 1972). Species richness also is low in the Kissimmee River; however, Warren and Hohlt (1996) found that richness and diversity in Pools A and C bracketed values for Fisheating Creek, a reference (i.e., minimally impacted) stream in the eastern Florida flatwoods region. Although data for Fisheating Creek are limited to one sampling period, and not sufficient to generalize about richness and diversity in undisturbed rivers of central and south Florida, biogeographical factors (peninsular effect, isolation from tropical source pools) may account for low species richness in the Kissimmee River and other lotic system of south Florida.

Core taxa based on density were heavily skewed toward microcrustaceans (Table 11-1 and 11-2). Forty percent of core taxa at the Control site, and 64% of core taxa at the Impact site were microcrustaceans. Although microcrustaceans are likely to be seasonally abundant in some habitats (e.g., BLM), restoration of flow likely will reduce density of many taxa in river channel habitats.

Previous studies in remnant channels of the Kissimmee River have characterized the invertebrate community as typical of standing water (Vannote 1971, Toth 1993, Warren and Hohlt 1996). Our functional habitat classification was developed to quantify this pattern, and showed near complete absence of taxa characteristic of flowing water, and a large proportion of taxa characteristic of lentic habitats.

Snag habitats within remnant channels of the Kissimmee River are dominated by gathering-collectors (primarily midges characteristic of lentic or depositional habitats), shredders (primarily *Glyptotendipes* spp. [Chironomidae]), and scrapers (primarily the amphipod *Hyalella azteca* and several gastropods). The filtering-collector guild is dominated by active filtering-collectors (primarily microcrustaceans); passive filtering-collectors accounted for <3% of total numbers on snags within remnant channels. Benke et al. (1984) report that passive filtering-collectors, including caddisflies (primarily *Hydropsyche* spp.) and blackflies (*Simulium* spp.), were the major consumers on snags in the Satilla River, Georgia, and accounted for 75–80% of mean annual density, 65–75% of mean annual biomass, and 72–79% of mean annual production at two sample locations. Smock et al. (1985) report passive filtering-collectors (primarily *Macronema carolina* [Hydropsychidae] and *Tanytarsus* sp. [Chironomidae]) were the dominant taxa on snags in Cedar Creek, South Carolina, and accounted for 28–39% of mean annual density, 25–65% of mean annual biomass, and 29–34% of mean annual production at two study sites. Benke and Meyer (1988) found that microfiltering-collectors and gathering-collectors strongly dominated invertebrate numbers on snags in the Ogeechee River, Georgia.

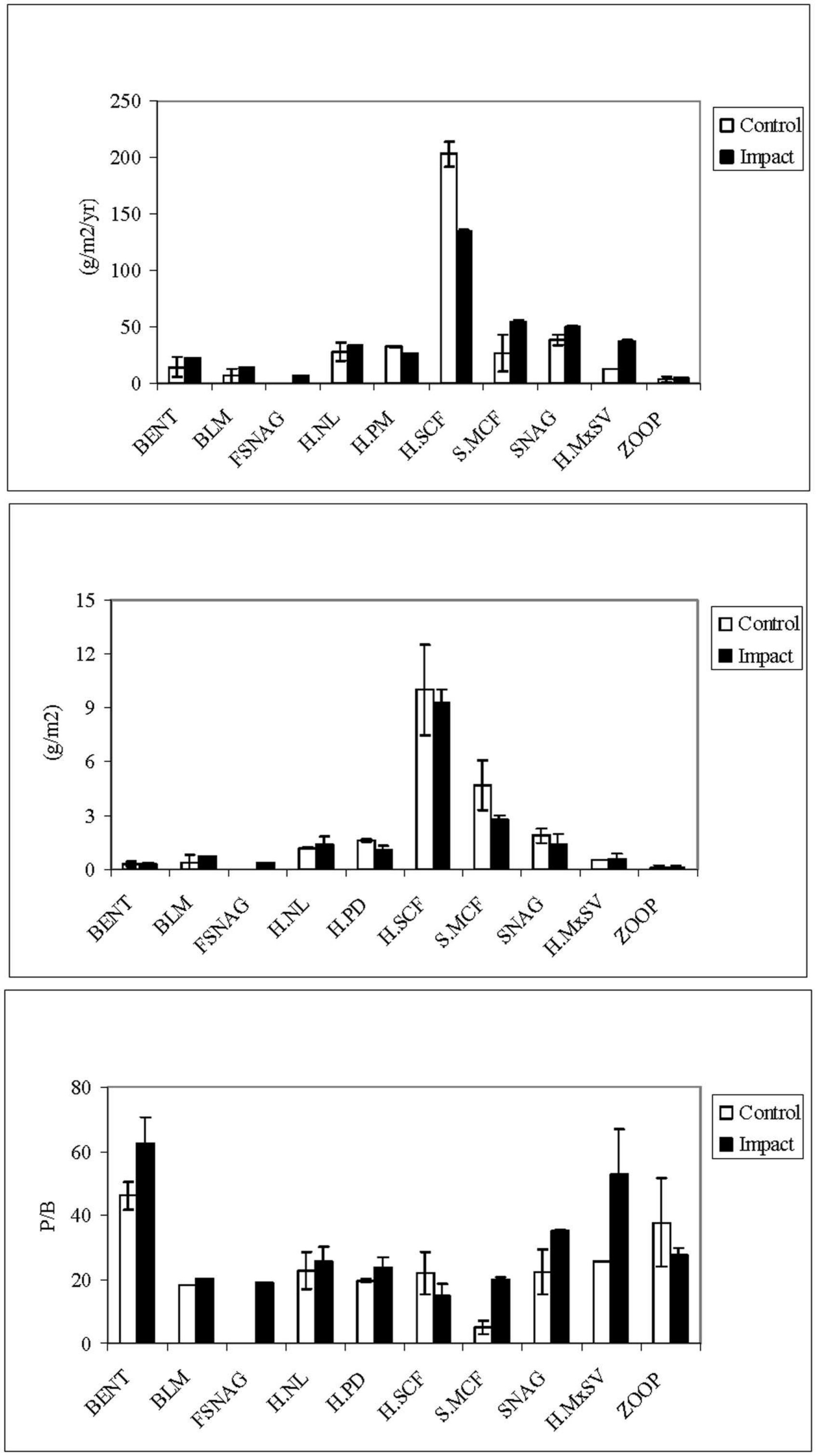


Figure 11-9. Mean annual production, biomass, and P/B ratio for all habitats at Control and Impact site. Estimates were obtained by averaging Year 1 + Year 2/2. BENT = Mid-channel, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, , S.MCF = Woody Shrub, SNAG = River Channel Woody Debris, H.MxSV = Miscellaneous Submerged Vegetation, ZOOP = Mid-channel Water Column.

Table 11-3. Annual production (mg m^{-2} yr^{-1}) at the Control site. BENT = Mid-channel, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, , S.MCF = Woody Shrub, SNAG = River Channel Woody Debris, H.MxSV = Miscellaneous Submerged Vegetation, ZOOP = Mid-channel Water Column.

Taxon	BENT	BLM	FSNG	H.NL	H.PD	H.SCF	S.MCF	SNAG	H.MxSV	ZOOP
Ablabesmyia	664									194
Acari		4940					2639		559	164
Bezzi/Palpomyia						9404	2141	1974		
Caecidotea							2450			
Caenis	1186				1615					370
Celina				3137						
Chaoborus	1421									
Chironomus	4655			2301	1812	12769			621	
Curculionidae										175
Cypria/Physocypria	2607			1547					1106	632
Dicrotendipes					1770	10397		3456	610	
Glyptotendipes				1937	7739	35254		13336	337	
Goeldichironomus						13930		2354		
Guttipelopia									325	
Helobdella									551	
Hyalella azteca					3200	24990		4746		
Mesocyclops										196
Oligochaete						12494	3374			
Polypedilum		462		7200		15514	2117			
Tipulidae							1243			
Other	3865	1151		12005	15982	68397	12376	12999	2341	1570
Total	14398	6553		28128	32119	203149	26340	38866	6450	3302

Channelization also altered benthic aquatic invertebrate community structure. Mid-channel benthic communities, while not highly diverse, are often composed of several dipteran, ephemeropteran, trichopteran, and molluscan species (Benke et al. 1984, Smock et al. 1985, Stites 1986, Stites and Benke 1989). Dominant species in the channelized Kissimmee include the microcrustacean group *Cypria/Physocypria*, several dipterans, and aquatic mites. Most of these taxa are common and widespread in lentic and lotic systems of the southeast United States, and are generally tolerant of organic pollution and low levels of dissolved oxygen.

Bivalves are probably more abundant in the Kissimmee River than indicated by our samples, and may require more attention in future studies because of national concern about declines in the biodiversity of this group, and because the exotic Asian clam (*Corbicula fluminea*) invaded the Kissimmee during channelization. Prior to channelization, a survey of freshwater mussels (Family Unionidae) in peninsular Florida, including two sampling sites in the Kissimmee River, identified seven species as occurring in the Kissimmee/Everglades drainage basin (Johnson 1972). Only one of these, *Elliptio buckleyi*, was collected in the Kissimmee River. After channelization, Vannote (1971) collected *P. buckleyi*, another unionid *Anodonta couperiana*, and *Corbicula*. We occasionally made qualitative collections of *P. buckleyi*, *A. couperiana*, and possibly a third unionid, *A. imbecilis*, as well as *Corbicula*.

It is difficult to predict how this group of benthic filtering-collectors will respond to restoration, but some insight may be gained from considering data collected during the Demonstration Project (Toth 1991). *Corbicula* populations increased at several river locations with reestablished flow approximately one year after construction of the demonstration project weirs, and attained a maximum density of 2757 m^{-2} at one location. When this location was sampled again in August, after three months of low or no flow, density had decreased to 9 m^{-2}. Similar declines were observed at the other river channel locations. Although density of *Corbicula* may increase within restored river channels, it is not expected to displace any native bivalves or play a major role in the trophic dynamics of the restored system.

Table 11-4. Annual production (mg m^{-2} yr^{-1}) at the Impact site. BENT = Mid-channel, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, , S.MCF = Woody Shrub, SNAG = River Channel Woody Debris, H.MxSV = Miscellaneous Submerged Vegetation, ZOOP = Mid-channel Water Column.

Taxon	BENT	BLM	FSNG	H.NL	H.PD	H.SCF	S.MCF	SNAG	H.MxSV	ZOOP
Acari		326	674				8730			221
Belostoma			172							
Bezzia/Palpomyia	1062						8339			
Caenis	1410			1267						
Chaoborus	9465									
Chironomus	6236			3800	2411				1857	
Collembola			198							
Cypria/Physocypria				3369	1878	6517			1910	1987
Dicrotendipes				4223	2238	15787		4520	3789	
Erythemis								4596		
Fittauimyia		1083	346							
Glyptotendipes					2549	15551		22883		
Hyalella azteca				1677	3359	7796		3731	2447	
Laccophilus		444								
Microtendipes	1724			3061	1286					
Natarsia		314								
Oligochaete			1198				3895			
Polypedilum						7292		2678		
Procambarus		625							14217	
Scirtes							3037			
Tipulidae							6437			
Other	2290	4100	709	15907	12996	81750	24741	11401	13210	1985
Total	22187	6894	3297	33304	26716	134695	55179	49910	37430	4193

Secondary Production

Our estimates of biomass and secondary production rely on estimates of individual mass and growth rates from regression equations developed in other systems, with minimal replication over a broad temporal scale. We expect some error to be associated with the cross-organism and cross-system use of these equations, and from the fact that our estimates of biomass were obtained from a few replicates. However, this error will be applied systematically across habitats at Control and Impact sites, which will allow us to make inferences about changes between sites, and between the baseline and post-construction periods (Benke et al. 1998). Also, we reduce the influence of errors for individual taxa by emphasizing estimates for communities in each habitat, and for guilds such as functional feeding groups (Morin and Dumont 1994).

Annual production in all habitats at Control and Impact sites was dominated by taxa atypical of relatively undisturbed rivers of the southeastern Coastal Plain. Core taxa, based on percent of total production, were dominated by lentic and depositional chironomids, and several larger lentic taxa, including *Hyalella azteca* and coleopterans.

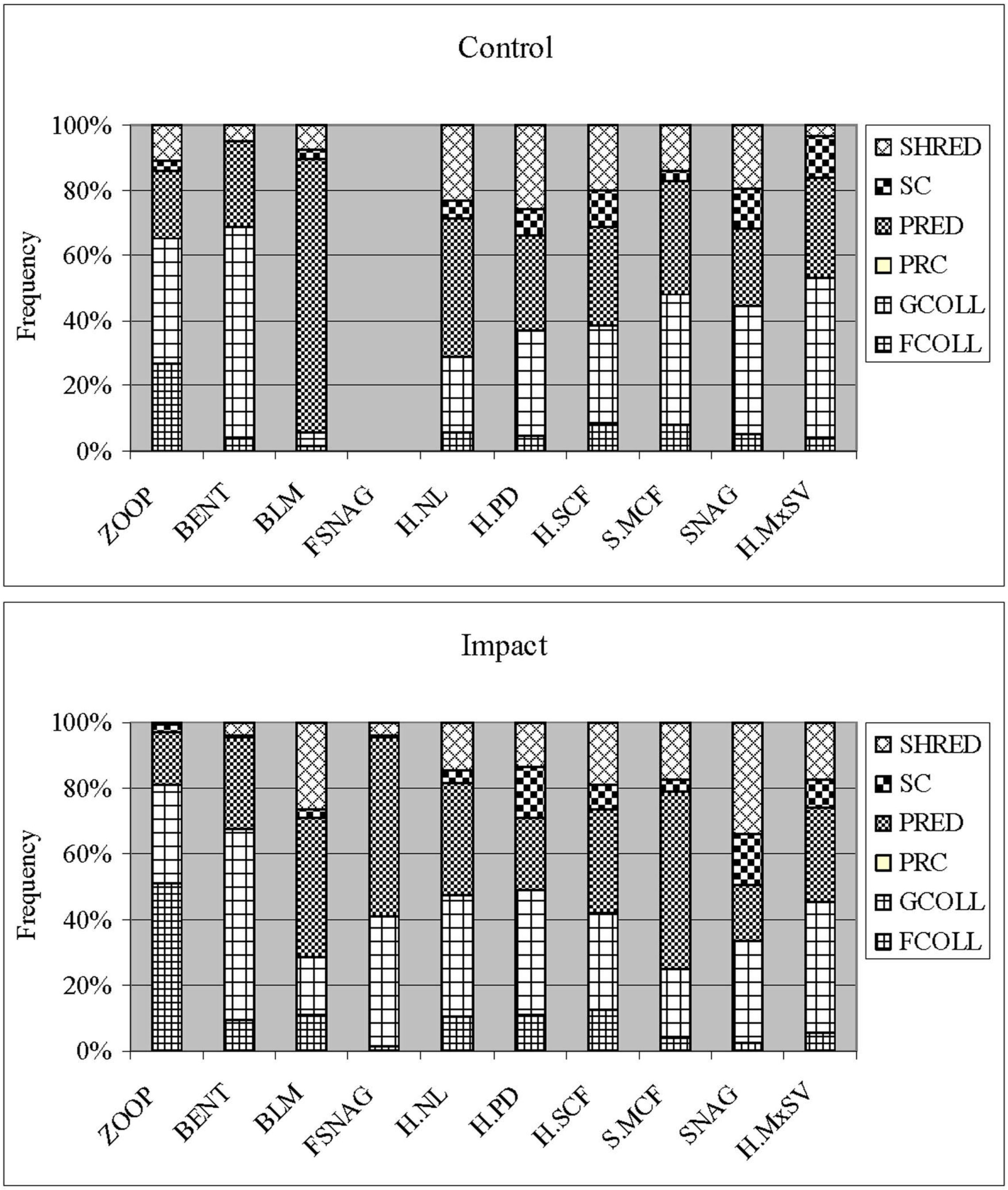
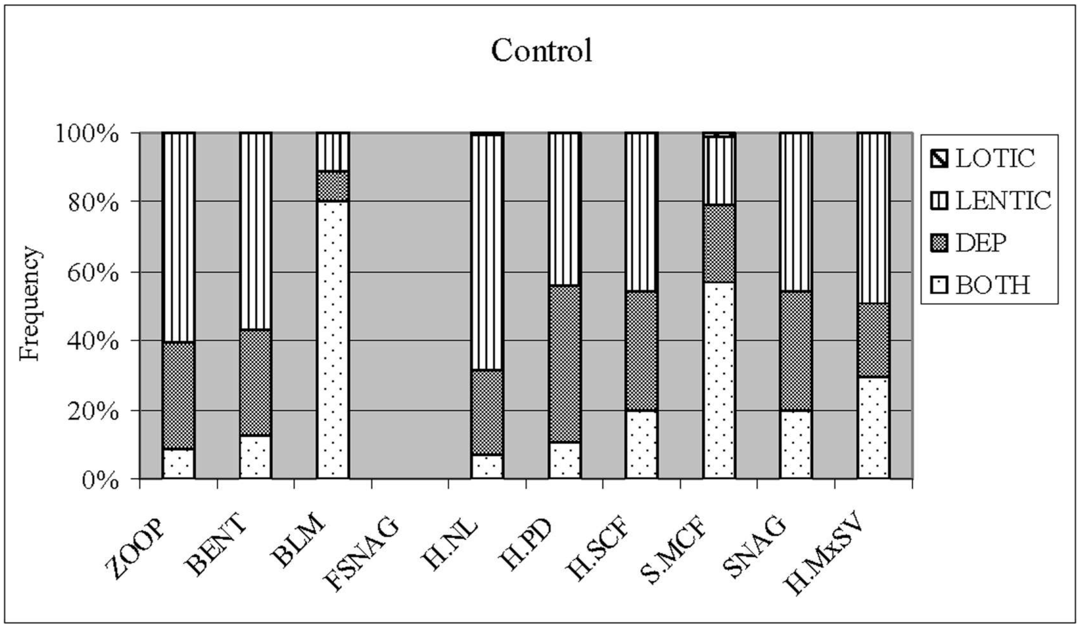


Figure 11-10. Distribution of total production among functional feeding groups at Control and Impact sites. ZOOP = Mid-channel Water Column, BENT = Mid-channel Benthic, H.MxSV = Miscellaneous Submerged Vegetation, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, SNAG = River Channel Woody Debris, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, S.MCF = Woody Shrub.



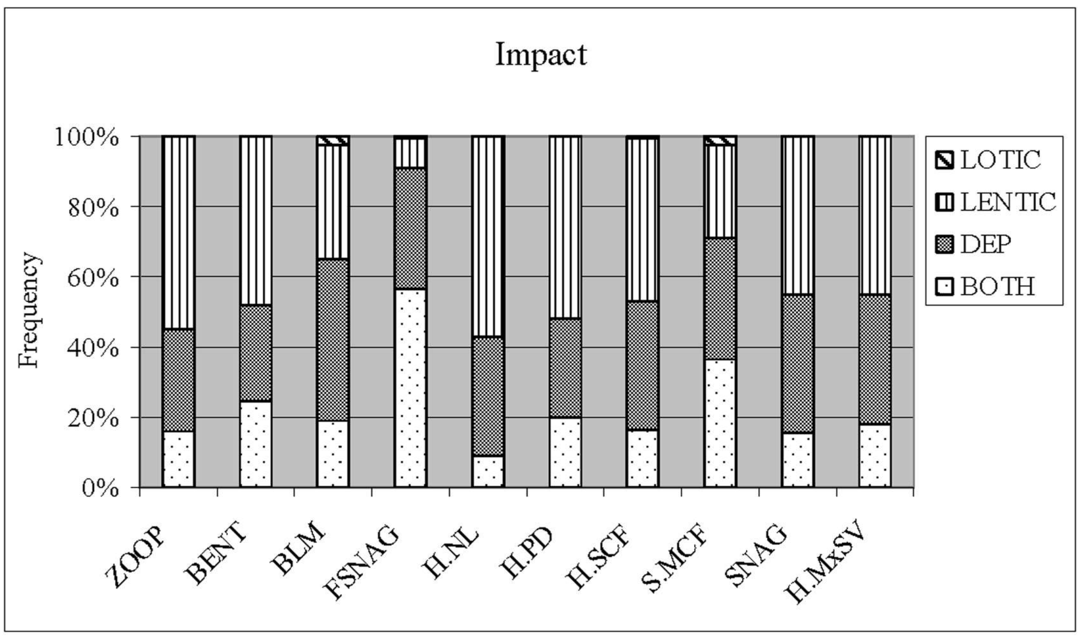


Figure 11-11. Distribution of total production among functional habitat groups at Control and Impact sites. ZOOP = Mid-channel Water Column, BENT = Mid-channel Benthic, H.MxSV = Miscellaneous Submerged Vegetation, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, SNAG = River Channel Woody Debris, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, S.MCF = Woody Shrub.

Community production estimates for 40 streams around the world range from 0.6 g m^{-2} to 612 g m^{-2}, but estimates >70 g m^{-2} occur at organically enriched sites, downstream of impoundments, or in warm desert streams (Benke 1993). By averaging baseline production estimates across habitats, we obtain a value of 40 g m^{-2} at the Control site and 37 g m^{-2} at the Impact site. Our estimates are much larger than the 3 g m^{-2} for Cedar Creek, South Carolina, (Smock et al. 1985), 2.4 g m^{-2} and 6.1 g m^{-2}, respectively in

Buzzards Branch and Colliers Creek, Virginia (Smock et al. 1992), which occur in the Coastal Plain but are smaller than the Kissimmee River. We estimated production of 14 and 22 g m^{-2} in the benthos at Control and Impact sites, respectively, while Benke et al. (1984) reported 21 g m^{-2} in sandy benthos and 18 g m^{-2} in mud benthos for the Satilla River. We estimated production of 39 g m^{-2} and 50 g m^{-2} on snags at Control and Impact sites, respectively, while production was 65 g m^{-2} on snags in the Satilla River. Because estimates of secondary production within Kissimmee River channel habitats is within the range of values reported for similar habitats in unmodified Coastal Plain rivers, post-construction estimates of secondary production within these habitats likely will not provide a useful measure of restoration success. However, changes in the distribution of production among functional feeding and functional habitat groups can be used as indicators of restored hydrology and restoration success.

River floodplains are typically highly productive environments that support abundant fish and wildlife resources. Most studies of floodplain macroinvertebrate production have occurred in systems much smaller than the Kissimmee or have focused on a small number of species rather than whole communities (Smock et al. 1985, Gladdon and Smock 1990, Smock et al. 1992, Duffy and LaBar 1994, Pickard and Benke 1996), making comparisons between these studies and our baseline data difficult. Estimates of total secondary production for floodplain macroinvertebrate communities within the channelized system are very low (6.0 and 6.4 g m^{-2} yr^{-1}, respectively for Pool A and C), and are within the range of values reported for single species and small groups of aquatic invertebrates.

Aquatic Invertebrate Drift

Aquatic invertebrate drift is a key functional attribute of flowing water systems. Drift can be an effective way for some aquatic organisms to colonize new areas (Cellot 1989), and can play an important role in energy transfer to higher trophic levels (Benke et al. 1985, Rader 1997). Aquatic organisms can enter the water column in a number of ways, including behavioral (i.e., periodic, for example, to escape from a predator), constant (i.e., background drift due to accidental dislodgement), and catastrophic (i.e., as a result of some major adverse event) drift mechanisms (Waters 1972). In the channelized (non-flowing) Kissimmee River, aquatic macroinvertebrates are rare in the drift. Those that do occur likely enter the water column through active swimming or rafting on floating vegetation (e.g., *Pistia stratiotes*).

Because the channelized Kissimmee River functions more like a lake than a river, and supports an aquatic invertebrate community more typical of a lentic system, drift composition in the channelized Kissimmee River is very different from free-flowing southeastern Coastal Plain blackwater rivers (Benke et al. 1986, 1991). In these systems, larval Coleoptera, Diptera, Ephemeroptera, and Trichoptera are the major contributors to drift numbers and biomass. Microcrustaceans generally account for a small proportion of drifting organisms (Table 11-5).

Reestablishment of an aquatic macroinvertebrate community typical of unmodified southeastern Coastal Plain rivers is a prerequisite for reestablishing invertebrate drift composition typically found in southeastern blackwater rivers. Restoration of continuous flow and in-channel habitat structure will be the impetus for macroinvertebrate (including Coleoptera, Ephemeroptera, and Trichoptera) colonization of restored habitats. Colonization by most river channel macroinvertebrate taxa likely to be found in the drift will occur through adult oviposition. As aquatic invertebrate community structure is restored, seasonal variable flow patterns are expected to result in a shift in macroinvertebrate drift composition from microcrustaceans to one more typical of unmodified Coastal Plain rivers (i.e., macroinvertebrates).

Reference Conditions, Comparisons, & Expectations

Introduction

Channelization of the Kissimmee River likely impacted aquatic invertebrate community structure, functional feeding group associations, productivity, and drift dynamics. Community structure and functional organization on snags and benthic habitats are very different from those of reference sites.

Table 11-5. Major invertebrate groups found in the drift of the Satilla and Ogeechee Rivers, Georgia (Benke et al. 1986, 1991) and Pool C of the channelized Kissimmee River. There was no significant difference between invertebrate drift numbers or biomass between Pools A and C; therefore, only Pool C data is presented. Numbers indicate frequency of occurrence.

	Satilla River		Ogeechee River		Kissimmee River (Pool C)	
Taxonomic Group	Density	Biomass	Density	Biomass	Density	Biomass
Diptera	52.9	53.8	27.3	10.6	< 1	11.2
Coleoptera	11.3	21.5	6.2	27.4	< 1	2.5
Ephemeroptera	5.8	6.2	15.4	34.6	< 1	7.4
Trichoptera	18.6	13.8	11.5	20.2	--	--
Odonata	1.4	4.6	1	5.3	<1	2.4
Crustacea*	10	< 1	31.9	1.9	96.8	54.6
Miscellaneous	--	--	6.7	--	2.7**	21.9**

* Includes macro- and microcrustaceans.
** Includes Hemiptera, Trichoptera, Megaloptera, Lepidoptera, Collembola, Gastropoda, Oligochaeta, and Nematoda

Aquatic invertebrate species richness and diversity in remnant Broadleaf Marsh are likely lower than pre-channelization marshes, and aquatic invertebrate drift is dominated by zooplankton rather than macroinvertebrates. To determine success of the Kissimmee River restoration project, specific comparisons must be made between reference and baseline conditions and between baseline and post-construction conditions. Comparisons between the reference and baseline conditions estimate whether the system has changed as a result of channelization, and to what extent, and provide clues as to what the pre-channelization condition may have been and what the restored condition might be. Following restoration, comparisons between the baseline and post-construction conditions will reveal if the system has responded to restoration efforts, and whether the response is in the expected direction and magnitude. The following sections describe development of reference conditions for habitat-specific aquatic invertebrate communities, compare reference conditions with baseline conditions, and predict how communities are expected to respond to restoration through development of specific habitat-based expectations for restoration.

River Channel Aquatic Invertebrate Community Structure and Production

Pre-channelization data from the lower Kissimmee River basin would provide the best reference conditions for assessing aquatic invertebrate responses to Kissimmee River restoration. However, an extensive literature search found no information on aquatic invertebrate community structure or functional characteristics in the pre-channelized Kissimmee River.

Large Woody Debris

Methods. In order to develop quantitative predictions of aquatic invertebrate responses to Kissimmee River restoration, published studies of invertebrate communities in other southeastern, blackwater Coastal Plain river/floodplain systems were reviewed. Based on this review, data from two Coastal Plain river systems, the Satilla and Ogeechee rivers in Georgia, were selected as appropriate reference sites for developing expectations for restoration of density, biomass, and production of passive filtering-collectors on large woody debris and aquatic invertebrate community structure in sand habitats.

The Satilla River provides the primary source of information on functional feeding group composition, density, biomass, and annual production of aquatic invertebrates on large woody debris within the pre-channelized Kissimmee River (Benke et al. 1984). The Satilla River is a sixth-order, blackwater southeastern Coastal Plain river characterized by a very low gradient, low pH, high organic carbon, and high color (Benke et al. 1986).

In order to quantify aquatic invertebrate community structure on large woody debris in the Satilla River, Benke et al. (1984) sampled snags from two locations for one year. Six samples per site were collected every two weeks from May through August, and monthly for the remainder of the year. Invertebrates were identified and measured. Invertebrate density and standing stock biomass were converted to amount per square meter of habitat surface for each snag sample. Production was estimated using the size-frequency method.

Results. Within the Satilla River, passive filtering-collectors accounted for 75–80% of total numbers, 65–75% of total biomass, and 72–79% of total production at two sample locations (Benke et al. 1984).

Discussion and Comparison with Baseline Condition. Filtering-collectors were selected as an indicator guild because they often account for the largest proportion of mean annual density, standing stock biomass, and production on snags in southeastern river systems. Because most passive filtering-collectors are sedentary and utilize various sieving mechanisms for removing particulate matter from suspension, continuous flows are necessary to transport fine particulate organic matter that can be captured and used as a food source. Additionally, many filtering-collectors respond predictably (decrease) to increased perturbation (e.g., no flow, low dissolved oxygen) (Lenat 1988, Lamberti and Berg 1995, Barbour et al. 1996). Channelization of the Kissimmee River eliminated flow through remnant river channels, reduced levels of dissolved oxygen within the water column (Colangelo 2005), and likely altered density, biomass, and production of passive filtering-collector guild on large woody debris.

Passive filtering-collector taxa are rare on large woody debris in the channelized Kissimmee River, accounting for <2% of mean annual density, <3% of mean annual biomass, and <1% of mean annual production in Pool A, and <1% of mean annual density, <2% of mean annual biomass, and <1% of mean annual production in Pool C. This is very different from the Satilla River, where passive filtering-collectors account for the greatest proportion of these metrics at two sample locations (Benke et al. 1984) (Figure 11-12). Although the Satilla River is the sole reference site for pre-channelization community structure and production on river channel snags, other studies (Thorp et al. 1985, Smock et al. 1985, Benke and Meyer 1988) support the fact that the passive filtering-collectors often make up the largest proportion of density, biomass, and production within this habitat.

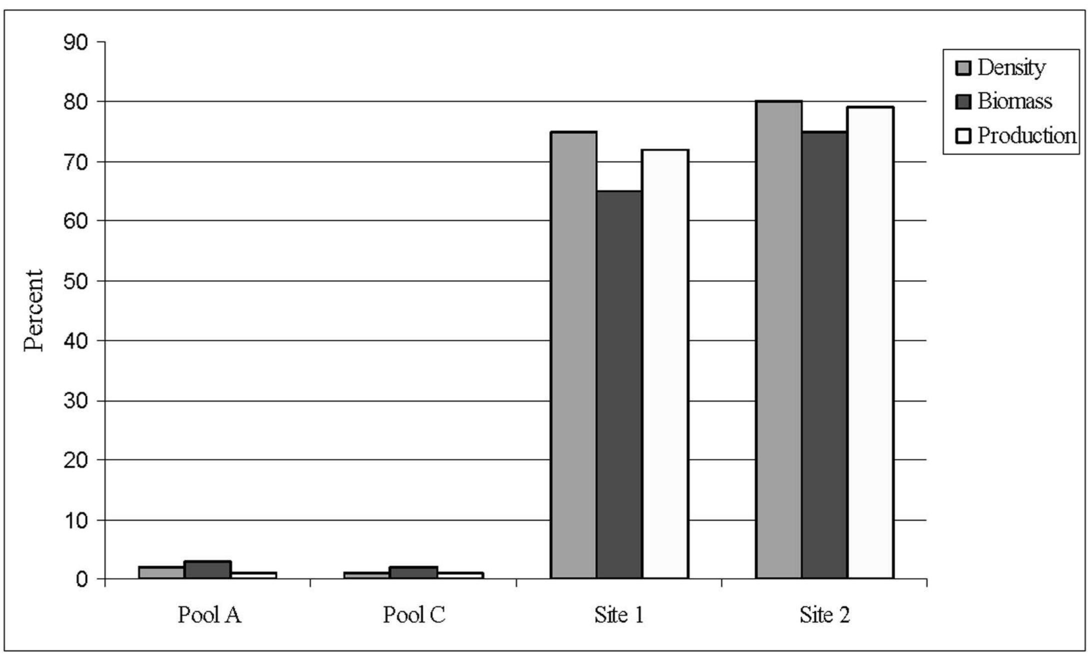


Figure 11-12. Mean annual density, biomass, and production of passive filtering-collectors on snags in the Kissimmee River (Pools A and C), and Satilla River, Georgia (Sites 1 and 2) (Benke et al. 1984).

Based on a comparison of baseline and reference data for mean annual density, biomass, and production of snag-dwelling passive filtering-collectors, restoration of physical and chemical habitat structure within the Kissimmee River likely will result in shifts in functional feeding group composition on snags within the restored river. The following expectation has been developed from baseline data and best available reference data.

<u>Expectation:</u> Increased relative density, biomass, and production of passive filtering-collectors on river channel snags.

Passive filtering-collectors are expected to respond quickly to restored flow and increases in levels of dissolved oxygen within the river channel, and account for the greatest proportion of mean annual density, mean annual biomass, and mean annual production on large woody debris in restored river channels (Koebel 2005a). However, because passive filtering-collector macroinvertebrates are rare in the channelized system, the time frame for redistribution of density, biomass, and production among functional feeding groups is primarily dependent on colonization by filtering-collectors and displacement of existing dominant functional feeding groups, which will depend on distance colonists must travel. It is expected that small and large-bodied filtering-collectors, primarily chironomids, simuliids, and caddisflies will immigrate from lotic systems within the Kissimmee basin (e.g., Fisheating Creek, Tiger Creek, Cypress Creek, Weohykapka Creek) and likely colonize within six to nine months. The potential for high standing stock biomass of several filtering-collectors (primarily caddisflies), and rapid biomass turnover rates for others (Simuliidae and Chironomidae), likely will result in the greatest proportion of mean annual density, mean annual biomass, and mean annual production being attributed to passive filtering-collectors.

Sampling of snags will commence approximately six months following initiation of the revised headwaters regulation schedule and reestablishment of continuous flow. Snag-dwelling macroinvertebrate density, biomass, and production will be analyzed for a minimum of three years following reestablished flow. Post-construction sampling will include collection of monthly, replicate (five) snag samples from randomly selected locations within reconnected channels in Pool C and remnant channels in Pool A. Samples will be analyzed for invertebrate species identity, functional feeding group composition, density, and standing stock biomass. Passive filtering-collectors will be identified according to Merritt and Cummins (1996). Production will be calculated using the instantaneous growth rate (IGR) method. Growth equations for major taxa will be determined experimentally or obtained from scientific literature. Monthly means will be averaged annually to determine mean monthly density and biomass for the filtering-collector guild. The three annual estimates of mean monthly density and biomass will be averaged to obtain a mean annual value. The three estimates of annual production also will be averaged to determine mean annual production. Although values for these metrics may vary from year to year, a multi-year, multi-metric evaluation of changes in macroinvertebrate functional composition and production on snags will provide an objective measure of restoration-related changes that integrate potential intra- and inter-annual variability.

Sand Substrates

<u>Methods.</u> The primary source of information on sand-dwelling macroinvertebrates within the pre-channelized Kissimmee River is derived from published data on community composition in the Ogeechee and Satilla Rivers, Georgia (Benke et al. 1984, Stites 1986). The Ogeechee River, a sixth-order, blackwater river in the lower Coastal Plain of Georgia, is characterized as low gradient (0.02%), with a high level of dissolved organic carbon, mean annual discharge of 66.8 m^3 s^{-1} (44 year period of record), mean annual water temperature ranging from 3–32°C (Stites 1986), and a river channel bottom consisting of 80–90% sand (Stites and Benke 1989). Detailed sampling methods for sand-dwelling macroinvertebrates can be found in Benke et al. (1984) and Stites (1986). Additional information was derived from published reports on the geographic distribution of sand-dwelling macroinvertebrates throughout central Florida (Dunkle 1989, Toth 1991, Epler 1992, Merritt et al. 1996, Berner and Pescador 1988)

<u>Results.</u> The sand-dwelling aquatic invertebrate community of the Ogeechee and Satilla Rivers are quite similar. Dominant macroinvertebrates included the dipterans *Corynoneura* sp., *Cladotanytarsus* sp., *Cryptochironomus* sp., *Parakiefferiella* sp., and *Robackia* sp., Certatopogonidae, and oligochaetes. Other

dominant taxa in the Ogeechee included *Lopescladius* sp., *Rheosmittia* sp., and *Corbicula fluminea* (Stites 1986). Additional dominant taxa in the Satilla River included *Polypedilum* sp., *Tanytarsus* sp., and *Thienemanniella* sp.

Based on habitat preferences and geographic distributions throughout Florida, other taxa likely to be present among the sandy benthos of the restored Kissimmee include Ephemeroptera, including *Stenonema* sp. and *Cercobrachys* sp. (Berner and Pescador 1998); mollusks, including *Musculium/Pisidium* complex (Toth 1991); odonates, including *Dromogomphus spinosus*, *Gomphus minutus*, *Gomphus dilatatus*, and *Stylurus plagiatus* (Dunkle, 1989); and Trichoptera, including *Oecetis* sp. and *Setodes* sp. (Merritt et al. 1996).

Discussion and Comparison with Baseline Condition. Most of the historic sand substrate within mid-channel habitats of remnant river channels is covered with large accumulations of organic matter, primarily derived from dead and decaying aquatic vegetation. The associated aquatic invertebrate community consists of taxa most often associated with organically enriched environments, and are generally tolerant of low levels of dissolved oxygen. Restoration of flow is expected to flush organic deposits, or redistribute existing sand to cover these deposits and form sand bars along the inside margins of meanders. Restoration of flow and reestablishment of a sand substrate is likely to result in increased levels of dissolved oxygen within restored channels by reducing microbial sediment oxygen demand (Colangelo 2005). These shifts in physical and chemical habitat structure are likely to induce changes in aquatic invertebrate community structure within mid- and marginal channel sand habitats.

Because of the lack of historical data, the Ogeechee and Satilla Rivers provide reasonable reference conditions for aquatic invertebrate community structure in sand habitats of the pre-channelized Kissimmee River. Although reference conditions are solely derived from these two systems, other studies (Whitman and Clark 1984, Strommer and Smock 1989) indicate that many of the same taxa dominate sand substrates in other lotic systems of the southern United States (Virginia and Texas). Most taxa occurring in sand habitats of the Ogeechee and Satilla Rivers are considered characteristic, or obligate sand-dwellers (Whitman and Clark 1984). These characteristic taxa are absent or rare in benthic habitats of the channelized Kissimmee River; however, most occur within the lower Kissimmee basin or adjacent watersheds, and many are likely to quickly colonize restored sand substrates (Table 11-6).

Based on a comparison of baseline and reference data for macroinvertebrate community composition in sand habitats, restoration of physical habitat structure (sand habitat) within the Kissimmee River likely will result in colonization of invertebrate taxa considered characteristic of sand habitats. The following expectation has been developed from baseline data and best available reference data.

Expectation: Aquatic invertebrate community structure in river channel benthic habitats.

The macroinvertebrate fauna of river channel benthic habitats will primarily consist of taxa that are common and characteristic of sand substrates (Koebel 2005b).

The expectation for shifts in aquatic invertebrate community structure in sand habitats of the restored Kissimmee River is less rigorously defined; however, sand substrates of many southeastern Coastal Plain rivers support a characteristic and consistent group of aquatic invertebrate taxa. Because many of these taxa appear to be habitat specialists, it is not unreasonable to expect that many of these taxa will colonize sand substrates in the restored Kissimmee River. It is unlikely that all taxa will be present in restored habitats; however, representative taxa (Table 11-6) are expected to show substantive change relative to the baseline condition and therefore be reasonable indicators of habitat restoration.

Sampling of sand habitats will commence approximately six months following initiation of the revised headwaters regulation schedule and reestablishment of continuous flow. Sand-dwelling macroinvertebrates will be collected for a minimum of three years following reestablished flow. Post-construction sampling will include collection of monthly, replicate (five) mid-channel sand samples and five marginal channel sand samples from randomly selected locations within reconnected channels in Pool C. For comparison, mid-channel benthic samples also will be collected in remnant channels in Pool A. Samples will be analyzed for invertebrate species identity. Community composition will be compared to the baseline condition and stated expectation.

Floodplain Macroinvertebrate Community Structure

Methods. A thorough literature search found no information on aquatic invertebrate community structure characteristics of pre-channelization Broadleaf Marshes of the Kissimmee River, or marshes that were structurally similar to pre-channelization marshes. Therefore, in the absence of historical data or suitable reference sites, baseline data collected in remnant, but altered BLM in Pool C, was used to predict a minimal response by aquatic invertebrates to restored hydroperiod and habitat structure.

An attempt was made to collect quarterly, replicate (three) aquatic invertebrate samples from remnant BLM in Pools A and C between August 1995 and May 1997. Each quarter, when water was present on the floodplain, replicate stovepipe (area = 1662 cm^2) samples were collected from randomly selected locations in BLM. Sample locations were determined by traveling a randomly determined distance ($\leq$400 m) and direction (0–360°) from a randomly determined starting point within BLM. Following trap placement, water depth within the trap was recorded and all vegetation was removed. A dip-net equipped with a 118 µm mesh net was used to remove invertebrates. A total of ten "dips" constituted a sample. All invertebrates were identified. Species richness and species diversity were calculated for each replicate on each date. Because pasture habitat in Pools A and C was dry during most of the baseline period, aquatic invertebrates were not quantified in this habitat.

Results. Broadleaf Marsh habitat in Pools A and C was dry during much of the study period. Pool A was sampled only once, and Pool C was sampled only three times. In Pool A, species richness was 21 and species diversity was 0.84. In Pool C, species richness ranged from 15 to 32 (total species richness = 65) and species diversity ranged from 1.86 to 2.75 (mean diversity = 2.37). Species richness and diversity in pasture habitat was assumed to be 0 and 0.00, respectively.

Discussion and Comparisons with Baseline Condition. Documented studies on aquatic invertebrate community structure of subtropical wetland systems are limited (Rader 1994, Evans et al. 1999, Rader 1999), and have focused on systems that are structurally different from pre-channelization Broadleaf Marshes of the Kissimmee River floodplain (i.e., Water Conservation Areas and flatwoods marshes). Rader (1994) found 174 taxa comprise the known aquatic invertebrate community in the Everglades, but indicates that the actual number of taxa may be as great as 250. Diversity estimates for benthic macroinvertebrates in natural flatwoods marshes of central Florida range from 3.94 to 4.50, with a mean of 4.23 (Evans et al. 1999). Although vegetation communities of the Everglades and flatwoods marshes are structurally different from pre-channelized marshes of the Kissimmee River, it is likely that the aquatic invertebrate community of restored Broadleaf Marshes will be species rich and diverse. Although these studies provide insight into the potential for high species richness and diversity within restored or natural marshes of central Florida, they can not be used to predict species richness and diversity in restored BLM. However, assuming that a restored BLM will support an aquatic invertebrate community with at least the same species richness and diversity as remnant marshes, baseline data from Pool C can provide a conservative estimate of species richness and diversity in restored BLM.

Expectation: Aquatic invertebrate community structure in Broadleaf Marsh.

Aquatic invertebrate species richness and species diversity will be $\geq$65 and $\geq$2.37, respectively in restored Broadleaf Marsh (currently pasture in the channelized system) (Koebel 2005c).

Unpredictable hydroperiods and homogeneous vegetation communities in remnant Broadleaf Marsh likely limit aquatic invertebrate species richness and diversity. Although data on pre-channelization species richness and diversity of floodplain wetlands do not exist for the pre-channelized Kissimmee, reestablishing long-term hydroperiods and associated development of a diverse, heterogeneous wetland plant community likely will allow for development and persistence of a diverse macroinvertebrate community.

Initial sampling of existing Broadleaf Marsh and future Broadleaf Marsh (existing pasture) will coincide with sampling of large-bodied fish and wading bird use of floodplain habitats (i.e., approximately one year after initiating the revised headwaters regulation schedule). Although this time frame is not sufficient to reestablish historic aquatic invertebrate community structure characteristics, these data may be useful for interpreting the initial response and distribution of large-bodied fishes and wading birds within floodplain habitats. Methods will be identical to those outlined in Anderson et al. (1998b), and include monthly, replicate (five) stovepipe (area = 0.105 m^2) or throwtrap (area = 0.25 m^2) samples from randomly selected locations within Pools A and C. Additional focus will be on density and biomass of "keystone" taxa (e.g., crayfish, grass shrimp, dragonflies, and snails) likely to serve as high quality prey items for higher trophic levels (e.g., wading birds and fishes). Sampling for these taxa will correspond with floodplain fish sampling and consist of monthly, replicate (ten) throwtrap (1 m^2) samples from existing BLM and pasture habitats undergoing transition to BLM in Pool C and remnant BLM and improved pasture in Pool A. Sampling will continue for at least three years.

Aquatic Invertebrate Drift

Methods. An extensive literature search found no information on aquatic invertebrate drift in the pre-channelized Kissimmee River. In order to develop quantitative predictions of aquatic invertebrate responses to Kissimmee River restoration, published studies of aquatic macroinvertebrate drift in other southeastern, blackwater Coastal Plain river/floodplain systems were reviewed. Based on this review, data from two southeastern Coastal Plain rivers were selected as appropriate reference sites for developing an expectation for restoration of aquatic macroinvertebrate drift composition in the restored Kissimmee River.

Reference conditions have been developed based on macroinvertebrate drift data from the Satilla and Ogeechee Rivers, Georgia (Benke et al. 1986, 1991). In order to characterize macroinvertebrate drift density and biomass in the Satilla River, Benke et al. (1984) collected samples from the water column using two nets (mesh = 400 µm, net opening = 0.135 m^2). One net was positioned 10–50 cm above the sand bottom, while the second net was placed just below the water surface. Current velocity was measured at each net in order to determine the volume of each sample. Samples were collected at two to four-week intervals just after dark for a period of one year. All organisms were identified and measured. Numbers per volume of water were converted to biomass per volume of water using taxon-specific length-mass relationships. Ogeechee River drift was characterized by Benke et al. (1991) in a similar manner, although mesh size (234 µm) and net opening (89.4 cm^2) differed between studies, and the Ogeechee River study was conducted for two years.

Results. These studies indicate larval Coleoptera, Diptera, Ephemeroptera, and Trichoptera are the major contributors to drift numbers and biomass in these three systems (Table 11-5).

Discussion and Comparison with Baseline Conditions. Because the channelized Kissimmee River is characterized by no flow, aquatic invertebrate drift is primarily due to active swimming or rafting on floating aquatic vegetation. Drift composition within the channelized Kissimmee consists primarily of zooplankton which is very different from the Satilla River. Drift community structure from the Satilla and Ogeechee Rivers provide reasonable reference conditions for macroinvertebrate drift in the restored Kissimmee. Reestablished continuous flow and restoration of habitat structure will be the impetus for changes in aquatic invertebrate community structure, as well as the subsequent shift in invertebrate drift density and biomass from dominance by zooplankton to dominance by macroinvertebrates. The following expectation has been developed from baseline data and the best available reference conditions.

Expectation: River channel Macroinvertebrate drift composition.

Macroinvertebrate drift composition will be dominated by Coleoptera, Diptera, Ephemeroptera, and Trichoptera (Koebel 2005d).

Invertebrate drift will be sampled monthly beginning two years after implementation of the revised headwaters regulation schedule, assuming that this time period is sufficient to reestablish river channel invertebrate communities typical of unmodified southern Coastal Plain rivers. Drift will be quantified monthly from two sites (upper and lower) in Micco Bluff Run. Paired drift nets (net opening = 900 cm2, mesh size = 125 µm), facing into the flow, will be placed at the water surface and 0.5 m above the channel substrate. Samples will be collected for a period of four hours beginning one-half hour after dusk. Flow will be measured at each net opening when nets are set or retrieved. All invertebrates will be identified to Order (minimally), and an appropriate length measurement will be taken to determine length-mass relationships. Numbers and biomass per volume of water will be calculated for each taxonomic group. Sampling will occur for a minimum of two years. Post-construction data will be compared to baseline data and the expectation in order to determine changes in drift density and biomass.

Table 11-6. Sand-dwelling taxa in reference sites and the channelized Kissimmee River, and taxa likely to colonize restored sand habitats of the Kissimmee River.

Taxon	Satilla River[1]	Ogeechee River[2]	Kissimmee-Pool A	Kissimmee-Pool C	Restored Kissimmee	Reference
Diptera						
Corynoneura	X***	X			X	Merritt et al. 1996
Cladotanytarsus	X**	X				
Cryptochironomus	X**	X			X	Merritt et al. 1996
Lopescaldius		X			X	Epler 1992
Parakiefferiella	X***	X			X	Epler 1992
Paracladoplelma					X	Epler 1992
Polypedilum	X**	X	X[#]	X[#]	X	Merritt et al. 1996
Rheosmittia		X			X?	Epler 1992
Robackia	X***	X			X	Epler 1992
Tanytarsus	X**				X	Merritt et al. 1996
Tanytarsini group			X[#]	X[#]	X	Merritt et al. 1996
Thienemaniella	X**				X	Epler 1992
Orthocladinae		X			X	Epler 1992
Ceratopogonidae	X***	X			X	Merritt et al. 1996
Ephemeroptera						
Stenonema					X	Berner&Pescador 1988
Cercobrachys					X	Berner&Pescador 1988
Mollusca						
Musculium					X	Toth 1991
Pisidium					X	Toth 1991
Corbicula fluminea		X			X	Toth 1991
Trichoptera						
Nectopsyche					X	Pescador et al. 1995
Oecetis					X	Merritt et al. 1996
Setodes					X	Merritt et al. 1996

** = frequent
*** = abundant
[#] = rare

1 = Benke et al. 1984, 2 = Stites 1986

Bi-directional Exchange of Aquatic Invertebrates between River Channel and Floodplain

Reliable reference conditions for bi-directional exchange of aquatic invertebrates between river channel and floodplain habitats do not exist; therefore, a specific expectation for restoration of this component can not be developed. However, because this functional attribute is a key characteristic of healthy river-floodplain systems, and critical to the productivity of higher trophic levels in the river channel and floodplain, it will be evaluated as part of the comprehensive restoration evaluation program to determine restoration of ecological integrity within the Kissimmee river-floodplain system.

Methods. A review of the literature revealed only one study that documented the bi-directional exchange of aquatic invertebrate numbers and biomass between river channels and floodplains (Smock 1994). Drift into and out of two first-order blackwater streams (Colliers Creek and Buzzards Branch) in Virginia was conducted between 1990–1991. Specific sampling methods can be found in Smock (1994).

Results. Because channelization eliminated stage fluctuations within remnant channels of the Kissimmee River, movement of invertebrates to and from the floodplain was considered zero for the baseline condition.

For Colliers Creek, total input of invertebrates to the floodplain over the year by drifting was 1.47 X 10^6 individuals and 0.25 kg dry mass; total output to the channel was 2.68 X 10^6 individuals and 0.15 kg dry mass. Therefore, net exchange through drift was 1.21 X 10^6 individuals to the channel and 0.10 kg dry mass to the floodplain. Copepods, chironomids, and ostracods accounted for most of the net output of individuals from the floodplain, while net input of biomass to the floodplain was primarily by Ephemeroptera, Trichoptera, and Isopods as well as several rare but large taxa (e.g., Odonata and Megaloptera). A total of 2.10 X 10^5 and 0.66 kg dry mass moved onto the floodplain by crawling, with total output to the channel of 0.40 X 10^5 individuals and 0.05 kg dry mass. Therefore, net movement by crawling was 1.70 X 10^5 individuals and 0.61 kg dry mass. Drift and crawling accounted for a net export of 1.04 X 10^6 individuals from the floodplain over the year, but an import of 0.71 kg of biomass from the channel.

For Buzzards Branch, drift densities, biomass concentration, and biomass drift rates were significantly higher in water flowing into than out of the Buzzards Branch floodplain. Copepods and chironomids were the most abundant taxa drifting between the floodplain and channel. Very few individuals crawled between the channel and floodplain at Buzzards Branch. Results of this study indicate that while there may be substantial exchange of organisms across the river-floodplain boundary in these two systems, the floodplains, which produce 67–95% of annual invertebrate production in the two stream systems, retained most of that production.

Discussion and Comparison with Baseline Condition. Although no specific expectation for bi-directional exchange of aquatic invertebrates between the river channel and floodplain has been developed due to lack of reference data, restoration of pre-channelization discharge and floodplain hydroperiod is expected to result in a net movement of invertebrate number and biomass from the river channel to the floodplain during the rising hydrograph (initial flood-pulse), and a net influx of invertebrate numbers and biomass from the floodplain to the river channel during the falling hydrograph.

Sampling of invertebrate exchange will begin approximately two years after initiating the revised headwaters regulation schedule. Paired, replicate (three) drift nets (900 cm^2), equipped with 125 µm mesh netting, will be placed at pre-determined locations at the interface between the floodplain and river channel, in order to capture invertebrates moving onto and off of the floodplain during the rising hydrograph, and onto and off of the floodplain during the falling hydrograph. Nets will be set for three-hour intervals, at four time periods, over a 24-hour period. Nets will be checked and replaced every one hour (or as necessary) to prevent clogging. Current velocity and water depth will be measured at the opening of each net prior to setting and upon retrieval to determine the volume of water sampled. Sampling will occur four times annually, twice on the rising hydrograph, and twice on the falling hydrograph. Actual sampling dates will be determined from daily river channel stage data and visual observations of overbank flow and recession of water from the floodplain. This sampling routine is designed to evaluate temporal variability of import and export from the floodplain over a 24-hour period, and may be adjusted following analyses of initial data.

Secondary Production of Floodplain Aquatic Invertebrates

Methods. A literature review found no information on macroinvertebrate production in pre-channelization marshes of the Kissimmee River or in marshes with similar characteristics as pre-channelization marshes. Therefore, baseline data collected in floodplain habitats (pasture and remnant Broadleaf Marsh) in Pool C was used to predict the minimum level of macroinvertebrate productivity in restored Broadleaf Marsh (currently pasture).

In order to estimate production of aquatic invertebrates in remnant marshes of the Kissimmee River floodplain, replicate (three) stovepipe samples were collected quarterly between August 1995 and May 1997 in pools A and Pool C. Samples were analyzed for species identity, density, and biomass. Production was calculated using the instantaneous growth rate (IGR) method.

Results. Pasture (UP, upland herbaceous vegetation, Bousquin and Carnal 2005) habitats in the channelized system were dry most of the year; therefore, aquatic macroinvertebrate community production within this habitat was assumed to be 0 $g/m^2/yr$. Production of aquatic invertebrates in altered Broadleaf Marsh of the Kissimmee River is low. Remnant BLM in Pools A and C was dry over much of the sample period. Annual invertebrate community production in Pool A and C was 6.4 and 6.0 $g/m^2/yr$, respectively.

Discussion and Comparison with Baseline Condition. Because production of aquatic invertebrates is critical to energy flow pathways in aquatic systems, and production of floodplain invertebrate communities can be several orders of magnitude greater than river channel production, it is important to estimate production of floodplain aquatic invertebrates in order to predict the amount of biomass available for transfer to higher trophic levels.

The expectation for increased aquatic macroinvertebrate production above that of the reference condition is based on expectations for restored aquatic invertebrate community structure, including an increase in species richness, year-round persistence of a diverse aquatic invertebrate community, increases in mean annual biomass for most taxa, and the potential for high biomass turnover rates (annual P/B ratios) for many taxa. Because the magnitude of production depends on standing stock biomass and biomass turnover rates, factors affecting one, or both, will influence rates of production (Benke 1984). Dipterans may account for >30% of all taxa and >50% of total individuals in natural flatwoods marshes of central Florida (Evans et al. 1999). Assuming a cohort P/B ratio of 5 (Waters 1969) and a mean developmental time of 21 days, annual P/B ratios for many dipterans can approach 90, which means biomass turnover time may be as short as four days. Annual P/Bs in this range and greater have been reported for numerous Diptera from a variety of aquatic systems (Benke 1998), and indicates the potential for high turnover rates for some taxa to contribute to high rates of annual production. Densities of large invertebrates (e.g., crayfish, grass shrimp, amphipods, and odonates) can be high in natural marshes of central and south Florida (Jordan et al. 1996a, 1996b, Milleson 1976, J.W. Koebel, personal observation). Mean crayfish density within a Broadleaf Marsh of the channelized Kissimmee River approached 40/m^2 when the marsh was inundated to a depth >20 cm (J.W. Koebel, personal observation). Moderate mean annual density and associated biomass of crayfish and other large invertebrates is expected in restored Broadleaf Marsh habitats, and likely will contribute to a high rate of annual invertebrate community production.

Sampling of remnant Broadleaf Marsh and reestablished Broadleaf Marsh (pasture in the channelized system) will commence approximately two years after initiating the revised headwaters regulation schedule. This time frame should be sufficient for reestablishing pre-channelization floodplain vegetation characteristics. Methods will be similar to those outlined in Anderson et al. (1998b), and include collection of monthly, replicate (five) throwtrap (area = 0.25 m^2) samples from randomly selected locations within remnant and restored Broadleaf Marsh in Pools A and C. Samples will be analyzed for species identity, density, and standing stock biomass. Production will be calculated using the instantaneous growth rate method (IGR). Sampling in remnant and restored marsh will continue for three years. The three independent estimates of annual production will be averaged to determine mean annual production, which will be compared to baseline data and the expectation.

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CHAPTER 12

AMPHIBIAN AND REPTILE COMMUNITIES OF THE LOWER KISSIMMEE RIVER BASIN PRIOR TO RESTORATION: BASELINE AND REFERENCE CONDITIONS AND EXPECTATIONS FOR RESTOATION

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ABSTRACT: To characterize baseline (channelized) conditions in the Kissimmee River ecosystem, herpetofauna were surveyed using multiple sampling techniques within several altered floodplain habitats. Amphibian and reptile species richness within the channelized lower Kissimmee basin was similar to that of other disturbed wetland sites of south-central Florida. Many taxa characteristic of undisturbed wetland and upland habitats of central Florida were absent from the baseline surveys. Data were compared to distributions of amphibians and reptiles in central Florida, and with data collected from undisturbed wetlands on the Avon Park Bombing Range, to define reference conditions and evaluate whether channelization altered herpetofaunal community structure and patterns of amphibian reproduction in floodplain habitats. Comparisons suggest that herpetofaunal community structure and patterns of amphibian reproduction in floodplain habitats were severely impacted by channelization. Expectations of changes predicted to result from restoration were developed based on the data presented in this report. The expectation for restoration of community structure predicts that at least 24 amphibian and reptile taxa considered "characteristic" or "frequently occurring" in natural broadleaf marshes (BLM) of central Florida will recolonize restored floodplain habitats within three years of reestablishing hydroperiod and vegetation characteristics similar to the pre-channelization period. The expectation for amphibian reproduction predicts that larval amphibians will be present in restored BLM for at least seven months each year.

INTRODUCTION

Amphibian and reptile (herpetofauna) communities can serve as indicators of the health of aquatic ecosystems, especially wetlands. Adult and larval herpetofauna play an integral role in food web dynamics and energy flow through aquatic and terrestrial ecosystems. They are major consumers of invertebrates and algae (Blaustein and Wake 1990) and, in turn, are consumed by a variety of invertebrates (Travis et al. 1985, Roth and Jackson 1987), fishes (Azevedo-Ramos et al. 1999), birds (Ogden et al. 1976, Collopy and Jelks 1989, Beissinger 1990), and other amphibians and reptiles (Morin 1983, Wilbur et al. 1983, Ashton and Ashton 1988).

Amphibians are of particular interest because of their complex life cycle which includes obligate association of larvae with water and may include a terrestrial or semi-terrestrial adult stage. Thus, environmental conditions within aquatic and terrestrial habitats must be favorable for reproduction, development, and survival. Adult and larval amphibians are vulnerable to low temperature, drought, and shifts in wetland hydrology (Pechmann et al. 1989, Stebbins and Cohen 1995).

Conversion of wetlands to uplands combined with shortened and unpredictable hydroperiods in remnant wetlands following the channelization of the Kissimmee River are likely to have altered herpetofaunal communities. Restoration of pre-channelization hydrology, including long-term floodplain inundation through the Kissimmee River Restoration Project, is expected to reestablish historic floodplain wetland plant communities in the central portion of the Kissimmee river/floodplain ecosystem. Herpetofauna are important biological components for assessing restoration of ecological integrity within the Kissimmee River ecosystem.

Objectives

The objectives of this study are to:

(1) Assess baseline (channelized, pre-restoration) amphibian and reptile community structure in of the Kissimmee River and floodplain;
(2) Assess temporal patterns of amphibian reproduction during the baseline period;
(3) Estimate pre-channelization conditions for amphibian and reptile community structure characteristics and patterns of anuran reproduction using reference data;
(4) Quantify impacts of channelization by comparing pre-channelization (reference) conditions and baseline conditions; and
(5) Develop specific expectations for restoration of herpetofaunal community structure and amphibian reproduction.

BASELINE CONDITIONS

Methods

Study Site

Sampling for herpetofaunal community structure characteristics and patterns of amphibian reproduction was stratified by habitat (plant community). Sampled habitats included Broadleaf Marsh (BLM); Woody Shrub (S.MCF); Upland Herbaceous plant communities (UP); Wetland Forest (WF); and Upland Forest (UF). Broadleaf Marsh habitats are spatially homogeneous, primarily consisting of arrowhead (*Sagittaria lancifolia*), pickerelweed (*Pontederia cordata*), and maidencane (*Panicum hemitomon*). Woody Shrub is characterized by dense stands of wax myrtle (*Myrica cerifera*) that exist on a bog-like floating mat. The understory is composed of a diverse mixture of broadleaf marsh, wet prairie, and upland vegetation including broomsedge (*Andropogon glomeratus*), sedges (*Cyperus* spp.), pennywort (*Hydrocotyle umbellata*), spatterdock (*Nuphar lutea*), rushes (*Rhynchospora* spp.), and Cuban bulrush (*Scirpus cubensis*). Upland herbaceous communities (pasture) are characterized by upland and mesic grasses, forbs, and shrubs. Wetland Forest habitats are characterized by the presence of red maple (*Acer rubrum*), cabbage palm (*Sabel palmetto*), live oak (*Quercus virginiana*), Pteridophyta, American cupscale (*Sacciolepis striata*), and greenbriar (*Smilax* spp.), while Upland Forest is characterized by *Q. virginiana* and *S. palmetto*. More explicit definitions of these plant communities can be found in Bousquin and Carnal (2005). Sample methods and sample habitats varied according to what metric was being measured.

Visual Encounter Surveys

Visual encounter surveys (VES) (Donnelly et al. 1998a) were conducted monthly over a 31 month period in BLM (Pools A and C) and S.MCF (Pools C and D) habitats, and a 15 month period in WF (Pools B and C) habitats, beginning in August 1996. Surveys were conducted over a 12 month period in UP (Pools A and C) habitats beginning in March 1998. One group of three 50 meter long permanent line transects, divided into five-meter intervals, was established within each habitat approximately 100 meters from and adjacent to the river channel. Transects were set perpendicular to the river channel and separated by 20 meters. In March 1998, in order to more accurately characterize the herpetofaunal community in BLM and S.MCF, six additional 50 meter transects (two groups of three) were established in BLM and

S.MCF. Nine 50 meter transects (three groups of three) also were established at this time in UP habitats. No additional transects were established in WF due to the limited areal extent of this habitat. The specific location of each transect group within each habitat was based on habitat availability, habitat size, and ease of access.

Each transect was surveyed once per sampling event. Sampling events began approximately 30 minutes after sunset. Head lamps or bright flashlights were used to illuminate a one-meter wide strip on each side of the transect line. For every amphibian and reptile encountered, species identity, age class (larva, juvenile, adult), perch height, and substrate association were recorded. Water depth was recorded at 0, 25, and 50 m on each transect using a permanently mounted stream gauge or meter stick.

Community structure was described by species richness (S = the total number of species present); relative abundance (the proportion of individuals of species *i* in relation to the total number of individuals); species diversity (H'), where $H' = -\Sigma(p_i \ln p_i)$ and p_i is the proportion of individuals belonging to the i^{th} taxa; and community evenness (J'), where $J' = H'/\ln S$ (Price 1984). A coefficient of community similarity (CCS) calculated as

$$\frac{\Sigma\,(2m_i)}{\Sigma\,(a_i + b_i)},$$

where a_i is the abundance of species *i* in community *a* (Control site), *b* is the abundance of species *i* in community *b* (Impact site), and m_i is the minimum value for that species in community *a* or *b* (Bray and Curtis 1957), also was calculated for each habitat. A species accumulation curve was developed for each habitat in each pool. An accumulation curve shows the cumulative number of species observed during successive sampling periods. Accumulation curves usually rise sharply during the initial sampling periods but approach an asymtote as the species list for an area or habitat nears completion (Heyer et al. 1994).

Drift Fence Arrays

Drift fence arrays (Donnelly et al. 1998b) were sampled monthly in UP and oak/cabbage palm (upland forest, UF) hammocks in Pools A (hammock only) and C from February–March 1997 through September 1998. Replicate (three), cross-shaped arrays consisting of four, 15 meter long sections of aluminum flashing were partially sunk into the soil. Each array was separated by at least 20 meters. Each fence had one pit-fall trap (plastic 19 L bucket) at each end (n=4). In the middle of each side of the fence were either funnel traps, which were constructed of flexible window screen, or pit-fall traps. Pit-fall traps were buried in the soil so that the bucket lip was approximately 2.5 cm below the soil surface. Funnel traps were held against each fence with duct tape. Holes were drilled into the bottom of each pit-fall trap to provide drainage. A damp sponge was placed in each trap to prevent desiccation of captured animals and each trap was shaded with a tempered Masonite© board. Traps were opened for 24–96 continuous hours and checked daily. Species identity was recorded for each captured animal. Species richness, relative abundance, species diversity, and community evenness were calculated for each habitat within each pool. Community similarity was calculated for each habitat between pools.

Larval Amphibians

Larval amphibians within BLM and S.MCF habitats were sampled monthly from March 1997 through February 1999 with a 1 m^2 aluminum throwtrap. Larval amphibians within UP habitats were sampled monthly from April 1998 through March 1999. Five replicates (first nine months) or ten replicates (last 14 months) were collected from randomly selected locations within each habitat type on each sampling date. Sample locations were determined by traveling a randomly determined distance ($\leq$400 m) and direction (0–360°) from a randomly determined starting point within each habitat. Following trap placement, all vegetation within the trap was identified and counted. Water depth within the trap was recorded at each corner and at the center of the trap. All vegetation was removed and larvae were dip-netted from the trap. Dip-nets were equipped with 1 mm mesh netting. Dip-netting continued until no larvae were collected from ten consecutive dips. All larvae were preserved in 10% buffered formalin and stored for future identification.

In the laboratory, larval salamanders were identified using Altig and Ireland (1984) and Conant and Collins (1991). Ronald Altig (Mississippi State University) identified larval anurans. Body length and total length of all larvae were measured to the nearest 0.1 mm. Developmental stage of larval anurans was determined from Gosner (1960). Larval amphibian species richness was calculated for each habitat within each pool on each sampling date.

River Channel Turtle Community Structure

River channel turtles were sampled monthly from January 1997–September 1998 within remnant river channels and C-38 in Pools A and C. During the first seven months, 1 m diameter, 2.5 m long, single-throated hoopnets were used, but proved inefficient at capturing turtles. Consequently, 1.3 m diameter, 5 m long, double-throated hoopnets were used for the remainder of the study. Three hoopnets and three aluminum frame box traps were set in randomly selected locations in each of three remnant channels and in C-38 on each sampling date. Sample locations were selected by traveling at a constant boat speed (~ 1000 rpm) for a randomly determined time period through each remnant channel. Box traps were baited with sardines and placed along the deep-water edge of littoral vegetation, or within open water areas. Hoopnets were baited with salt pork or raw chicken and placed in deeper sections of each channel adjacent to emergent or floating vegetation. Nets contacted the substrate and were supported with 5 cm diameter PVC poles anchored to the substrate. Traps were set for a maximum of 96 hours during each month; however, time of deployment usually was less than 12 hours. Additionally, if time permitted, turtles were captured using a long-handled dip-net. Each turtle was identified to species, weighed, and marked with a unique coded tag or carapace mark (Cagle 1939), and released. Testudine species richness was calculated for each pool.

Casual Observations

Opportunistic observations of amphibians and reptiles also were recorded during this and other non-herpetological studies within the restoration project area from August 1995 through March 1999. When possible, amphibians and reptiles were captured and identified to the lowest possible taxonomic level, and released.

Results

A total of 48 taxa (Table 12-1; see Appendix 12-1A for common names of taxa), including 20 amphibians and 28 reptiles, were captured or encountered with all sampling methods. Nine taxa were encountered only once, and four are introduced species (Wilson and Porras 1983). Species richness was highest in Upland Hammock (20), followed by Broadleaf Marsh (19), Woody Shrub (17), Upland Herbaceous (14), and Wetland Forest (5). Species diversity and community evenness were low in all floodplain habitats in all pools (Table 12-2).

Visual Encounter Surveys

Visual encounter surveys detected 14 amphibian and reptile species within four floodplain habitats of the channelized Kissimmee River (Table 12-3). The number of species observed quickly accumulated in WF, with all species encountered within two months (Figure 12-1). Accumulation of species in UP was slower, with all species encountered within eight months. Species accumulated even more slowly in S.MCF and BLM, with all species encountered after 23 months (Figure 12-1).

Species richness was highest in S.MCF habitats with eight and eleven species present in Pools C and D, respectively (Table 12-2). Seven species were encountered in both pools; one species *(Rana grylio)* was found only in Pool C, and four species (*Gastrophryne carolinensis, Nerodia fasciata, Notopthalmus viridescens piaropicola,* and *Elaphe guttata*) were found only in Pool D (Table 12-3). Eight and six species were observed within BLM habitats of Pools A and C, respectively. Five species were present in both pools, three species (*Thamnophis sauritus, G. carolinensis*, and *Rana sphenocephala*) were found only in Pool A, and one species (*Agkistrodon piscivorous*) was found only in Pool C (Table 12-3). Three and five species were observed in WF habitats of Pools B and C, respectively. Three species were present in both pools, with two additional species (*Hyla femoralis* and *G. carolinensis*) found only in Pool C (Table 12-3). Pasture habitat within Pools A supported four species (19 total encounters) while UP habitat in Pool C supported one species (three total encounters) (Table 12-3).

Table 12-1. Herpetofauna captured or encountered within surveyed habitats in the lower Kissimmee basin. BLM = Broadleaf marsh, S.MCF = Woody Shrub, WF = Wetland Forest, UH = Upland Hammock, UP = Upland Herbaceous, KR = Kissimmee River, C38 = C-38 canal, and B = Building.

REPTILES	BLM	S.MCF	WF	UH	UP	KR	C38	B
Emydidae:								
Pseudemys floridana						X	X	
Pseudemys nelsoni						X	X	
Kinosternidae:								
Kinosternon baurii				X		X		
Kinosternon subrubrum						X		
Sternotherus odoratus						X		
Testudinidae:								
Gopherus polyphemus				X		X*		
Trionychidae:								
Apalone ferrox						X	X	
Alligatoridae:								
Alligator mississippiensis		X				X	X	
Anguidae:								
Ophisaurus attenuatus				X				
Gekkonidae:								
Hemidactylus sp.								X
Iguanidae:								
Anolis carolinensis	X	X	X	X	X			
Anolis sagrei								X
Scincidae:								
Eumeces inexpectatus					X			
Scincella lateralis				X	X			
Colobridae:								
Coluber constrictor				X	X			
Diadophis punctatus				X	X			
Drymarchon corais				X				
Elaphe guttata					X			
Elaphe obsoleta		X						X
Nerodia fasciata	X	X				X	X	
Opheodrys aestivus					X			
Regina alleni						X		
Seminatrix pygaea				X				
Storeria dekayi					X			
Thamnophis sirtalis				X				
Thamnophis sauritus	X	X		X				
Viperidae:								
Agkistrodon piscivorus	X	X				X		
Crotalus adamanteus							X	
AMPHIBIANS								
Amphiumidae:								
Amphiuma means	X					X		
Plethodontidae:								
Eurycea quadridigitata	X	X						
Salamandridae:								
Notopthalmus viridescens	X	X						
Sirenidae:								
Pseudobranchus a. axanthus	X							
Siren intermedia	X					X		
Siren lacertina	X	X				X		

* Observed swimming across river channel.

Table 12-1. Continued.

	BLM	S.MCF	WF	UH	UP	KR	C38	B
Bufonidae:								
Bufo terrestris				X				
Bufo quercicus				X				
Hylidae:								
Acris gryllus	X	X						
Hyla cinerea	X	X	X	X	X	X	X	
Hyla femoralis	X		X	X				
Hyla squirella	X	X		X				
Osteopilus septentrionalis							X	
Pseudacris nigrita	X			X				
Pseudacris ocularis	X	X		X				
Leptodactylidae:								
Eleutherodactylus planirostris				X				
Microhylidae:								
Gastrophryne carolinensis	X	X	X	X	X			
Ranidae:								
Rana catesbeiana	X							
Rana grylio	X	X						
Rana sphenocephala	X	X	X	X	X			

Table 12-2. Community structure indices calculated from total encounters and captures during visual encounter surveys and drift fence sampling within baseline floodplain habitats. BLM = Broadleaf Marsh, S.MCF = Woody Shrub, WF = Wetland Forest, and Up = Upland Herbaceous.

Visual Encounter Survey	BLM		S.MCF		WF		UP	
Metric:	Pool A	Pool C	Pool C	Pool D	Pool B	Pool C	Pool A	Pool C
Species Richness (S')	8	6	8	11	3	5	4	1
Diversity (H')	0.43	1.25	1.19	1.19	0.21	0.31	1.11	0.00
Evenness(J')	0.21	0.70	0.57	0.50	0.19	0.19	0.80	0.00
Coefficient of Similarity	0.22		0.51		0.94		0.27	

Drift Fence Array	UH		UP	
Metric:	Pool A	Pool C	Pool B	Pool C
Species Richness (S')	10	14	7	5
Diversity (H')	0.99	1.95	0.69	0.81
Evenness(J')	0.41	0.74	0.35	0.5
Coefficient of Similarity	0.42		0.45	

Hyla cinerea was the most frequently observed species in each habitat at all times during this study (Table 12-3), accounting for 52.4, 60.4, 84.0, and 94.4% of total numbers within UP, S.MCF, BLM, and WF, respectively. Only four other species, *Eurycea quadridigitata*, *Anolis carolinensis*, *Pseudacris ocularis*, and *R. sphenocephala* accounted for greater than 5% of total numbers within any habitat.

Species diversity was low in all habitats (Table 12-2). Values of community evenness were low in Pool C UP (0.0), Pool A BLM (0.21), and Pools B and C WF (0.19 and 0.19, respectively), moderate in Pools C and D S.MCF (0.57 and 0.50, respectively) and high in Pool C BLM (0.70) and Pool A UP (0.80). A coefficient of community similarity, which was calculated for each habitat, indicated that WF habitats are very similar between Control and Impact pools, S.MCF habitats are moderately similar, and BLM and UP habitats are dissimilar in species abundance (Table 12-2).

Table 12-3. Total herpetofaunal observations during 31 monthly visual encounter surveys (VES) in Broadleaf Marsh (BLM) and Woody Shrub (S.MCF), 15 monthly VES in Wet Forest (WF), and 11 monthly VES in Upland Herbaceous (UP) habitats.

	BLM		S.MCF		WF		UP	
	Pool A	Pool C	Pool C	Pool D	Pool B	Pool C	Pool A	Pool C
Taxon:								
Acris gryllus dorsalis	1	5	0	0	0	0	0	0
Agkistrodon piscivorous conti	0	3	2	4	0	0	0	0
Anolis carolinensis	50	40	54	77	9	13	2	0
Elaphe guttata	0	0	0	1	0	0	0	0
Eurycea quadridigitata	11	1	20	169	0	0	0	0
Gastrophryne carolinensis	5	0	0	3	0	2	7	0
Hyla cinerea	1006	72	163	480	318	294	9	3
Hyla femoralis	0	0	0	0	0	5	0	0
Nerodia fasciata	0	0	0	1	0	0	0	0
Notopthalmus viridescens piaropicola	0	0	0	1	0	0	0	0
Pseudacris ocularis	24	54	3	17	0	0	1	0
Rana grylio	0	0	1	0	0	0	0	0
Rana sphenocephala	1	0	30	27	6	1	0	0
Thamnophis sauritus	9	0	1	11	0	0	0	0
Totals	1107	181	274	791	333	315	19	3

Drift Fence Arrays

Drift fence arrays revealed a combined total of nine amphibian and reptile species in UP habitats of Pools B and C, and a combined total of 18 species in oak hammock (UF) habitats of Pools A and C (Table 12-4). The number of captured species quickly accumulated in UP habitats, with four of five species in Pool C captured within four months, and all species in Pool B captured within five months (Figure 12-2a). The number of species accumulated more slowly in (UF) hammock habitats, with all species in Pools A and C captured after 17 months (Figure 12-2b).

Gastrophryne carolinensis accounted for 84% and 77% of total numbers in Pool B and C UP, respectively. *Rana sphenocephala* was the only other taxon accounting for greater than 5% of total numbers in UP habitats. Three species, *Bufo quercicus, B. terrestris*, and *H. cinerea* only occurred in Pool B, while *Eumeces inexpectatus* and *Diadophis punctatus* were collected only in Pool C.

Gastrophryne carolinensis accounted for 73% and 35% of total numbers in Pool A and C hammocks. *Rana sphenocephala*, *Scincella lateralis*, and *E. inexpectatus* also accounted for greater than 5% of total numbers in oak hammocks (UF).

Species diversity was low in both habitats, ranging from 0.69 in Pool B UP, to 1.95 in Pool C hammock. Community evenness was variable, ranging from 0.35 in Pool B UP to 0.74 in Pool C hammock (Table 12-2). A coefficient of community similarity indicates moderately dissimilar communities in UP habitats of Pools B and C, and upland hammocks of Pools A and C (Table 12-2).

Larval Amphibians

Larval amphibians occurred sporadically in BLM, S.MCF, and UP habitats of Pools A, C, and D. When there was water on the floodplain in Pool A BLM, larvae were present seven of nine months in 1997–1998 and one of seven months in 1998–1999. When there was water on the floodplain in Pool C BLM, larvae were present six of nine months in 1997–1998 and one of seven months in 1998–1999. When there was water on the floodplain in Pool C S.MCF, larvae were present six of 12 months in 1997–1998 and three of nine months in 1998–1999. When there was water on the floodplain in Pool D S.MCF, larvae were present seven of 12 months in 1997–1998 and five of nine months in 1998–1999. One larval *Rana* *sphenocephala* was found in both Pool A and C UP habitat during one month, which was the only month that water was present during the 1998–1999 sampling period.

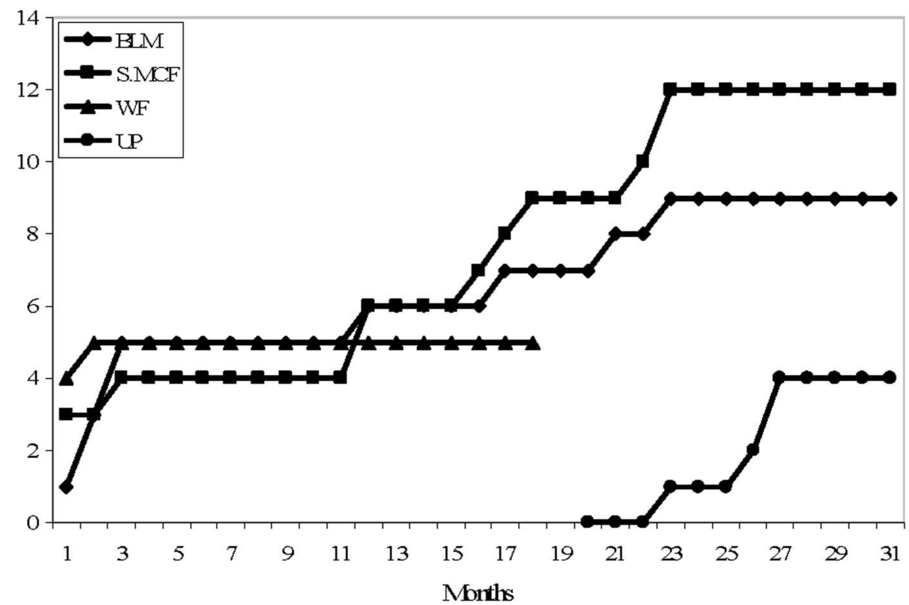


Figure 12-1. Species accumulation curves for floodplain visual encounter surveys. Accumulation curves show the cumulative number of species observed during successive sampling periods. BLM = Broadleaf Marsh, S.MCF = Woody Shrub, WF = Wetland Forest, and UP = Upland Herbaceous.

<u>Salamanders</u>

A total of five larval salamander taxa were collected from S.MCF, BLM, and UP habitats (Table 12-5). Species richness (4) and composition were identical between BLM habitat in Pools A and C. Species richness (4) was identical between S.MCF habitat in Pools C and D; however, these habitats had only three species in common. Pasture habitat in Pools A and C supported two and one larval salamander taxa, respectively.

Eurycea quadridigitata was collected most frequently and was most abundant in S.MCF habitats. Larvae first appeared in December 1997. Mean snout-vent (S-V) length increased from 11.0 mm to 19.1 and 19.3 mm, in Pools D and C respectively, between December 1997 and March 1998. Only adults were captured between May and December 1998, with larvae (mean S-V length = 13.0 and 15.7 mm in Pools D and C, respectively) reappearing in January 1999 in both pools. Larvae (mean S-V length = 16.4) also were collected in Pool C S.MCF in March 1999. *Eurycea quadridigitata* was less common in BLM habitats, although the seasonal pattern of reproduction was similar to S.MCF. *Eurycea quadridigitata* was not collected from UP habitats.

Larval *Siren lacertina* were collected from UP (Pool A), BLM (Pools A and C), and S.MCF (Pools C and D) habitats between December 1997 and April 1998. Other taxa rarely collected from any habitat included *Amphiuma means*, *Notopthalmus viridescens piaropicola*, *Siren intermedia intermedia*, and *Pseudobranchus axanthus axanthus*.

Table 12-4. Total herpetofaunal captures in drift fence, pit-fall trap, and funnel trap arrays. UH = Upland Hammock, UP = Upland Herbaceous.

	UH		UP	
Taxon:	Pool A	Pool C	Pool B	Pool C
Anolis carolinensis	1	1	0	0
Bufo quercicus	3	0	5	0
Bufo terrestris	2	0	2	0
Coluber constrictor	0	1	0	0
Diadophis punctatus	0	5	0	1
Drymarchon corais	0	1	0	0
Eleutherodactylus planirostris	9	0	0	0
Eumeces inexpectatus	0	8	0	2
Gastrophryne carolinensis	155	33	90	27
Hyla cinerea	5	5	2	0
Hyla femoralis	0	2	0	0
Kinosternon baurii	0	1	0	0
Ophisaurus attenuatus	0	1	0	0
Pseudacris nigrita verrucosa	0	0	1	0
Rana sphenocephala	29	19	6	4
Scincella lateralis	5	14	1	1
Seminatrix pygaea cyclas	1	0	0	0
Thamnophis sauritus sackenii	1	2	0	0
Thamnophis s. sirtalis	0	1	0	0
Total	211	94	107	35

Anurans

Ten larval anuran taxa were collected from floodplain habitats between April 1997 and February 1999 (Table 12-5). Overall larval anuran species richness was highest in BLM (10), followed by S.MCF (5), and UP (1); however, most taxa including *Acris gryllus*, *G. carolinensis*, *H. cinerea*, *H. femoralis*, *H. squirella*, *Pseudacris nigrita*, and *Rana catesbeiana* were captured infrequently.

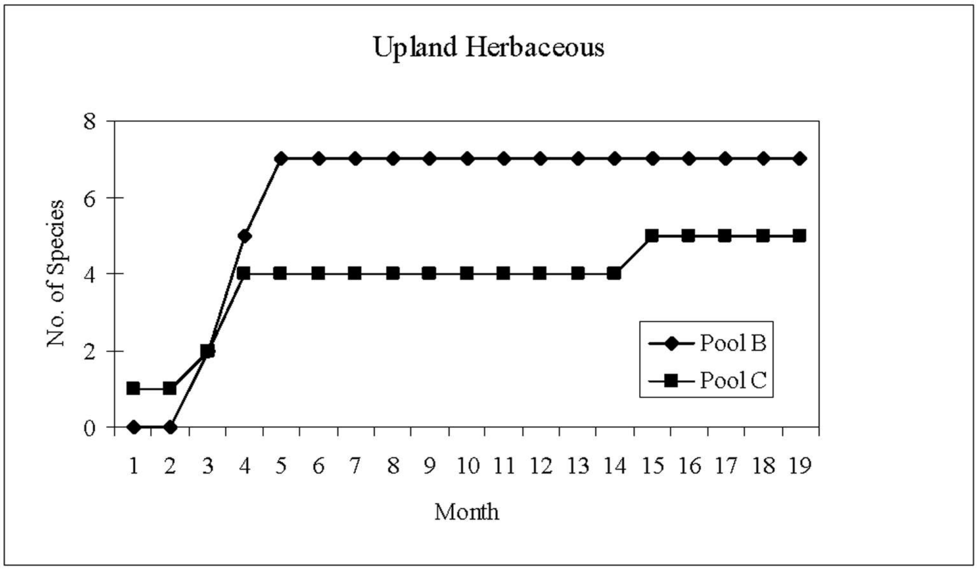
Pseudacris ocularis occurred monthly from October 1997 through January 1998 in Pool C BLM. Developmental stages of *P. ocularis* ranged from 27–36 in October to 39 in January. Larval *Rana sphenocephala* were captured on three dates between December 1997 and March 1998. Developmental stages ranged from 25 in December to 28–44 in March.

Within S.MCF habitats, mid-summer and spring patterns of development were apparent for *R. sphenocephala* and *R. grylio*, with larvae present in July–August (1997), December–April (1997–1998), and July–August (1998). Larvae collected in July–August (1997) were at developmental stage 25. Developmental stage of individuals collected in December–April ranged from 25–42. Individuals collected in July–August (1998) had attained a developmental stage of 25–26. Within UP habitats of Pools A and C, larval *R. sphenocephala* were each captured on one date. No other larval anurans were collected from UP habitats.

River Channel Turtle Community Structure

A total of 81 turtles (46 and 35 in Pools A and C, respectively), representing six taxa, were captured by hoopnet, box trap, or dip-net from remnant river channels and C-38 over a 20 month period beginning in January 1996. Captures occurred during approximately 6000 trap hours in Pool A and 6200 trap hours in Pool C. Seventy-nine percent of all turtles were captured in remnant river channels. In Pool A, *Pseudemys floridana peninsularis* accounted for 45.6% of total numbers and 50.1% of total mass, followed by *Pseudemys nelsoni* (43.5% and 35.8%, respectively), and *Apalone ferox* (8.7% and 14%, respectively). In Pool C, *P. floridana peninsularis* accounted for 34.3% of total numbers and 44.9% of total mass, followed by *P. nelsoni* (28.6% and 25%, respectively), and *A. ferox* (22.8% and 29.7%, respectively). Other less frequently captured turtles included *Stenotherus odoratus*, *Kinosternon bauri*, and *Kinosternon subrubrum steindachneri*.

A



B

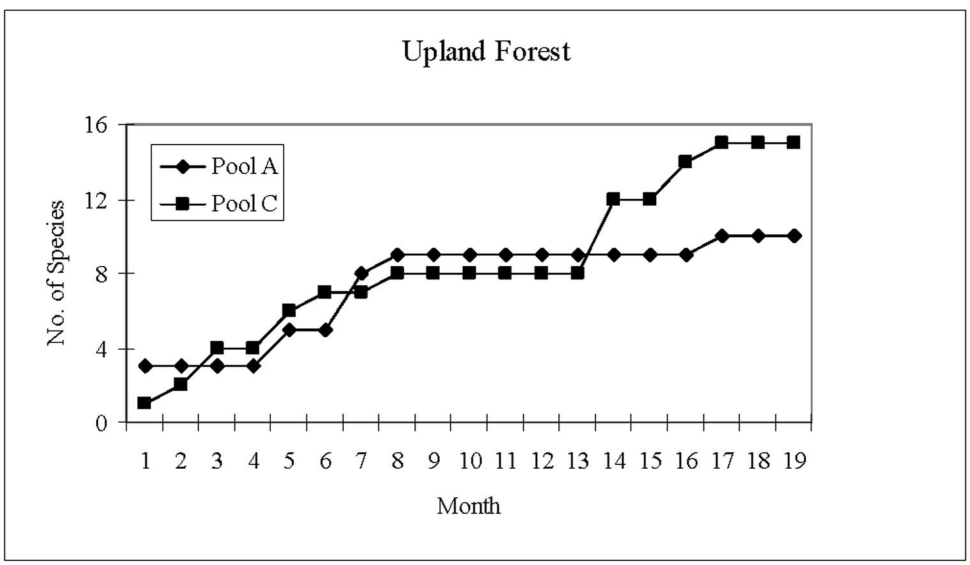


Figure 12-2. Species accumulation curves for drift fence, pit-fall, and funnel trap arrays.

Table 12-5. Habitat-specific occurrence of larval amphibians on the channelized Kissimmee River floodplain. BLM = Broadleaf Marsh, S.MCF = Woody Shrub, and UP = Upland Herbaceous.

	BLM		S.MCF		UP	
Taxa:	Pool A	Pool C	Pool C	Pool D	Pool A	Pool C
Salamanders:						
Eurycea quadridigitata	X	X	X	X		
Notopthalmus viridescens	X	X	X	X		
Pseudobranchus a. axanthus	X	X			X	
Siren i. intermedia			X			X
Siren lacertina	X	X	X	X	X	
Anurans:						
Acris gryllus dorsalis		X				
Gastrophryne carolinensis	X					
Hyla cinerea	X		X	X		
Hyla femoralis	X					
Hyla squirella	X	X	X	X		
Pseudacris nigrita verrucosa		X				
Pseudacris ocularis		X		X		
Rana catesbeiana	X					
Rana grylio		X	X	X		
Rana sphenocephala	X	X	X	X	X	X

Discussion

Visual Encounter Surveys

Visual encounter surveys can be an effective and economical means to determine species richness, species composition, and relative abundance of amphibians and reptiles within similar habitats. In addition, VES is an appropriate technique for both inventory and monitoring studies (Heyer et al. 1994, Pearman et al. 1995). The nine taxa observed in BLM and 12 taxa observed in S.MCF habitats over the baseline study period represent approximately 36% and 48% of all taxa likely to occur in natural wetlands of central Florida, respectively (Carr 1940, Franz et al. 2000). Although rare or cryptic taxa likely were overlooked during baseline surveys, data clearly indicate that remnant BLM and S.MCF habitats within the channelized Kissimmee River ecosystem support a depauperate wetland herpetofaunal community that is dominated in numbers by two or three species.

Visual encounter survey data from UP habitats (former broadleaf marsh) of the channelized river system indicate a severely impacted wetland herpetofaunal community. Although a total of 14 taxa were observed in UP habitats over the course of the baseline period (all methods), only seven wetland taxa were recorded, and only four (22 observations) were encountered by VES. These seven taxa represent 16% of taxa considered “characteristic” or “frequently occurring” in natural wetland habitats of central Florida (Carr 1940).

No historical or reference data on amphibian and reptile relative abundance, evenness, or diversity are available from the Kissimmee River ecosystem; therefore, no specific expectation for change in these metrics has been developed. However, based on expectations for hydrologic and habitat restoration, and knowledge of the occurrence of characteristic wetland herpetofaunal taxa within the lower Kissimmee basin (Franz et al. 2000), it is reasonable to hypothesize that species richness will increase within restored habitats (UP and BLM).

Drift Fence Arrays

Although some sampling bias is associated with drift fence sampling (Dodd 1991), drift fences combined with pitfall traps and funnel traps can be an effective technique for quantifying some animal populations, and usually capture some individuals of most species (Gibbons and Semlitsch 1981, Greenberg et al. 1994, Heyer et al. 1994). If one assumes that capture rates are similar between similar habitats, these data can be used to compare relative abundance of species among study areas (Heyer et al. 1994).

Pasture: Drift fence data clearly indicate that existing UP supports a depauperate herpetofaunal community dominated in numbers by one taxon and uncharacteristic of natural wetlands of central Florida (Carr 1940, Franz et al. 2000). Restoration of historic hydrologic patterns is expected to result in shifts in species richness, relative abundance, diversity, and evenness in UP habitats as they revert to BLM. Taxa characteristic of terrestrial habitats (e.g., *Bufo quercicus*, *B. terrestris*, *Eumeces inexpectatus*, and *Diadophis punctatus*) should emigrate to upland habitats while aquatic and semi-aquatic taxa colonize restored wetlands.

Upland Hammock: Taxa captured in pit-fall and funnel traps within oak hammocks (UF) represent approximately 33% of the species known to occur in upland hammocks of central Florida (Tennant 1997, Bartlett and Bartlett 1999). Although data indicate a somewhat depauperate community in upland hammocks, several factors may have contributed to low capture rates. Optimally, drift fences and pit-fall traps should be run continuously, with captured animals removed daily (Heyer et al. 1994). Available resources during the baseline period only allowed us to run traps for 24–96 hours per month. Extreme rainfall events associated with an El Niño Southern Oscillation Event (November 1997–March 1998) flooded hammocks and made sites inaccessible and pit-fall traps inoperable for approximately four months. The absence of most serpentines, which are often major components of the herpetofauna in upland habitats, may have been influenced by funnel trap design. Double-ended funnel traps used in this study were composed of lightweight window screen that had a tendency to collapse when taped to the drift fence. This likely prevented or deterred entrance by snakes, especially large-bodied individuals.

Because of the potential biases cited above, rare taxa likely were overlooked; however, taxa considered common and conspicuous in upland habitats within the Florida peninsula (Carr 1940) including *Elaphe guttata guttata*, *Elaphe obsoleta quadrivittata*, *Masticophis flagellum flagellum*, *Micrurus fulvius fulvius*, *Terrapene carolina bauri*, *Ophisaurus ventralis*, *Cnemidophorus sexlineatus*, *Scaphiopus h. holbrookii*, *Hyla gratiosa*, and *Hyla squirella* were never captured in upland habitats of the channelized Kissimmee River, indicating that channelization, or post-channelization impacts to uplands, may have altered population numbers and/or spatial patterns of distribution for some taxa.

Although no specific expectation for restoration of upland herpetofaunal communities has been developed, post-construction changes in community composition are likely to occur. We suggest, if sufficient resources are available, that these populations be monitored biannually (wet and dry season) to determine seasonal patterns of richness and abundance. Because seven taxa (~15% of the total) were unique to upland hammocks, these data are important in developing an accurate herpetofaunal inventory, which may serve as a useful indicator of biodiversity within the lower Kissimmee basin. Because post-construction data will not be directly compared to baseline data, additional sampling techniques including coverboards and PVC pipes should be incorporated into the sampling design to potentially encounter cryptic species. We also recommend that drift fences with pit-fall traps and rigid funnel traps be run for a minimum of 30 consecutive days during each season.

Larval Amphibians

Salamanders: Six salamander species are known to occur within the lower Kissimmee River basin (Table 12-2), and may be seasonally abundant in suitable habitats (Bartlett and Bartlett 1999). The dwarf salamander, *Eurycea quadridigitata*, was the most abundant salamander encountered during the baseline sample period. Increased visual observations of adult *E. quadridigitata* between August 1997 and February 1998 within S.MCF preceded a sharp increase in the occurrence of larval *E. quadridigitata* from January through April 1998 and again in January 1999. This correlation between increased adult and larval abundance corresponds well with breeding migrations and reproduction of *E. quadridigitata* on the upper Coastal Plain of South Carolina and in Alabama (McMillan and Semlitsch 1980, Trauth 1983).

Less frequently encountered taxa including *Amphiuma means*, *Notopthalmus v. piaropicola*, *Siren i.intermedia*, *S. lacertina*, and *Pseudobranchus a. axanthus* are likely more common in the Kissimmee River ecosystem than the results of this survey indicate. All are typical of shallow, heavily vegetated, soft-bottom habitats including littoral margins of remnant channels and long hydroperiod wetlands (e.g., S.MCF); however, they are often undetected due to their nocturnal and cryptic behavior (Bartlett and Bartlett 1999). Little is known about the reproductive habits of these taxa; however, it is likely that they will persist, reproduce, and become more obvious in the restored system as long-term floodplain hydroperiods and suitable habitat are restored.

Anurans: In central Florida, most anurans can breed during any month (Conant and Collins 1991). Given the prolonged floodplain inundation frequencies within the pre-channelized system, it is likely that anuran reproduction and larval recruitment occurred during most of the year. Although larval amphibians likely were present year-round, community structure characteristics (e.g., species richness and relative abundance) within pre-channelization marshes likely were heavily influenced by the presence of avian predators during periods of low water, and piscine predators during periods of high water.

Within the channelized Kissimmee River system, the availability of suitable habitat likely is the critical factor influencing reproduction by adult anurans (and salamanders), and the development and recruitment of larvae. Channelization eliminated seasonal, long-term floodplain inundation frequencies and fluctuating stage, thereby eliminating much of the historic breeding habitat for anurans. Under channelized conditions, floodplain habitats are often only inundated during the rainy season (typically June–September) with hydroperiods varying from days to months, depending on frequency and amount of rainfall. During this study, atypical floodplain inundation patterns resulted from rainfall associated with the 1997–1998 El Niño Southern Oscillation event. During this period, larvae from at least five taxa were collected from floodplain habitats, with several taxa collected consistently over a seven-month period. The presence of at least one larval anuran taxa in ten of the 16 months (~62%) in which water was present on the floodplain, indicate the potential for extended anuran reproduction.

River Channel Turtle Community Structure

Turtles are common and often conspicuous inhabitants of slow-flowing rivers and marshes of the southeastern Coastal Plain of the United States (Meylan et al. 1992), and often represent the majority of vertebrate biomass in aquatic systems (Iverson 1982, Congdon et al. 1986). Predatory fish, large wading birds (Ernst et al. 1994), and raptors (Cagle 1950, Beissinger 1990, Walley 1993, Means and Harvey 1999) occasionally consume hatchling and juvenile turtles, whereas adult turtles have few natural enemies except *Alligator mississippiensis* (Valentine et al. 1972, Delany and Abercrombie 1986).

Turtles were observed along river channel margins during most times of the year, and were frequently observed basking on floating vegetation and small woody debris. A total of six taxa (Table 12-2) were captured during this study. *Chelydra serpentina osceola* and *Deirochelys reticularia chrysea* were not observed or captured within the lower Kissimmee basin although their presence is likely.

All turtle species present in the Kissimmee River ecosystem are typical of large river systems of the southeastern United States (Ernst et al. 1994) and are expected to remain a highly visible component of the restored system. Although there is no intent to measure shifts in testudine community structure following restoration, opportunistic observations of river channel turtles will be recorded. Specific attention will be given to restored floodplain habitats that should become primary sites for foraging and reproduction by aquatic turtles. Additionally, all turtles observed in upland habitats will be recorded to determine seasonal shifts in habitat use.

REFERENCE CONDITIONS

Methods

Amphibian and Reptile Community Structure and Amphibian Reproduction

Samples collected during the baseline study period from remnant but altered BLM in Pool C provide some insight into wetland herpetofauna taxa richness and amphibian reproduction in pre-channelization BLM habitats.

In order to locate additional potential sources of reference conditions for amphibian and reptile community structure and patterns of amphibian reproduction within BLM habitats, a thorough literature search was conducted using the State Library of Florida Online Computer Library Center FirstSearch service.

Results

Baseline Surveys

Fourteen amphibian and reptile taxa considered characteristic or frequent inhabitants of permanent wetlands of central Florida were captured or observed in remnant marshes of Pool A and C during the baseline study period (Table 12-6) (Carr 1940, Franz et al. 2000). These taxa represent approximately 56% of taxa most likely to occur in broadleaf marsh habitats in central Florida, and are expected to occur in restored marshes within the Kissimmee River ecosystem.

Table 12-6. Potential wetland taxa for indicating restoration of amphibian and reptile community structure in reestablished broadleaf marsh habitats of the Kissimmee River ecosystem. These taxa occur in natural marshes of the Avon Park Bombing Range (APBR) and are considered characteristic or frequent inhabitants of natural marshes of central Florida (Franz et al. 2000). Taxa that are underlined were collected from remnant, but altered, Broadleaf Marsh (BLM) in Pools A and C.

Amphibians	Reptiles
	<u>*Agkistrodon piscivorus conanti*</u>
<u>*Acris gryllus dorsalis*</u> *	*Alligator misissippiensis*
<u>*Amphiuma means*</u>	*Anolis carolinensis* *
<u>*Eurycea quadridigitata*</u>	*Farancia abacura abacura*
<u>*Hyla cinerea*</u>	*Nerodia floridana*
<u>*Hyla squirella* *</u>	*Pseudemys floridana penninsularis*
<u>*Notophthalmus viridescens piaropicola*</u>	*Pseudemys nelsoni*
<u>*Pseudacris nigrita verrucosa*</u>	*Regina alleni*
<u>*Pseudacris ocularis*</u>	*Seminatrix pygaea cyclas*
<u>*Rana grylio*</u>	*Sistrurus miliarius barbouri*
<u>*Rana sphenocephala* spp.</u>	*Storera dekayi victa*
Siren intermedia intermedia	<u>*Thamnophis sauritus sackenii*</u>
<u>*Siren lacertina*</u>	*Thamnophis sirtalis sirtalis*
	Apalone ferox

* Although these taxa are not considered characteristic or frequent inhabitants of APBR marshes, they do occur in remnant marshes of the Kissimmee River and are likely to occur in restored BLM.

Reference Site

Pre-channelization data on herpetofaunal community structure from the Kissimmee River ecosystem are limited. Our primary source of information on herpetofaunal species richness of pre-channelization Kissimmee River marshes is herpetofaunal surveys of permanent wetlands of APBR. The APBR borders the Kissimmee River in Pools A and B (Highlands and Polk Counties) and contains over 54,000 acres of natural wetlands, of which less than 5% have been directly disturbed or impacted. Franz et al. (2000) surveyed the APBR for sensitive herpetofaunal species between October 1996 and May 1998. Data from these surveys indicate that 24 wetland amphibian and reptile taxa are characteristic or frequently occur in permanent wetlands of the APBR (Table 12-6). Because these relatively undisturbed habitats are directly adjacent to the Kissimmee River, it is likely that these taxa also occurred in pre-channelization marshes of the Kissimmee River (Table 12-6). Additionally, Carr (1940) presents a comprehensive review of amphibian and reptile habitat distributions throughout Florida, and lists species that are characteristic or frequently occur within each habitat. Based on this review, 25 amphibian and reptile taxa likely inhabited BLM habitats of the pre-channelized Kissimmee River during some portion of their lifetime. Although reference conditions are solely derived from Franz et al. (2000), information from Carr (1940) provides supporting information on herpetofaunal taxa likely to occur in post-channelization marshes. Taxa that occur in marshes of the APBR were judged likely to occur in pre-channelization marshes of the Kissimmee River, and are expected to occur in restored floodplain marshes. Table 12-6 lists taxa that are characteristic or frequently occur in permanent wetlands of APBR (Franz et al. 2000).

Larval Anurans

No data on temporal patterns of amphibian reproduction within the pre-channelized Kissimmee River exist; however, baseline data collected from remnant BLM in Pools A and C provide some indication of the possible temporal patterns of reproduction by amphibians in the pre-channelized system. Data indicate the presence of larval amphibians in seven of nine months (78%) in 1997–1998 and one of seven months (14%) in 1998–1999 when water was present on the floodplain in Pool A remnant BLM (Table 12-7). Larval amphibians were present in six of nine months (67%) in 1997–98 and one of seven months (14%) in 1998–1999 when water was present on the floodplain in Pool C remnant BLM (Table 12-7). Overall, larval amphibians were present in 11 of 16 months (69%) when water was present on the floodplain in either Pool A or C (Table 12-7).

Discussion

Amphibian and Reptile Community Structure

Based on reference condition data, it is possible to estimate species richness of amphibian and reptile taxa inhabiting pre-channelization Kissimmee River floodplain marshes. Although data do not provide insights into temporal patterns of abundance or diversity, they do provide enough information to develop an expectation for the occurrence of amphibians and reptiles in restored (currently UP) floodplain marshes of the Kissimmee River. This expectation is based on reestablishing a full range of hydrologic variation within floodplain UP habitats, including floodplain hydroperiod and variable depth patterns. Restoration of pre-channelization hydrologic patterns will be the impetus for reestablishing BLM vegetation and an aquatic invertebrate community necessary for colonization and persistence of amphibians and reptiles. Adult colonists likely will emigrate from existing wetland depressions within the UP, or from the river's littoral zone.

Larval Anurans

Specific data on anuran reproduction and larval development in pre-channelization marshes of the Kissimmee River do not exist. However, this does not preclude the development of an expectation for temporal patterns of anuran reproduction in restored BLM. Several studies (Blair 1961, Brooks 1980, Diaz-Paniagua 1988) have documented the reproductive phenology of multiple-anuran species assemblages over several years. In each of these studies, reproduction by individual species was partitioned over many months, often encompassing spring, summer, fall and winter. In these cases, larvae of at least one species were present during the entire year. Given the subtropical climate and prolonged floodplain inundation frequencies within the pre-channelized Kissimmee River system, it is likely that anuran reproduction and larval recruitment occurred during most of the year. Table 12-8 presents the known breeding periods of anurans likely to occur in pre-channelization marshes of the Kissimmee River.

GENERAL DISCUSSION

Baseline Conditions

The herpetofaunal community of the lower Kissimmee River basin is moderately species rich (48); however, numerous taxa characteristic of natural wetlands and upland hammocks were rare or not recorded during the baseline period. Dalrymple (1988) and Meshaka (1997) encountered 51 and 53 species of amphibians and reptiles from four habitats on Long Pine Key, Everglades National Park, and a five-year study of seven habitats at a disturbed wetland site in central Florida, respectively. Enge and Wood (1998) captured or identified 64 taxa (25 amphibians and 39 reptiles) from 12 habitats in the Big Bend Wildlife Management Area, Taylor County, Florida, while Franz et al. (2000) identified 68 taxa from wetland and upland sites on the Avon Park Air Force Range, Highlands and Polk Counties, Florida.

Hydrology and habitat quality are two critical factors influencing species composition, distribution, and reproduction in herpetofaunal communities (Skelly 1997, Adams 1999, Bodie and Semlitsch 2000). Loss of floodplain habitat combined with irregular and unpredictable hydroperiods following channelization, likely has altered patterns of abundance, distribution, and reproduction for many taxa within the channelized system. However, without historical records, it is difficult to reach any conclusions regarding shifts in species composition of amphibian and reptile species from the Kissimmee basin following channelization.

Table 12-7. Seasonal distribution of larval amphibians in altered broadleaf marsh and pasture habitats of the Kissimmee River. Months underlined indicate months when water was present on the floodplain. BLM = Broadleaf Marsh, UP = Upland Herbaceous.

Pool A BLM

					1997												1998						
	A	M	<u>J</u>	J	<u>A</u>	<u>S</u>	<u>O</u>	<u>N</u>	<u>D</u>		<u>J</u>	<u>F</u>	<u>M</u>	<u>A</u>	M	J	<u>J</u>	<u>A</u>	<u>S</u>	<u>O</u>	<u>N</u>	D	<u>J</u>
<u>Anurans</u>																							
Gastrophryne carolinensis			X																				
Hyla cinerea						X																	
Hyla femoralis													X										
Hyla squirella					X																		
Hylidae			X																				
Rana catesbeiana			X																				
Rana sphenocephala									X			X											
<u>Salamanders</u>																							
Eurycea quadridigitata									X		X	X	X										X

Pool C BLM

					1997												1998						
	A	M	<u>J</u>	J	<u>A</u>	<u>S</u>	<u>O</u>	<u>N</u>	<u>D</u>		<u>J</u>	<u>F</u>	<u>M</u>	<u>A</u>	M	J	<u>J</u>	<u>A</u>	<u>S</u>	<u>O</u>	<u>N</u>	D	<u>J</u>
<u>Anurans</u>																							
Acris gryllus																		X					
Hyla cinerea													X										
Hyla femoralis																		X					
Hylidae													X										
Pseudacris nigrita							X																
Pseudacris ocularis							X	X	X	X													
Rana grylio							X						X										
Rana sphenocephala									X			X	X										

Pool A UP

						1998						
	<u>M</u>	A	M	J	J	A	S	O	N	D	J	F
<u>Anurans</u>												
Rana sphenocephala	X											

Pool C UP

						1998						
	<u>M</u>	A	M	J	J	A	S	O	N	D	J	F
<u>Anurans</u>												
Rana sphenocephala	X											

Taxa collected or observed during the study (excluding introduced species) represent approximately 65% of native taxa likely to occur within wetland and upland habitats of the lower Kissimmee basin. The rarity or absence of characteristic and common taxa from floodplain habitats suggests that channelization and loss of habitat contributed to the decline or temporary elimination of some taxa.

Reference Conditions

Amphibian and Reptile Community Structure

Pre-channelization data from the lower Kissimmee River basin would provide the best reference conditions for assessing amphibian and reptile responses to Kissimmee River restoration. However, in the absence of pre-channelization data, records of amphibian and reptile distributions in natural wetlands of the APBR provide reasonable reference conditions for comparing pre- and post-restoration herpetofaunal communities (Franz et al. 2000). Additionally, historical records on the distribution and habitat preferences of amphibians and reptiles of central Florida provide additional information on potential taxa that may occur following restoration (Carr 1940).

S.MCF habitats will be excluded from initial post-construction studies. Although herpetofaunal community structure characteristics in S.MCF habitats are eventually expected to change as BLM vegetation becomes reestablished, this change is not expected for several (three–five or more) years. Once S.MCF habitats revert to BLM, post-construction sampling will commence.

Table 12-8. Florida breeding periods of amphibian species likely to colonize existing Broadleaf Marsh, Woody Shrub, and restored Broadleaf Marsh habitats currently characterized as pasture. Breeding periods are from Mount (1975) and Conant and Collins (1991).

Indicator Species	Spring	Summer	Autumn	Winter
Anurans:				
Acris gryllus dorsalis	X	X	X	X
Gastrophryne carolinensis	X	X	X	
Hyla cinerea	X	X	X	
*Hyla femoralis**	X	X	X	
*Hyla gratiosa**	X	X		
*Hyla squirella**	X	X	X	
Pseudacris nigrita verrucosa	X	X	X	X
Pseudacris ocularis	X	X	X	
Rana catesbeiana	X	X	X	
Rana grylio	X	X	X	X
Rana sphenocephala	X	X	X	X
Salamanders:				
Amphiuma means	X			
Eurycea quadridigitata	X		X	X
*Pseudobranchus a. axanthus***				
Siren i.intermedia	X			
Siren lacertina	X			

* Likely to occur near upland edge of floodplain.
** Breeding habits unknown.

Larval Amphibians

Reference conditions for the presence of larval amphibians in restored floodplain marshes are less rigorously defined. However, assuming that adult amphibians colonize restored marshes, there are no known factors that should prohibit adults from initiating breeding activities. Because of the potential for temporal partitioning of breeding among a multi-species assemblage, it is likely that larval amphibians will be present at least seven months each year.

The presence of larval amphibians in restored BLM will be determined from replicate, monthly throwtrap samples collected in the same BLM and UP habitats sampled during the baseline period in Pools A and C. Sampling of larval amphibians will commence approximately three years after reestablishing pre-channelization floodplain hydroperiods, and continue for a period of three years.

Comparisons and Expectations

Channelization of the Kissimmee River and subsequent draining of wetlands, severely impacted amphibian and reptile community structure and temporal patterns of anuran reproduction in floodplain habitats. Species richness in UP habitats (formerly BLM) is approximately five times lower than natural marshes of the APBR (Franz et al. 2000) and natural marshes of central Florida, as described by Carr

(1940). Periods of anuran reproduction in the channelized system appear to be governed by floodplain inundation patterns, which are highly unpredictable. Based on comparisons of baseline and reference data for community structure characteristics and patterns of amphibian reproduction, restoration of the Kissimmee River ecosystem should result in increased amphibian and reptile species richness ($\geq$24) in restored BLM, and near year-round reproduction by amphibians. The following expectations have been developed from baseline data and best available reference data.

Expectation: Number of amphibians and reptiles using the floodplain

Herpetofaunal taxa were rare in sampled UP habitats, all of which were BLM habitat prior to channelization. Five taxa (22 individuals) were observed over the 12 month sample period in Pool A and C, and represent approximately 20% of all wetland taxa considered characteristic or frequently occurring in BLM throughout central Florida. Additionally, these five taxa account for approximately 21% of the wetland taxa occurring in natural marshes of the APBR (Figure 12-3). Restoration of pre-channelization hydrologic characteristics within the lower Kissimmee basin will be the impetus for reestablishing BLM communities in areas that currently exist as UP. Our expectation for restoration of amphibian and reptile community structure in restored BLM, which currently exist as UP, predict the presence of at least 24 taxa. A community composed of 24 taxa represents nearly all taxa that are throughout undisturbed wetlands of central Florida (Carr 1940, Franz et al. 2000), and a >400% increase over the number of wetland taxa currently found in UP habitats of Pool C. This expectation does not imply the continuous presence of 24 taxa; rather, 24 taxa will be observed cumulatively within these habitats three years after restoration of pre-channelization hydrologic characteristics (Koebel 2005a).

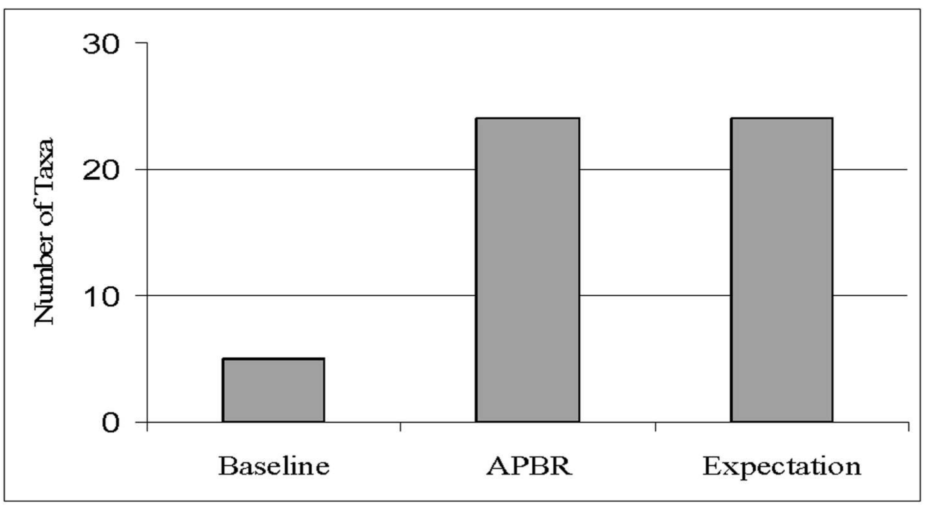


Figure 12-3. Number of taxa occurring in pasture habitats during the baseline period and number of taxa expected to occur in restored BLM following restoration. The expectation is based on the number of characteristic or frequently occurring wetland taxa in natural marshes of Avon Park Bombing Range, Highlands and Polk Counties, Florida (Franze et al. 2000).

Monthly visual encounter surveys, larval amphibian sampling, and casual observations (aural and visual) will commence in the same BLM and UP locations sampled during the baseline period within one year of reestablishing pre-channelization floodplain hydroperiods, and continue for a period of three years. Visual encounter surveys repeated at regular intervals (monthly) over several years and a variety of environmental conditions likely will detect a large percentage of total taxa present. We anticipate that the use of multiple sampling techniques will be sufficient to document changes in species composition, species richness, and relative abundance within restored wetlands, and that these changes will be useful indicators of restoration success.

<u>Expectation:</u> Use of floodplain for amphibian reproduction and larval development

Adult amphibians should respond quickly to restored hydrologic patterns and increased plant community heterogeneity within restored marshes, and are likely to begin breeding shortly after colonizing. Because amphibian breeding activity in subtropical climates may occur during most of the year (Stebbins and Cohen 1995), larval amphibians are likely to be present year-round. However, because reference conditions documenting amphibian breeding periods are not available for the pre-channelized Kissimmee River, our expectation for the presence of larval amphibians is based on the occurrence of larval amphibians in remnant but altered BLM habitat during the baseline study period. During this period, when water was present on the floodplain, larval amphibians were collected a maximum of seven months during either year in Pool A or C remnant marsh. Assuming that a restored marsh will support larval amphibians at least as often as remnant marsh, a conservative estimate predicts the presence of larval amphibians for at least seven of 12 months in restored marshes in Pool C (Koebel 2005b).

The presence of larval amphibians will be determined from replicate throwtrap samples collected in the same BLM and UP habitats sampled during the baseline period in Pools A and C. Table 7-8 lists amphibians likely to use floodplain habitats for reproduction within the restored system, and typical breeding periods.

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CHAPTER 13

STATUS OF FISH ASSEMBLAGES OF THE KISSIMMEE RIVER PRIOR TO RESTORATION: BASELINE CONDITIONS AND EXPECTATIONS FOR RESTORATION

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ABSTRACT: Fish surveys addressing multiple metrics were conducted within severely altered habitats of the Kissimmee River following channelization. Attributes of baseline fish assemblages were compared to pre-channelization assemblages, where data were available, to determine if channelization-related impacts have occurred. Comparisons indicate that floodplain and river channel fish assemblage structure has shifted and that respective assemblages are dominated by taxa or guilds more characteristic of lentic and/or degraded conditions. Fishing effort for largemouth bass *Micropterus salmoides* has decreased by approximately 30% and catch rates for sport fishes are varied. Expectations for restoration-related change in specific fish assemblage metrics were developed to evaluate restoration success. Floodplain fish assemblages are characterized by guild according to macrohabitat use and, based on reference data, are expected to be dominated by off-channel dependent taxa in the restored system. Mean annual density of small fishes (< 10 cm total length) in floodplain habitats is expected to be greater than 18 fish/m^2. The expectation for river channel fish assemblages describes changes in the mean annual relative abundance of specific taxa and families and predicts that less than 1% bowfin *Amia calva* and 3% Florida gar *Lepisosteus platyrhincus*, greater than 16% redbreast sunfish *Lepomis auritus*, and greater than 58% centrarchids will be present in the post-restoration assemblage.

INTRODUCTION

Fishes are ecologically important components of large river-floodplain ecosystems (Welcomme 1979). Fish taxa representing a range of trophic categories (herbivore, piscivore, omnivore, invertivore, planktivore, detritivore) consume foods from aquatic and terrestrial environments (Karr et al. 1986) and serve as a critical link in the energy pathway between primary producers and higher trophic level consumers, including amphibians, reptiles, and birds (Karr et al. 1991, Gerking 1994). Fishes are used often as bioassays for contaminants within aquatic environments (Sprague 1973, USEPA 1977). Because freshwater fishes are relatively long-lived (Carlander 1977) and can travel considerable distances within their watershed (Gent et al. 1995, Furse et al. 1996), they integrate aspects of aquatic ecosystems across broad temporal and spatial scales (Karr et al. 1986). Fishes are therefore useful indicators of aquatic ecosystem health or integrity (Karr et al. 1986, Ohio EPA 1987, Oberdorf and Hughes 1992, Gammon and

Simon 2000). For these reasons, fishes were chosen as a biotic component of the Kissimmee River Restoration Evaluation Program.

Channelization of the Kissimmee River through the construction of the C-38 canal in 1962-1971 dramatically altered the hydrology of the system and resulted in drainage or obliteration of approximately 8,000 ha of floodplain wetlands, elimination of instream and overbank flow, and isolation of the river from its floodplain (Koebel 1995). These hydrologic alterations propagated changes in physical, chemical, functional, and biological aspects of the ecosystem that influence fish assemblages. These characteristics include depressed levels of dissolved oxygen, re-structuring of the food web, and habitat loss or degradation (Welcomme 1979, Junk et al. 1989, Gladden and Smock 1990).

Restoration of pre-channelization hydrologic characteristics through the Kissimmee River Restoration Project is expected to restore the physical habitat template, as well as reestablish chemical and functional attributes of the ecosystem that influence fish assemblages. Reestablishment of the pre-channelization river channel/floodplain linkage is critical for restoring food web pathways through transport of fish prey and organic inputs to the river channel and for providing essential nesting, nursery and foraging habitat. Reintroduction of flow is projected to alleviate seasonally low levels of dissolved oxygen and increase heterogeneity of in-channel microhabitat. Fish assemblages are expected to respond favorably to restored conditions and should approximate pre-channelization conditions or those of natural systems within the region (Trexeler 1995).

Objectives

The objectives of this study are: (1) to assess the baseline condition of floodplain and river channel fish assemblage structure, fish reproductive effort and larval fish assemblage structure, fish diets from nine taxa representing a range of trophic levels, angling effort and catch rate for specific sport fish taxa, largemouth bass and bluegill movement patterns, and methylmercury bioaccumulation in largemouth bass, (2) to estimate the reference condition of floodplain and river channel fish assemblage structure and angling effort and catch rate, (3) to quantify impacts of channelization by comparison of estimated pre-channelization and baseline conditions for floodplain and river channel fish assemblage structure and angling effort and catch rate, and (4) define and discuss specific expectations for selected attributes of floodplain and river channel fish assemblage structure.

STUDIES WITH ASSOCIATED RESTORATION EXPECTATIONS

I. FLOODPLAIN FISH ASSEMBLAGE STRUCTURE

Baseline Condition

Methods

Floodplain fishes were sampled with a 1-m^3 aluminum throw trap, which provides accurate estimates of density, size structure, and relative abundance of small-fish (<10 cm total length) populations within heavily vegetated habitats (Kushlan 1981, Chick et al. 1992, Jordan et al. 1997). Sampling was conducted quarterly between August 1996 and April 1997, and monthly from August 1997 through January 1999. Two habitat units (one Control unit and one Impact unit) were sampled in three vegetation types each, which included Broadleaf Marsh (BLM, Bousquin and Carnal 2005), Woody Shrub (*Myrica cerifera* Floating Mat Shrubland Bcode group; S.MCF), and Pasture (Upland Herbaceous Bcode group; UP) (Figure 13-1). Ten replicate samples were collected in randomly selected locations in each habitat on each sampling date. Following trap placement, all vegetation within the trap was removed.

Water depth was recorded at each corner and at the center of the trap. All vegetation within the trap was removed, and fishes were removed with a dip-net (1-mm mesh). Dip-netting continued until no fish were collected in 10 consecutive attempts. All fishes were preserved in 10% buffered formalin. In the laboratory, all fishes were identified to species, counted, and measured to the nearest mm (total length).

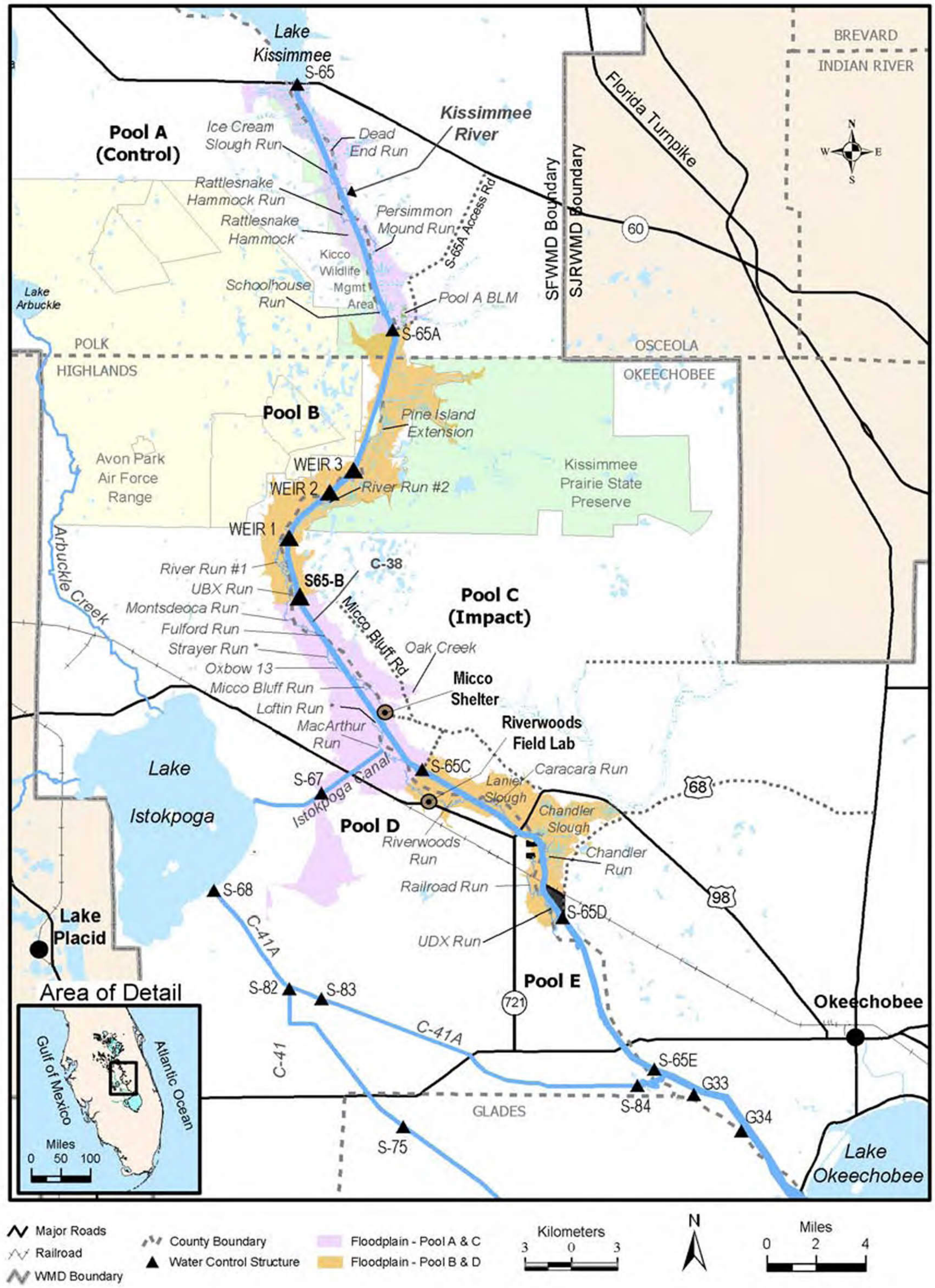


Figure 13-1. Map of the Kissimmee River showing the locations of pools and remnant river runs where fish were sampled.

Two metrics were used to develop restoration expectations for floodplain fish assemblages — relative abundance according to macrohabitat guild and fish density (number of fish/m^2). The macrohabitat guild structure developed by Bain (1992) used to assess guild relative abundance was augmented to include two new guild categories based on fish dependence on off-channel habitats (Figure 13-2).

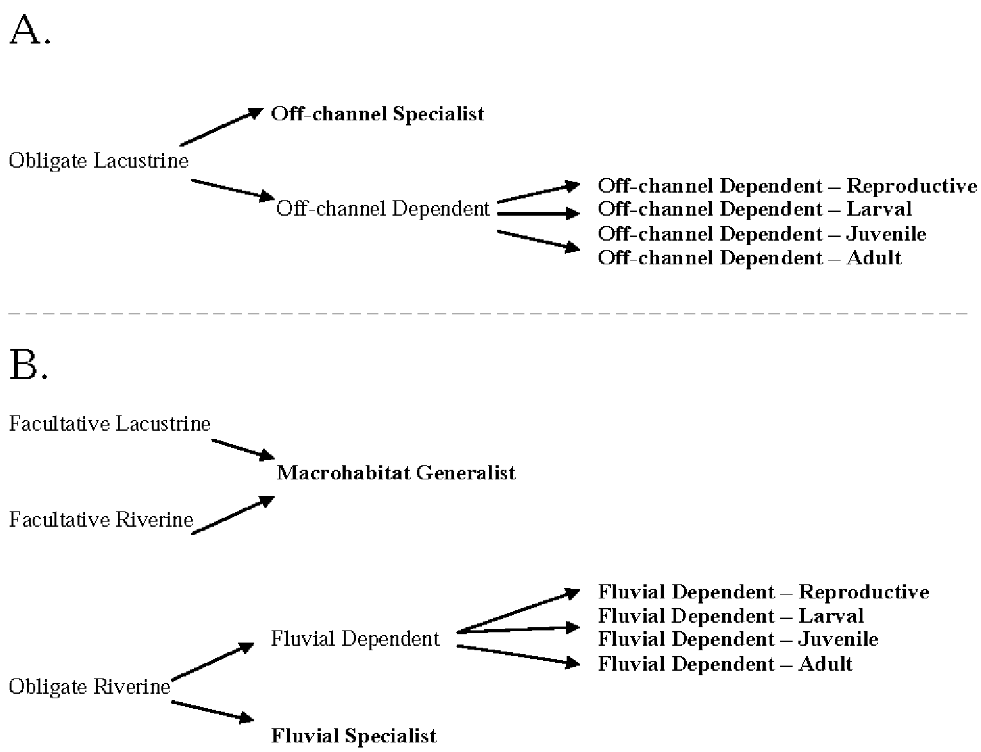


Figure 13-2. Schematic representation of modified macrohabitat guild structure derived by Bain (1992). (A) New guild categories based on dependence of associated taxa on off-channel habitat. The new category termed off-channel dependent includes species that are found in a variety of habitats, but require access or use of off-channel habitats, or are limited to nonflowing, vegetated waters at some point in their life cycle. These species may have significant riverine populations during particular life history stages. The off-channel specialist category refers to species that are almost always found only in off-channel habitats or species that are limited to non-flowing, vegetated habitats throughout life. Occasionally, individuals may be found in the river channel, but the vast majority of information on these fishes pertains to off-channel habitat. (B) Original macrohabitat guild classification developed by Bain (1992).

The new guild categories were constructed based on habitat required for reproduction according to Balon (1975), general habitat use listed by Lee et al. (1980), Eenier and Starnes (1993), and Mettee et al. (1996), and from results of a literature review (Appendix 13-1A) conducted to identify off-channel habitat use by Kissimmee River fishes and their life-history stage(s). All terms follow Bain (1992), with the addition of "off-channel" (of, or related to, any habitat not included in the open water portion of the river channel). These areas include river channel littoral vegetation and any floodplain habitat. Guild relative abundance is defined as the proportion of individuals of guild *i* in relation to the total number of individuals recorded (Bain 1992).

Mean annual fish density was calculated for each habitat by first calculating a sample mean for each month by averaging the ten monthly replicates for each habitat, and then calculating a monthly mean by averaging sample means. Finally a mean annual value was determined by averaging monthly means for each year of study. Mean annual fish density was compared among habitats using ANOVA (SAS Institute 1990). Relationship between mean monthly density and water depth was tested using Linear Regression.

Results

The augmented macrohabit guild structure classified fish taxa known to occur in the Kissimmee River as follows: 29% off-channel specialist, 52% off-channel dependent, 10% habitat generalist, 0% fluvial dependent, and 8% fluvial specialist (Table 13-1).

A total of 3159 fishes representing ten species, six families, and three guilds were collected from floodplain habitats during the baseline (1996–1999) survey (Table 13-2).

Table 13-1. Macrohabitat guild classification of fishes occurring in the Kissimmee River. The off-channel dependent guild includes classification according to dependence on off-channel habitat for reproduction (R) or by life history stage (larval - L or juvenile - J).

Scientific name	Common name	Off-channel specialist	Off-channel dependent	Habitat generalist	Fluvial dependent	Fluvial specialist
Amia calva	bowfin	X				
Esox americanus	redfin pickerel	X				
Esox niger	chain pickerel	X				
Ameiurus natalis	yellow bullhead	X				
Ameiurus nebulosus	brown bullhead	X				
Noturus gyrinus	tadpole madtom	X				
Aphredoderus sayanus	pirate perch	X				
Jordanella floridae	flagfish	X				
Lucania goodei	bluefin killifish	X				
Gambusia holbrooki	mosquitofish	X				
Heterandria formosa	least killifish	X				
Poecilia latipinna	sailfin molly	X				
Elassoma evergladei	Everglades pygmy sunfish	X				
Elassoma okefenokee	Okefenokee pygmy sunfish	X				
Enneacanthus gloriosus	bluespotted sunfish	X				
Lepisosteus osseus	longnose gar		R			
Lepisosteus platyrhincus	Florida gar		R			
Dorosoma cepedianum	gizzard shad		L			
Dorasoma petenense	threadfin shad		J			
Cyprinus carpio	common carp		R			
Ctenopharyngodon idella	grass carp		R			
Notemigonus crysoleucas	golden shiner		L			
Notropis maculatus	taillight shiner		R, L, J			
Notropis petersoni	coastal shiner		J			
Opsopoedus emiliae	pugnose minnow		J			
Erimyzon sucetta	lake chubsucker		J			
Ameiurus catus	white catfish		R			
Ictalurus punctatus	channel catfish		R			
Clarius batrachus	walking catfish		R			
Hoplosternum littorale	brown hoplo		R, L, J			
Fundulus seminolis	Seminole killifish		J			
Labidesthes sicculus	brook silverside		L			
Lepomis auritrus	redbreast sunfish		R, L, J			
Lepomis gulosus	warmouth		R, L, J			
Lepomis machrochirus	bluegill		R, L, J			
Lepomis microlophus	redear sunfish		R, L, J			
Lepomis punctatus	spotted sunfish		R, L, J			
Micropterus salmoides	largemouth bass		R, L, J			
Pomoxis nigromaculatus	black crappie		R, L, J			
Astronotus ocellatus	oscar		J			
Oreochromis aureus	blue tilapia		R			
Fundulus chrysostus	golden topminnow			X		
Fundulus lineotus	lined topminnow			X		
Fundulus rubifrons	redface topminnow			X		
Menidia beryllina	tidewater silverside			X		
Etheostoma fusiforme	swamp darter			X		
Anguilla rostrata	American eel					X
Strongylura marina	Atlantic needlefish					X
Percina nigrofasciata	blackbanded darter					X
Mugil cephalus	stripped mullet					X
Pterygoplichthys disjunctivus	sailfin catfish					

Table 13-2. Fishes collected from Kissimmee River floodplain habitats in a 1957 survey (FGFWFC 1957) and during the baseline period between 1996 and 1999. Habitats sampled included Broadleaf Marsh (BLM), Woody Shrub (S.CMF) and Pasture (UP). (Ψ denotes off-channel specialist taxa, Φ denotes off-channel dependent taxa, and Λ denotes habitat generalist taxa).

Species	1957	Number collected 1996-1999 BLM		S.CMF		UP	
		Site 1	Site 2	Site 1	Site 2	Site 1	Site 2
Esocidae							
Ψ Redfin pickerel *Esox americanus*	5						
Cyprinidae							
Φ Golden shiner *Notemigonus crysoleucas*	363						
Φ Tailight shiner *Notropis maculatus*	96						
Φ Coastal shiner *Notropis petersoni*	2						
Catostomidae							
Φ Lake chubsucker *Erimyzon sucetta*	13						
Ictaluridae							
Φ White catfish *Ameiurus catus*	2						
Ψ Brown bullhead *Ameiurus nebulosus*	1						
Φ Channel catfish *Ictalurus punctatus*	1						
Ψ Tadpole madtom *Noturus gyrinus*	18						
Clariidae							
Φ Walking catfish *Clarias batrachus*			2				
Aphredoderidae							
Φ Pirate perch *Aphredoderus sayanus*	1						
Fundulidae							
Λ Golden topminnow *Fundulus chrysotus*	6			12	13		
Ψ Bluefin killifish *Lucania goodei*	15	1					
Poeciliidae							
Ψ Eastern mosquitofish *Gambusia holbrooki*	14	50	120	123	263	3	5
Ψ Least killifish *Heterandria formosa*	3	83	47	468	712	13	1
Atherinidae							
Φ Brook silverside *Labidesthes sicculus*	12			1	29		
Elassomatidae							
Ψ Everglades pygmy sunfish *Elassoma evergladei*	7	304	226	361	94	16	16
Ψ Okefenokee pygmy sunfish *Elassoma okefenokee*		64	12	70	44	3	
Centrarchidae							
Φ Bluespotted sunfish *Enneacanthus gloriosus*	28	1	1				
Φ Redbreast sunfish *Lepomis auritus*	298						
Φ Warmouth *Lepomis gulosus*	7						
Φ Bluegill *Lepomis machrochirus*	1				1		
Φ Redear sunfish *Lepomis microlophus*	9						
Φ Largemouth bass *Micropterus salmoides*	8						
Φ Black crappie *Pomoxis nigromaculatus*	1						
Percidae							
Λ Swamp darter *Etheostoma fusiforme*	11						
Total	922	503	408	1035	1156	35	22

All fishes, except three individuals (bluegill *Lepomis macrochirus* and walking catfish *Clarias batrachus*), were small-bodied fishes. Large-bodied fishes were collected only during the wet season. Distribution of taxa according to guild included five off-channel specialists (50%), four off-channel dependents (40%), and one habitat generalist (10%) (Table 13-2). The assemblage was dominated in abundance by off-channel specialists (98%), especially least killifish *Heterandria formosa* (42%), Everglades pygmy sunfish *Elassoma evergladei* (32%), and eastern mosquitofish *Gambusia holbrooki* (18%) (Table 13-2). The remainder of the assemblage was comprised of off-channel dependents (1%) and generalists (1%) (Table 13-2). Only a single immature, large-bodied off-channel dependent (bluegill) individual was collected. Guild composition was similar among sampling periods for each habitat over the period of study and was dominated by off-channel specialist (Figure 13-3).

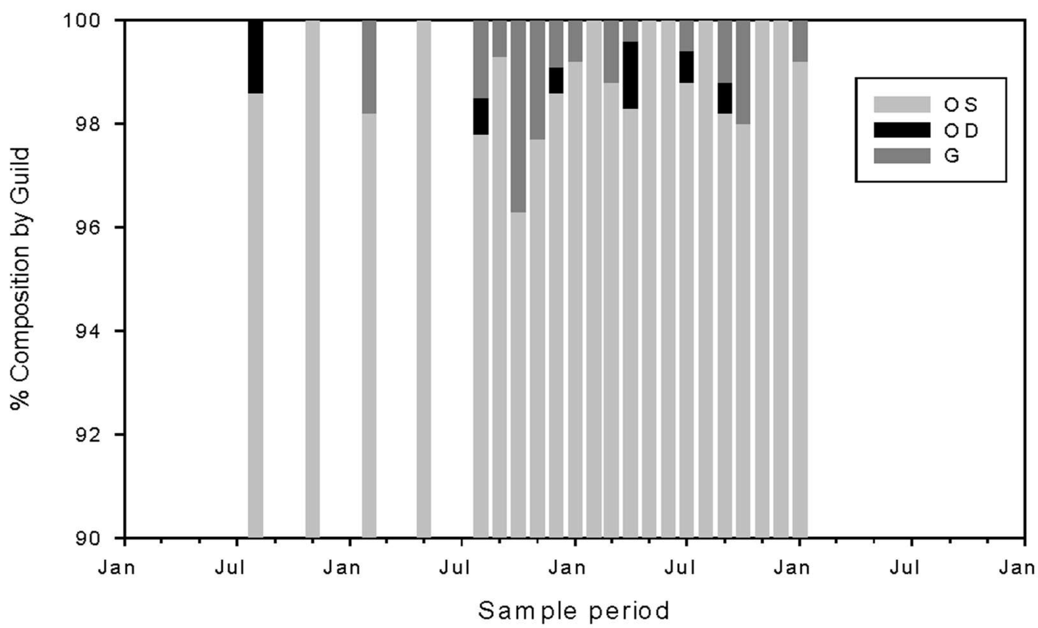


Figure 13-3. Percent composition of fishes collected in floodplain habitats by macrohabitat guild for each sampling period during the baseline survey (1996–1999). Guilds include off-channel specialist (OS), off-channel dependent (OD), and habitat generalist (G).

Mean annual density was low in all habitats (Table 13-3). Mean annual density was highest in S.CMF habitats (3.93–5.35 fish/m^2) and did not differ significantly (ANOVA; $p = 0.6314$) between pools. Broadleaf Marsh had lower mean annual densities (1.49–1.70 fish/m^2), which also were not significantly different between Control and Impact sites (ANOVA; $p = 0.9123$). Mean annual densities were lowest within UP sites (not exceeding 0.30 fish/m^2 for either pool) and were not significantly different between Control and Impact sites (ANOVA; $p = 0.7457$).

Regression analysis showed a weak, but not significant, relationship between monthly fish density and water depth at BLM sites (Figure 13-4; Pool A $R^2 = 0.21$, Pool C $R^2 = 0.18$). This relationship was stronger, but not significant, at S.CMF sites (Figure 13-5; Pool C $R^2 = 0.37$, Pool D $R^2 = 0.45$).

Reference Condition

Methods

Between 1956 and 1957 the Florida Game and Fresh Water Fish Commission (FGFWFC) sampled fish assemblages of the pre-channelized Kissimmee River to provide consideration and guidance to the Army Corps of Engineers (ACOE) for the planned channelization of the river. The sampling method employed and habitat characteristics of the sample area are unclear. Fishes were collected from a single 0.1 ha sample of floodplain marsh to which rotenone was applied. Water depths in the sample area ranged from "shallow" to 1.0 m (FGFWFC 1957). Sampling was conducted in June 1957, one year following an extreme drought. Floodplain fish assemblage structure was described by guild relative abundance according to criteria outlined under baseline conditions.

Table 13-3. Mean (± SE) annual density (number of fish/m^2) of fishes collected from Broadleaf Marsh (BLM) and Woody Shrub (S.CMF) habitats at Control and Impact sites during baseline sampling. Density values for Pasture (UP) habitat are monthly sample means because data were collected only over a single year.

Habitat	Control	Impact
BLM	1.7 ± 1.5	1.5 ± 1.1
S.CMF	3.9 ± 2.5	5.4 ± 1.1
UP	0.3 ± 0.3	0.2 ± 0.2

Fish density data for marshes of south and central Florida were compiled and summarized from published papers, theses, technical reports, and unpublished data (Jordan 1999). A total of 5314 independent samples were synthesized strictly from enclosure methods with clearly defined sampling areas capable of providing quantitative density estimates. Sample locations included marshes of the Everglades, marshes associated with lakes (including Lake Okeechobee) and canals, and marshes associated with rivers (including the upper St. Johns River). Sample methods included throw traps, Wegner rings, and block nets. Habitat types at sample locations were defined according to dominant vegetation taxa present and only data for marshes characterized by emergents (i.e., *Pontedaria* sp., *Sagittaria* sp., *Peltandra* sp.) were included for deriving the reference condition for Kissimmee River marshes. Mean fish density was calculated by averaging sample density across studies.

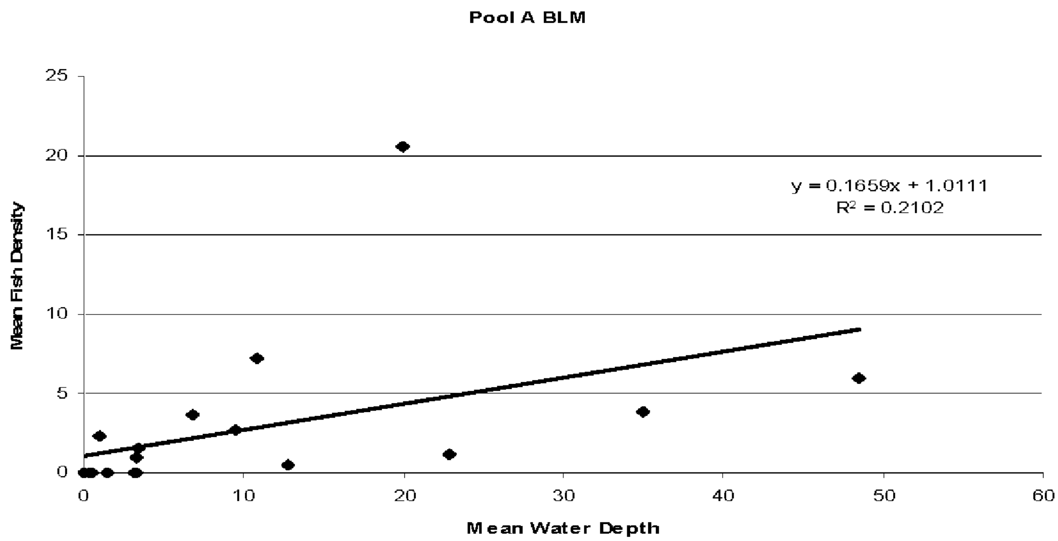
Results

The Florida Game and Freshwater Fish Commission (1957) collected 922 individual fish representing 24 taxa, 11 families, and three guilds (Table 13-2). This assemblage included large (adults >80 mm SL) and small-bodied fishes. Distribution of taxa according to guild included seven off-channel specialists (29.1%), 15 off-channel dependents (62.5%), and two habitat generalists (8.3%). The assemblage was dominated in abundance by off-channel dependents (88.1%), especially golden shiner *Notemigonus crysoleucas* (39%) and redbreast sunfish *Lepomis auritus* (32%) (Table 13-2). The remainder of the assemblage was comprised of off-channel specialists (10.1 %) and habitat generalists (1.8%) (Table 13-2). Of the 812 off-channel dependents collected, 39.7% were juvenile or young of the year centrarchids and esocids. Mean density of fishes in emergent marshes of south and central Florida was 23.4 (± 0.9) fish/m^2.

Discussion

Although collection methods and sample sizes differed between surveys, it is clear that dramatic changes have occurred in fish use of floodplain habitats since channelization. Approximately 60% of all species documented in the Kissimmee River during the pre-channelization survey (FGFWFC 1957) were found to use floodplain habitats, which is supported by previous studies indicating facultative use of floodplain habitats by a majority of fish taxa in river-floodplain systems (Guillory 1979, Welcomme 1979, Kwak 1988, Bayley et al. 1991, Leitman et al. 1991). Timing, depth, and duration of flood events are the critical factors regulating fish use of floodplain habitats. Results of pre-channelization surveys indicate that hydrologic conditions on the floodplain were capable of supporting a large proportion of taxa inhabiting the river-floodplain system. Also, the pre-channelization assemblage comprised both juvenile and adults of off-channel dependent taxa, implicating the floodplain's function as a nursery area.

The augmented macrohabitat guild structure reclassifies 41 taxa (82%) that would have been categorized as habitat generalist to either off-channel dependent or off-channel specialist (Table 13-1), thereby illustrating the importance of off-channel habitat availability to Kissimmee River fishes. However, fish assemblages of the channelized floodplain were dominated exclusively by small-bodied, off-channel specialist taxa. These fishes typically are not limited by minimal inundation depths, and were able to exploit floodplain habitats year-round. Large-bodied individuals, including juvenile and especially adult off-channel dependent taxa, would not be expected within floodplain habitats when depths are less than 50 cm, a depth generally required for immigration of large-bodied fishes from the river channel to the floodplain (F. Jordan, Jacksonville University, personal communication). During the baseline sampling period, mean monthly water depths on the floodplain exceeded 50 cm only once (February 1998 - Pool A BLM).



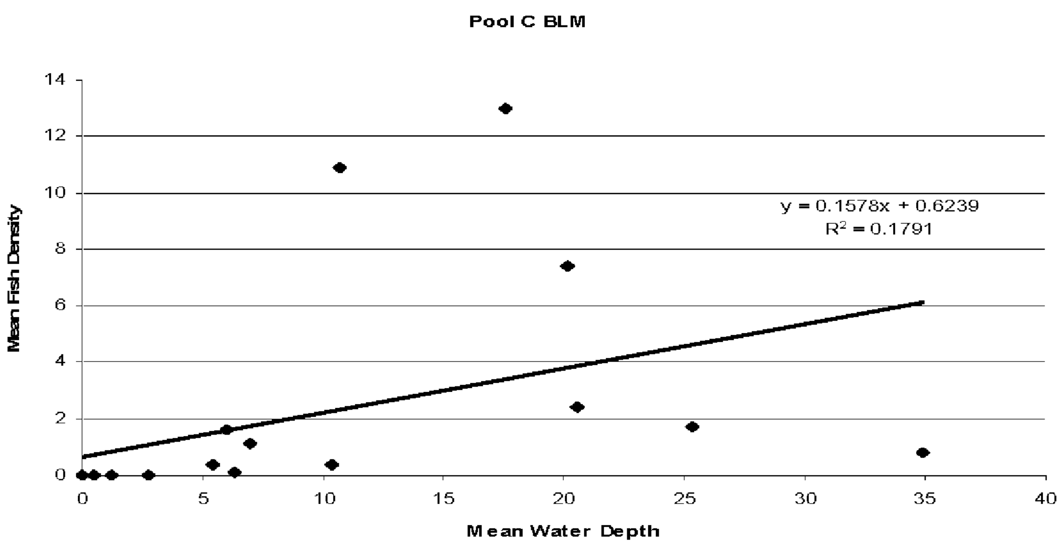
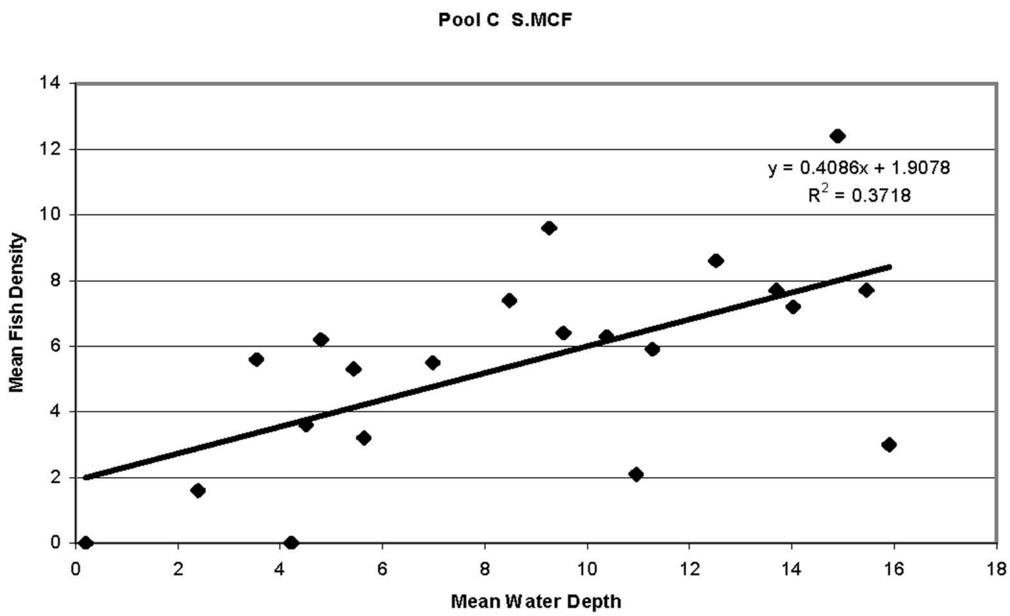


Figure 13-4. Relationship between mean monthly fish density and mean monthly water depth at Broadleaf Marsh (BLM) sites during the baseline period.

Although members of the off-channel dependent guild require access to off-channel habitat during a particular life history stage, most are also capable of using these habitats during non-dependent life history stages when conditions are favorable (Lee et al. 1980) (Appendix A). Bayley (1991) argues that species capable of using inundated floodplains benefit from increased production associated with a moving littoral zone and gain a competitive advantage (i.e., flood-pulse advantage) over taxa that cannot. Facultative fish use of floodplains is common in unaltered river systems (Welcomme 1979, Leitman et al. 1991), due in part to the temporal availability of floodplain habitats and resources associated with climatic cycles (e.g., wet and dry seasons), and is believed to have occurred frequently in the pre-channelization Kissimmee River, due to protracted floodplain inundation. Results of the baseline study suggest that the habitat requirements necessary to support off-channel dependent taxa were not present under channelized conditions.



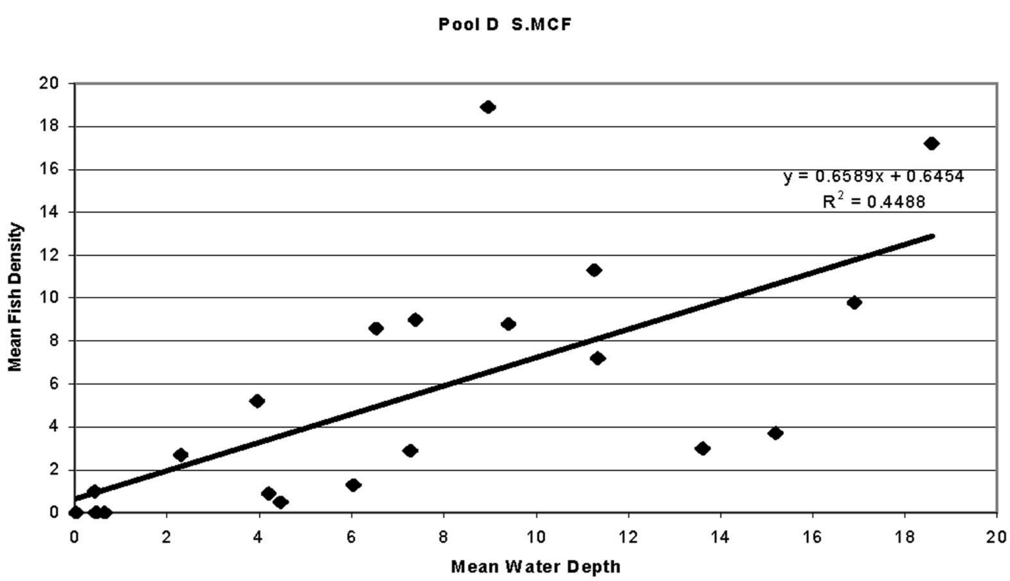


Figure 13-5. Relationship between mean monthly fish density and mean monthly water depth at Woody Shrub (S.CMF) sites during the baseline period.

The shift in numerical dominance from off-channel dependent taxa (88%) under pre-channelization conditions to dominance by off-channel specialist taxa (98%) under channelized conditions, coincides with loss of the seasonal flood pulse and associated floodplain accessibility. Even though the single sample from the 1957 survey depicts floodplain fish community structure as only a snapshot in time, it is believed to accurately portray, at a minimum, seasonal use by off-channel dependent taxa. No seasonal change in guild composition was indicated from monthly sampling over two years in the latter survey (Figure 13-3). Fishes that dominate biomass and production in river–floodplain systems depend on periodically inundated floodplain habitats for reproduction (Shaeffer and Nickum 1986, Copp 1989), foraging (Gladden and Smock 1990), and refugia (Savino and Stein 1982, Welcomme 1985) at some life history stage, unlike off-channel specialist, which are able to complete their entire life history on the floodplain. Pre-channelization data indicate that 37% of off-channel dependent fishes collected were juvenile or young-of-the-year (YOY) centrarchids, which are the dominant taxa in most peninsular Florida rivers (Bass and Cox 1985). The results suggest that the hypothesized nursery function afforded to centrarchids, which are off-channel dependent, in the pre-channelized system was compromised due to channelization, as only a single immature centrarchid was collected under channelized conditions.

Although off-channel dependent taxa were represented by only one individual in the floodplain, members of this guild were abundant in remnant river channels (see Section II below). Several factors may account for the limited use of floodplain habitats by immature off-channel dependent taxa under channelized conditions: (1) adult access to floodplain habitats for spawning was limited by inundation depth or dense vegetation; therefore, these species were restricted to littoral habitats within the river channel; (2) floodplain habitats under the baseline condition do not receive a seasonal flood-pulse due to hydrologic regulation of the system and therefore the cue for initiating lateral migration is absent; or (3) elimination of flow and resulting increased coverage of littoral vegetation in remnant river channels (Bousquin 2005) provided the necessary habitat structure within remnant channels.

The observed shift in numerical dominance by off-channel specialists, especially poeceiliids (59%) and elassomatids (38%), in floodplain fish assemblages also may indicate decline in floodplain macrohabitat quality. Members of this guild are capable of completing their entire life cycle in non-flowing environments and often possess adaptations for harsh conditions that may occur in altered floodplain habitats. Poeciliids and elassomatids dominant in channelized floodplain habitats are tolerant of protracted shallow inundation depths and of low levels of dissolved oxygen, and can exist in highly degraded habitats (Meffee and Snelson 1989). Poeciliids often remain dominant under these conditions due to the high reproduction rates associated with their reproductive mode (live bearer) (Meffee and Snelson 1989).

Additionally, degraded floodplain habitats within the channelized system likely lack the heterogeneity required to support diverse fish communities (MacArthur and Wilson 1967, Trexler 1995). The principle factors affecting habitat heterogeneity within floodplain habitats are hydroperiod, inundation depth, areal extent of inundation, and macrophyte and emergent vegetation type and density (Lowe 1986, Copp 1989, Chick and McIvor 1997). These factors create niches capable of supporting greater numbers of species than can be supported in more homogenous habitats within the channelized system.

Expectations

Restoration of the physical form and pre-channelization hydrology of the Kissimmee River is expected to reestablish ecological integrity to over 100 km^2 of river-floodplain ecosystem (Toth 1993). Floodplain fish assemblage composition is expected to shift and more closely resemble that occurring before channelization, notably with the off-channel dependent guild reestablishing dominance. Potential evidence for this shift is illustrated by increased use of "enhanced" floodplain habitat in Pool B of the Kissimmee River by off-channel dependent taxa. Hydroperiod and inundation depths in floodplain habitats at the southern end of Pool B have been enhanced by the Demonstration Project (Toth 1993). Limited throw trap sampling (n=10 samples) of BLM within this area produced juveniles of two off-channel dependent taxa (bluegill and warmouth), which comprised approximately 8% of the total number of fishes collected. These results suggest that floodplain use by juvenile centrarchids and other large-bodied off-channel dependent species is likely to increase following restoration of pre-channelization hydrologic conditions. Increases in floodplain use will result from reproduction and population expansion by resident fishes, lateral migrations of small and large-bodied riverine fishes during periods of overbank flow (flood pulse), and from increased areal coverage of both temporary and permanently inundated floodplain habitat. Concurrent increases in primary and secondary production within floodplain habitats will provide the necessary food base to support increased fish populations.

Expectation for floodplain fish assemblages. Applying guilds to biotic community data has been found to simplify analyses and predictions of community change (Austen et al. 1994). The benefit of using guilds rather than individual indicator taxa to indicate environmental change is that guilds function as a "super-species" (Austen et al. 1994) that uses a particular resource similarly. The presence of one or more guild members is indicative that at least a minimal amount of the resource in question is available (Austen et al. 1994). If the dramatic decline in floodplain use by members of the off-channel dependent guild depicts elimination of floodplain connectivity or degradation of floodplain habitat quality, then the expected increase in floodplain use by the same guild infers reestablishment of that resource, especially if the magnitude of change in use is great. The expectation for floodplain fish assemblages states that following restoration, the off-channel dependent guild will constitute >50% of the assemblage and will be comprised of ≥12 taxa. Young-of-the-year or juveniles will comprise ≥30% of the off-channel dependent guild. Figure 13-6 shows pre-channelization and baseline values of percent composition and number of taxa of off-channel dependent guild members in floodplain habitats. Dashed line indicates expected value for each metric following restoration.

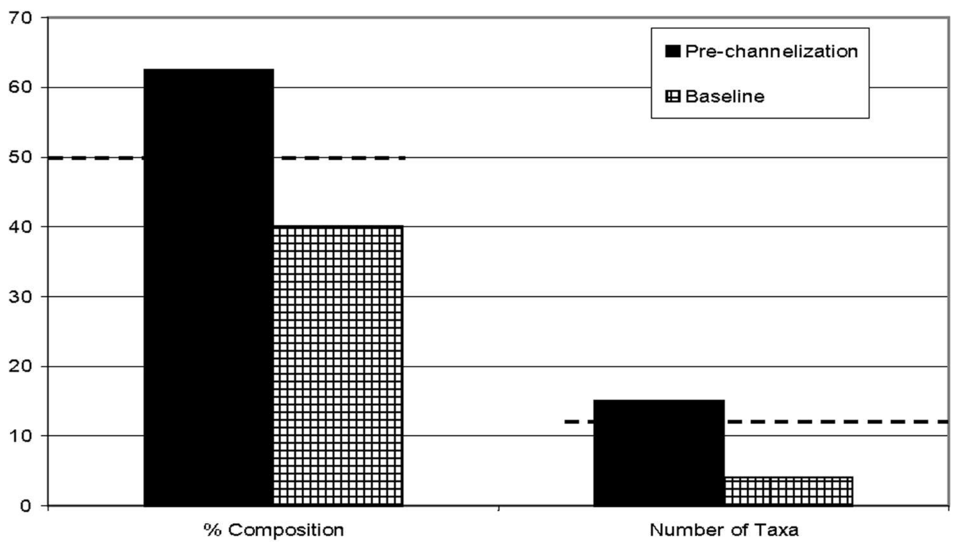


Figure 13-6. Baseline percent composition and number of taxa of off-channel dependent guild members in floodplain fish assemblages of the Kissimmee River. Dashed line indicates expected value for each metric following restoration.

All success criteria for expectation metrics of guild relative abundance are approximately 80% of historic values for the 1957 GFC sample (12 taxa and 50% relative abundance). Although conservative, these expected values account for the natural variability of floodplain fish assemblages, potential use of the floodplain by non-indigenous taxa that were introduced since channelization, and limited quantity of historic data on which the expectation is based.

Expectation for floodplain fish density. Mean annual fish density within floodplain habitats was low (<5.4 fishes/m^2 in all sampled habitats; range 0.2–5.35 fishes/m^2) during the baseline period. Fish density within floodplain habitats is related to prey availability, composition of predator assemblages, heterogeneity of floodplain vegetation, areal coverage of floodplain inundation, and depth and duration of floodplain inundation (Welcomme 1979, Lowe 1986, Heck and Crowder 1991, Connolly 1994, Loftus and Ekland 1994, Jordan et al. 1996, 1998). Fish density is expected to increase following restoration through reestablishment of these features, but is projected to fluctuate with inundation patterns. Fish densities within restored floodplain habitats are likely to be greater during periods of floodplain recession, due to concentration within topographic depressions scattered throughout the landscape. Although baseline sampling results indicate mean fish density was greater during the dry season, this increase likely was attributable to uncharacteristic floodplain inundation patterns associated with the 1997–1998 El Niño event. At S.CMF sites, mean monthly density increased with water depth. The expectation for density of fish in inundated floodplain habitats states that mean annual density of small fishes (fishes <10 cm total length) within restored BLM habitats will be ≥ 18 fish/m^2 (Figure 13-7).

The success criterion for the expectation metric of fish density is approximately 80% of the reference value for freshwater marshes of central and south Florida. Although conservative, these expected values account for the natural variability of floodplain fish assemblages and limited quantity of historic data on which the expectation is based.

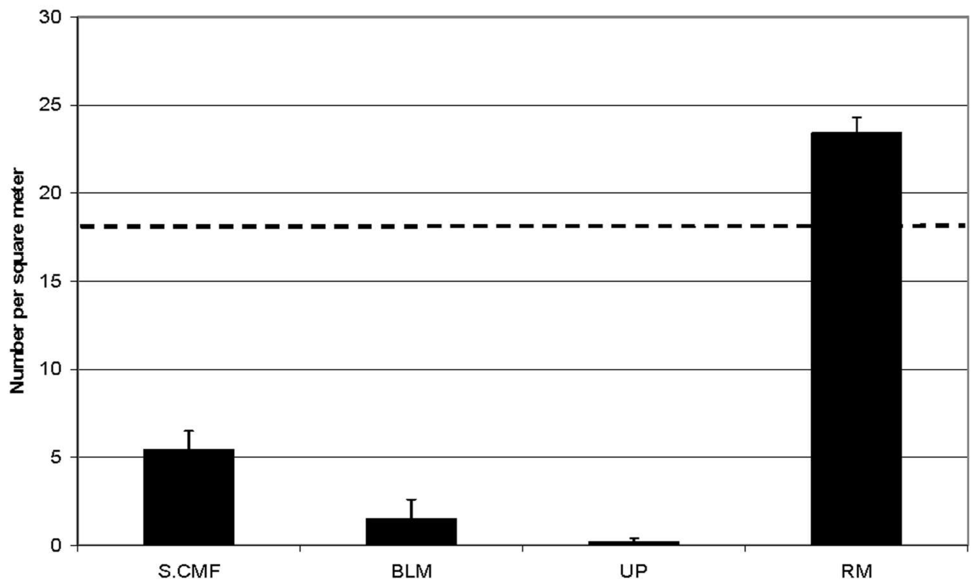


Figure 13-7. Mean density of fishes collected from Broadleaf Marsh (BLM), Woody Shrub (S.CMF), and Pasture (UP) habitats of the Kissimmee River under baseline conditions and from reference marshes (RM) of south and central Florida. Dashed line indicates expected value of small fishes within floodplain marshes following restoration.

Expectation evaluation

Throw trap sampling will be used to evaluate post-restoration floodplain fish assemblages at the same locations as baseline sampling. Sampling will begin immediately following inundation of floodplain habitats associated with implementing the Final Headwaters Regulation Schedule. Methods will be identical to those utilized for baseline studies, including monthly collection of ten random samples in each habitat. Sampling will be conducted in three-year periods beginning on the first and sixth years following implementation of the Final Headwaters Regulation Schedule.

Samples will be analyzed for composition, age class, and relative abundance of small- and large-bodied taxa according to macrohabitat guild. These metrics will document restoration of river channel-floodplain exchange and use of floodplain habitats as spawning and nursery grounds. Age classes of centrarchids and esocids will be based on total body length (Table 13-4). Mean annual relative abundance for all taxa will be based on each three-year block of post-restoration data. Annual means will be derived by averaging monthly relative abundance, generated from total numbers pooled from ten replicates each month. Seasonal effects (especially prolonged floodplain inundation during the wet season) on relative abundance are expected to be reflected in yearly means. Although this expectation is based on mean annual relative abundance, data also will be analyzed by season to evaluate the potential significance of seasonality.

II. RIVER CHANNEL FISH ASSEMBLAGE STRUCTURE

Baseline Condition

Methods

River channel fish communities inhabiting areas within and adjacent to littoral vegetation were sampled annually in June between 1992 and 1994 by the FGFWFC using electrofishing gear. Electrofishing adequately samples fish populations in shallow, vegetated habitats and does not alter community composition, as collected individuals are released alive following work-up. Sampling gear consisted of a 5.5 meter jon boat outfitted with a 5-kilowatt generator, Coffelt electrofishing unit (Model #VVP-15), and cable electrodes, with the boat serving as the anode. Pulsed AC current varied between 200–240 volts and 4–8 amperes. Triplicate 15 minute shocking episodes were conducted along fixed transects within C-38 and remnant river channel. Electrofishing was conducted in C-38 and three remnant river runs in Pools A (Ice Cream Slough Run, Persimmon Mound Run, and School House Run) and C (Montsdeoca Run, Micco Bluff Run, and MacArthur Run) (Figure 13-1). Sampling was conducted by two-person crews (one driver and one dip-netter) along the deep water edge of littoral vegetation as the boat traveled downstream. Fish were identified to species, counted, and weighed. All fishes except Florida gar *Lepisosteus platyrhincus* and bowfn *Amia calva* (due to difficulty in handling) were measured to the nearest millimeter. Body lengths for unmeasured gar and bowfin were derived from length-weight regressions generated from a subset of measured and weighed fishes.

Table 13-4. Body lengths for age class determination of centrarchid and esocid taxa in the Kissimmee River (modified from Carlander 1977 and Lee et al. 1980).

Taxa	Common Name	Young-of-the-year	Juvenile
Esox ameicanus	redfin pickerel	--	<250 mm
Esox niger	chain pickerel	--	<300 mm
Micropterus salmoides	largemouth bass	0–64 mm	65–120 mm
Lepomis auritrus	redbreast sunfish	0–35 mm	36–60 mm
Lepomis gulosis	warmouth	0–32 mm	33–75 mm
Lepomis machrochirus	bluegill	0–45mm	46–90 mm
Lepomis microlophus	redear sunfish	0–56 mm	57–134 mm
Lepomis punctatus	spotted sunfish	--	<55 mm (SL)
Pomoxis nigromaculatus	black crappie	0–51 mm	52–130 mm

Catch per unit effort (CPUE) was calculated for abundance data. Catch per unit effort is the number or weight of organisms captured within a defined unit of sampling or fishing effort (e.g., fish/min). Mean annual relative abundance was calculated as the average of replicate samples for each pool for each year. Mean annual CPUE for abundance was calculated similarly for individual taxa and centrarchids. Mean annual relative abundance CPUE was compared between years and sites using ANOVA (SAS Institute 1990) and associated means separation test.

Results

A total of 6247 fishes representing 32 species were collected by electrofishing (Table 13-5). Dominant species (>5% of mean annual relative abundance) at Control sites in Pool A included *L. platyrhincus* (36.8%), *L. macrochirus* (19.9%), *A. cavla* (8.4%), and *Micropterus salmoides* (7.9%) (Table 13-5). Assemblage composition at Impact sites (Pool C) was similarly dominated by *L. platyrhincus* (19.6%), *L. macrochirus* (16.5%), and *M. salmoides* (9.5%), but also included *G. holbrooki* (16.9%) and

Notemigonus crysoleucas (11.7%) (Table 13-5). Centrarchids accounted for only 31.8% and 38.3% of the fish assemblages in Pools A and C, respectively (Table 13-5). Centrarchid mean annual CPUE was significantly greater than that for lepisostids/amiids at canal sites in both Pools (ANOVA; $p < 0.05$ in all cases), however no difference occurred between groups at river channel sites in either pool (ANOVA; $p > 0.05$ in all cases).

Reference Conditions

Methods

Annual river channel fish sampling was conducted between 1983 and 1990 by FGFWFC in the lower St. Johns, Withlacoochee, and Oklawaha Rivers using electrofishing gear. Sampling was conducted in the lower St. Johns River between 1984 and 1988, in the Oklawaha River between 1983 and 1990, and in the Withlacootchee during 1983, 1985, 1986, and 1989. Sampling gear consisted of a 5.5 meter jon boat outfitted with a 5-kilowatt generator, Coffelt electrofishing unit (Model #VVP-15), and cable electrodes, with the boat serving as the anode. Pulsed AC current varied between 200–240 volts and 4–8 amperes. Duplicate 15 minute shocking episodes were conducted at fixed transects along each river. Four sites were sampled in the Oklawaha and lower St. Johns Rivers and six were sampled in the Withlacootchee. Sampling was conducted by two-person crews (one driver and one dip-netter) along the deep water edge of littoral vegetation as the boat traveled downstream. Fishes were identified to species, counted, and weighed.

Table 13-5. Mean ± SE annual relative abundance (percentage of total numbers) of fish species sampled during baseline conditions within remnant river channels of the Kissimmee River by electrofishing.

Species	Common Name	FGFWFC Electrofishing 1992-1994	
		Pool A	Pool C
Ameiurus natalis	yellow bullhead	--	0.5 ± 0.2
Ameiurus nebulosus	brown bullhead	0.07 ± 0.07	0.3 ± 0.1
Amia calva	bowfin	8.3 ± 2.5	4.4 ± 0.7
Clarias batrachus	walking catfish	0.4 ± 0.4	1.4 ± 0.4
Dorosoma cepedianum	gizzard shad	0.2 ± 0.2	--
Dorosoma petenense	threadfin shad	0.06 ± 0.06	--
Elassoma okeefenokei	Okeefenokee pygmy sunfish	--	0.1 ± 0.1
Ennecanthus gloriosus	bluespotted sunfish	0.1 ± 0.1	0.5 ± 0.2
Erimyzon sucetta	lake chubsucker	1.4 ± 0.5	3.9 ± 1.2
Esox niger	chain pickerel	0.3 ± 0.1	0.3 ± 0.1
Etheostoma fusiforme	swamp darter	--	0.1 ± 0.05
Fundulus chrysotus	golden topminnow	0.3 ± 0.2	0.4 ± 0.3
Gambusia holbrooki	mosquitofish	4.5 ± 2.4	16.9 ± 9.0
Heterandria formosa	least killifish	0.2 ±0.2	0.7 ± 0.6
Jordanella floridae	flagfish	--	0.2 ± 0.2
Labidesthes sicculus	brook silverside	0.2 ± 0.2	0.1 ± 0.1
Lacania goodei	bluefin killifish	--	0.2 ± 0.2
Lepisosteus osseus	longnose gar	--	0.1 ± 0.05
Lepisosteus platyrhincus	Florida gar	36.8 ± 2.9	19.6 ± 3.0
Lepomis gulosus	warmouth	1.6 ± 0.4	4.8 ± 1.6
Lepomis macrochirus	bluegill	19.1 ± 4.8	16.5 ± 4.0
Lepomis marginatus	dollar sunfish	--	0.3 ± 0.1
Lepomis microlophus	redear sunfish	2.6 ± 1.0	4.4 ± 0.9
Lepomis punctatus	spotted sunfish	0.1 ± 0.1	1.5 ± 0.7
Micropterus salmoides	largemouth bass	7.9 ± 3.5	9.4 ± 0.7
Notemigonus crysoleucas	golden shiner	14.4 ± 5.5	11.7 ± 4.3
Poecilia latipinna	sailfin molly	0.1 ± 0.1	0.2 ± 0.1
Pomoxis nigromaculatus	black crappie	0.3 ± 0.1	0.9 ± 0.02

Catch per unit effort for individual taxa was calculated for each year of study by dividing the total number of fishes collected at all sites (site data were pooled) by total pedal time (total amount of electrofishing effort). Mean annual CPUE was calculated by summing yearly CPUE values and dividing by the number of sample years.

Results

Lepomis auritus and *L. macrochirus* were dominant in all reference rivers, with mean annual relative abundances exceeding 18% (range: 18.7–23.2%) and 14% (range: 14.8–35.0%), respectively (Table 13-6). Other centrarchids contributing greater than 5% mean annual relative abundance included *L. punctatus, L. microlophus, L. gulosus*, and *M. salmoides* (Table 6). *Gambusia holbrooki* and *Notropis petersoni* were the remaining dominant species in the Withlacoochee River, while *N. crysoleucas* and *Fundulus seminolis* contributed greater than 5% in the St. Johns River (Table 13-6). Centrarchids collectively comprised ≥ 70% of the river channel fish community in all three reference rivers (Table 13-7).

Discussion

Results of electrofish sampling data indicate mean annual relative abundance of centrarchids at Control and Impact sites was 31.8% and 38.3%, respectively. Centrarchids are abundant in most freshwater river systems in Florida and are dominant in several (Bass and Cox 1985, Bass 1990). The relative contribution of centrarchid species to fish populations within peninsular Florida rivers is great when compared to the rest of the southeastern United States (Swift et al. 1986, Gilbert 1987). Members of the family Centrarchidae (sunfishes) made up more than 70% of CPUE relative abundance in the three reference rivers. Thus, decreased relative abundance of centrarchids in the channelized system is a likely indication that riverine habitat is no longer suitable for sustaining the abundance of centrarchids typical of the region.

Reestablishment of continuous flow will facilitate increased mean annual relative abundance of *L. auritus* and *L. punctatus* in restored river channels. *Lepomis auritus* is considered to be a predominantly stream-dwelling species (Lee et al. 1980, Aho and Terrell 1986). Abundance of *L. auritus* increased in Pool B river channels following implementation of the Demonstration Project and was believed to reflect a response to reestablished flows (Wullschleger et al. 1990). Although *L. punctatus* occurs in more diverse habitats than *L. auritus* (Loftus and Kushlan 1987), it is common in moderately flowing waters with vegetation or other cover (Lee et al. 1980). Abundance of *L. punctatus* also increased in Pool B following reintroduction of flow (Wullschleger et al. 1990). Centrarchid relative abundance will increase as a result of restoration and will be due, in part, to increased abundance of *L. auritus* and *L. punctatus*.

Abundance of tolerant species (least affected by seasonally low levels of dissolved oxygen) in river channel habitats at Control sites suggests this group has increased by 900% since channelization, and is an indication of decreased habitat quality in the channelized system. Florida gar (*Lepisosteus platyrhincus*), bowfin (*Amia calva*), and mosquitofish were the dominant tolerant species at Impact sites. These taxa typically increase in relative abundance in rivers with reduced water quality, especially in those rivers exhibiting chronically low levels of dissolved oxygen (Bass and Cox 1985, Bass 1990, Champeau 1990).

Dissolved oxygen levels were typically low within remnant river channels under channelized conditions, especially during summer months when water temperatures were high. Relative composition of fishes in the river channel is expected to significantly change following restoration, as relative abundance of tolerant species declines. Electrofishing conducted to evaluate effects of the Demonstration Project indicated revitalized runs in Pool B supported greater species richness, and centrarchids contributed a higher percentage of the total catch (numbers and biomass) than in a stagnant run in Pool E (Wullschleger et al. 1990). Increased levels of dissolved oxygen will allow centrarchids and other less tolerant species to better compete with tolerant species for available resources.

<u>Expectation for River Channel Fish Assembleages.</u> Four relative abundance metrics *(L. platyrhinchus, A. calva, L. auritus*, and centrarchids) show strong differences between baseline and reference conditions (Figure 13-8) were used to develop the expectation for assessing change in river channel fish assemblage

structure following restoration. Relative abundances of *L. platyrhincus* and *A. calva* are typically higher in river systems with degraded water quality (Champeau 1990, Bass 1991). Relative abundance of *L. auritus* is positively correlated with increased flow (Aho and Terrell 1986). Relative abundances of *L. platyrhincus* and *A. calva* are influenced by flow dependent habitat availability, and both species prefer little to no flow and abundant aquatic vegetation. (Lee et al 1980, Mettee et al. 1996). Reestablishment of historic sand substrate and sandbars will increase spawning habitat for *L. auritus* and other centrarchids (Carlander 1977, Aho and Terrell 1986), with increased recruitment resulting from reestablishment of river channel–floodplain linkage that historically provided floodplain habitat as refugia for juveniles (FGFWFC 1957).

Table 13-6. Mean ± SE annual relative abundance of fishes collected by electrofishing by Florida Game and Fresh Water Fish Commision between 1983 and 1990 in the St. Johns (STJ), Oklawaha (OKL), and Withlacoochee (WIT) Rivers

Species	Common Name	STJ	OKL	WIT
Alosa sapidissima	American shad	0.02 ± 0.01	0.3 ± 0.04	--
Ameiurus catus	white catfish	0.3 ± 0.2	0.1 ± 0.04	0.1 ± 0.01
Ameiurus natalis	yellow bullhead	0.1 ± 0.01	0.5 ± 0.2	0.1 ± .06
Ameiurus nebulosus	brown bullhead	0.3 ± 0.1	0.1 ± 0.03	0.04 ± 0.02
Amia calva	bowfin	0.6 ± 0.2	0.8 ± 0.1	1.3 ± 0.4
Anguilla rostrata	American eel	0.2 ± 0.1	--	0.1 ± 0.05
Aphredoderus sayanus	pirate perch	0.03 ± 0.01	2.0 ± 0.4	0.9 ± 0.4
Centrarchus macropterus	flier	0.01 ± 0.01	--	--
Dorosoma cepedianum	gizzard shad	0.9 ± 0.4	0.3 ± 0.2	0.03 ± 0.02
Dorosoma petenense	threadfin shad	0.3 ± 0.2	0.05 ± 0.02	0.04 ± 0.03
Elassoma evergladei	Everglades pygmy sunfish	--	0.01 ± 0.01	0.07 ± 0.02
Elassoma zonatum	banded pygmy sunfish	--	0.01± 0.01	--
Ennecanthus gloriosus	bluespotted sunfish	0.03 ± 0.02	0.02 ± 0.01	0.5 ± 0.2
Erimyzon sucetta	lake chubsucker	0.6 ± 0.1	2.5 ± 0.3	1.6 ± 0.4
Esox americanus	redfin pickerel	--	0.03 ± 0.01	0.2 ± 0.1
Esox niger	chain pickerel	0.08 ± 0.01	0.6 ± 0.1	0.1 ± 0.03
Etheostoma fusiforme	swamp darter	--	0.6 ± 0.2	0.2 ± 0.08
Fundulus chrysotus	golden topminnow	--	0.01 ± 0.01	0.1 ± 0.06
Fundulus seminolis	Seminole killifish	6.0 ± 1.8	0.1 ± 0.07	0.1 ± 0.04
Gambusia holbrooki	mosquitofish	0.3 ± 0.2	0.5 ± 0.1	6.4 ± 2.3
Heterandria formosa	least killifish	0.03 ± 0.03	--	0.1 ± 0.04
Ictalurus punctatus	channel catfish	0.1 ± 0.06	0.02 ± 0.01	0.03 ± 0.02
Jordanella floridae	flagfish	0.03 ± 0.03	--	0.01 ± 0.01
Labidesthes sicculus	brook silverside	0.4 ± 0.1	1.5 ± 0.3	2.7 ± 1.2
Lacania goodie	bluefin killifish	0.1 ± 0.05	0.03 ± 0.01	0.2 ± 0.1
Lepisosteus osseus	longnose gar	0.1 ± 0.03	0.2 ± 0.04	0.2 ± 0.03
Lepisosteus platyrhincus	Florida gar	2.4 ± 0.4	1.3 ± 0.2	2.9 ± 0.9
Lepomis auritus	redbreast sunfish	18.7 ± 1.2	23.2 ± 1.6	19.2 ± 2.9
Lepomis gulosus	warmouth	1.3 ± 0.2	4.9 ± 0.5	6.1 ± 0.4
Lepomis macrochirus	bluegill	35.0 ± 1.1	27.7 ± 2.4	14.8 ± 2.8
Lepomis marginatus	dollar sunfish	0.03 ± 0.03	0.1 ± 0.04	2.5 ± 0.7
Lepomis microlophus	redear sunfish	8.1 ± 1.1	9.3 ± 0.6	6.7 ± 1.8
Lepomis punctatus	spotted sunfish	3.4 ± 0.3	10.7 ± 1.5	18.5 ± 2.1
Lucania parva	rainwater killifish	0.05 ± 0.03	--	--
Menidia beryllina	inland silverside	0.7 ± 0.3	0.01 ± 0.01	--
Menidia peninsulae	tidewater silverside	0.5 ± 0.4	--	--
Micropterus salmoides	largemouth bass	4.8 ± 0.2	5.3 ± 0.4	5.8 ± 2.3
Morone saxatilis	striped bass	0.02 ± 0.02	--	--
Morone sp.	sunshine bass	0.1 ± 0.1	--	--
Mugil cephalus	striped mullet	2.7 ± 0.3	0.1 ± 0.04	0.1 ± 0.07
Myrophis punctatus	speckled worm eel	--	--	0.01 ± 0.01
Mugil curema	white mullet	0.03 ± 0.03	--	--
Notemigonus crysoleucas	golden shiner	6.3 ± 0.8	1.7 ± 0.3	0.5 ± 0.1
Notropis maculates	taillight shiner	1.5 ± 2.4	0.8 ± 0.2	0.6 ± 0.1
Notropis petersoni	coastal shiner	0.01 ± 0.01	2.0 ± 0.6	5.6 ± 2.3
Noturus gyrinus	tadpole madtom	--	0.04 ± 0.01	0.3 ± 0.1
Noturus leptacanthus	speckled madtom	--	0.06 ± 0.01	--
Opsopoedus emilidae	pugnose minnow	0.1 ± 0.1	0.01 ± 0.01	--
Oreochromis aureus	blue tilapia	0.05 ± 0.02	0.01 ± 0.01	--
Percine nigofasciata	blackbanded darter	--	1.3 ± 0.4	--
Poecilia latipinna	sailfin molly	0.03 ± 0.03	0.1 ± 0.05	0.5 ± 0.1
Pomoxis nigromaculatus	black crappie	2.1 ± 0.3	0.5 ± 0.1	0.3 ± 0.2
Strongylura marina	Atlantic needlefish	0.8 ± 0.3	0.05 ± 0.01	0.08 ± 0.04
Trinectes maculates	hogchoker	0.03 ± 0.02	0.02 ± 0.01	0.2 ± 0.1

Table 13-7. Percent contribution by centrarchids collected using electrofish sampling within three reference rivers between 1983 and 1990 and the Kissimmee River between 1992 and 1994. (KIS = Kissimmee River, STJ = St. Johns River, OKL = Oklawaha River , WIT = Withlacoochee River).

Species	KIS	STJ	OKL	WIT
Centrarchus macropterus	--	0.01 ± 0.01	--	--
Ennecanthus gloriosus	0.5 ± 0.2	0.03 ± 0.02	0.02 ± 0.01	0.5 ± 0.2
Lepomis auritus	--	18.7 ± 1.2	23.2 ± 1.6	19.2 ± 2.9
Lepomis gulosus	4.8 ± 1.6	1.3 ± 0.2	4.9 ± 0.5	6.1 ± 0.4
Lepomis macrochirus	16.5 ± 4.0	35.0 ± 1.1	27.7 ± 2.4	14.8 ± 2.8
Lepomis marginatus	0.3 ± 0.1	0.03 ± 0.03	0.1 ± 0.04	2.5 ± 0.7
Lepomis microlophus	4.4 ± 0.9	8.1 ± 1.1	9.3 ± 0.6	6.7 ± 1.8
Lepomis punctatus	1.5 ± 0.7	3.4 ± 0.3	10.7 ± 1.5	18.5 ± 2.1
Micropterus salmoides	9.4 ± 0.7	4.8 ± 0.2	5.3 ± 0.4	5.8 ± 2.3
Pomoxis nigromaculatus	0.9 ± 0.02	2.1 ± 0.3	0.5 ± 0.1	0.3 ± 0.2
TOTAL	38.3	73.4	81.7	74.4

The remaining metric, percent centrarchid composition, was chosen because peninsular Florida river systems are typically dominated by centrarchids (Swift et al. 1986, Gilbert 1987) (Table 13-4). The restoration expectation for river channel fish assemblages states that mean annual relative abundance of fishes in the restored river channel will consist of ≤ 1% bowfin *Amia calva*, ≤ 3% Florida gar *Lepisosteus platyrhincus*, ≥ 16% redbreast sunfish *Lepomis auritus*, and ≥ 58% centrarchids (sunfishes) (Figure 13-8).

Restoration of pre-channelized discharge patterns will increase levels of dissolved oxygen due to reaeration through turbulent mixing, flushing of accumulated organic deposits, and reduction in associated biological oxygen demand (Toth 1993, 1996). Baseline dissolved oxygen regimes persist at the tolerance threshold (2.0 ppm) for many fish species (Moss and Scott 1961, Davis 1975) and periodically reach critically low levels (<0.5 ppm) during summer months (Toth 1993, Koebel 1995). Depressed levels of dissolved oxygen negatively affect survivorship of all life history stages of most large-bodied species currently inhabiting the system, and may be the primary factor influencing decreased densities of large-bodied fish since channelization. Dissolved oxygen profiles are expected to become less stratified (especially during summer months), with higher levels of dissolved oxygen throughout the water column. Increased levels of dissolved oxygen will allow for increased survivorship of all life history stages of large-bodied fishes, especially species intolerant (i.e., centrarchids) of low levels of dissolved oxygen, thus allowing them to better compete with tolerant species (i.e., *L. platyrhinchus* and *A. calva*).

Numerous physical changes within restored river channels will confer change in river channel fish assemblage structure. Changes in river channel geomorphology also will affect riverine fish diversity and density. Existing cross sections impede community partitioning through lack of depth diversity and decreased availability of instream microhabitats. Geomorphic features including erosion and deposition zones provide a range of flow velocities that are used differently by dissimilar species and life history stages (Lobb and Orth 1991, Sheldon and Meffe 1995). Reintroduction of instream flow will flush accumulated organic deposits and provide the topographic diversity necessary to produce a range of flow velocities useful to a larger consort of species and life history stages (Bain et al. 1988, Lobb and Orth 1991, Sheldon and Meffe 1995). Newly created zones of erosion and deposition will include scour areas (providing deep-water habitat), point bars (creating back eddies and slower current velocities), and shoals (creating spawning grounds and shallow water habitat). River channel depth diversity can be positively correlated with fish community attributes including biomass, species richness, density, and mean size (Lobb and Orth 1991, Sheldon and Meffe 1995). Erosional processes also will create snags as riparian trees are displaced into the river. Snags provide relief from high velocities, as well as an abundance of prey items such as aquatic invertebrates, which use woody debris as a substrate for attachment and feeding (Benke et al. 1985, Lobb and Orth 1991). These physical attributes and processes will be responsible to some degree for influencing changes in the metric developed for river channel fish assemblages.

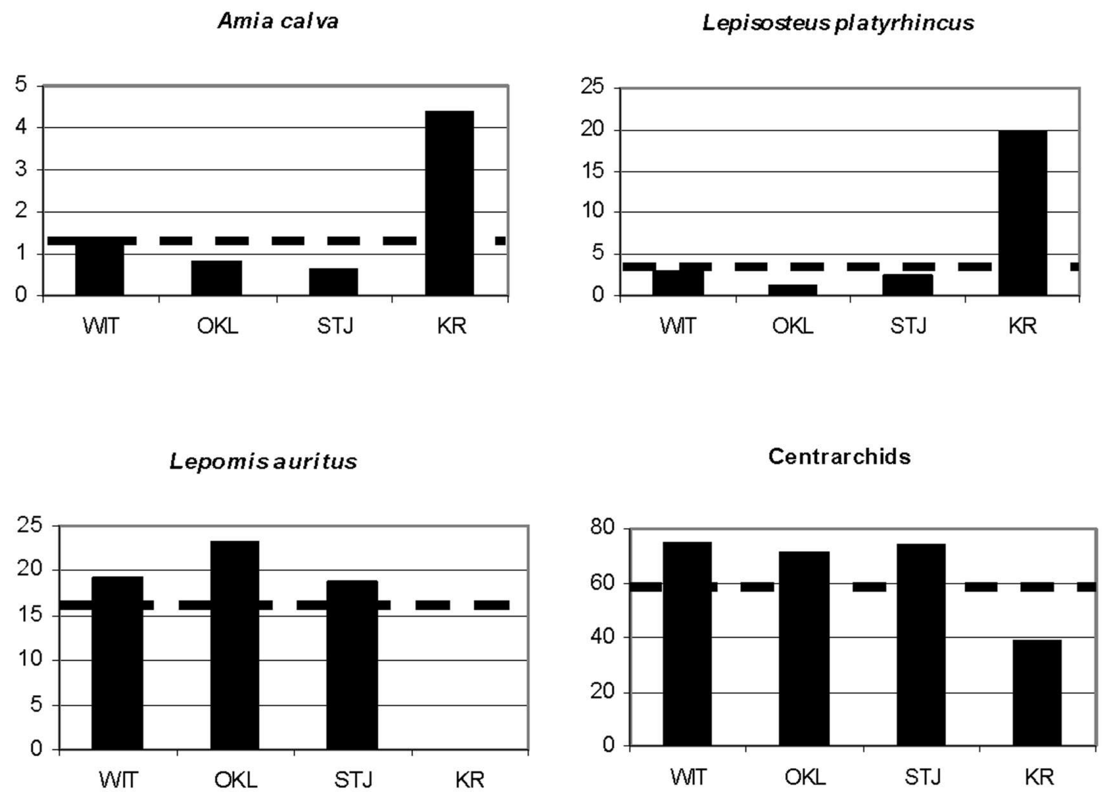


Figure 13-8. Baseline mean annual relative abundance of fish taxa or family that will be used as metrics to evaluate restoration success in reestablishing river channel fish assemblage structure. Dashed line indicates expected value for each taxon or family following restoration. (WIT = Withlacoochee River, OKL = Oklawaha River, STJ = St. Johns River, KR = Baseline data from Kissimmee River).

Similar effects of channelization on fish assemblages have been documented in other systems. Tarplee et al. (1971) found channelized Coastal Plain streams in North Carolina had reduced biomass, diversity, carrying capacity, and number of harvestable sized game fishes, notably centrarchids. They also noted that channelization adversely affected game fish to a greater degree than nongame fish. Hortle and Lake (1983) attributed decreased abundance and species richness of fishes in Australian streams after channelization to loss of suitable habitat (i.e., area of snags, area of slack water, length of bank fringed with vegetation). Other studies attribute reduced standing crop, density, and diversity of stream fish assemblages to decreased habitat, as well as decreased cover and shelter, prey or other food items, and available spawning areas (Guillory 1979, Welcomme 1985, Scheaffer and Nickum 1986, Copp 1989, Junk et al. 1989). Karr and Schlosser (1978) suggested that as much as 98% of the standing crop of fishes in a river may be lost when the flood regime is altered by channelization.

Sampling will be conducted annually, for three year-periods, beginning on the second year following implementation of the Final Headwater Regulation Schedule. Sample methods will be identical to baseline studies (FGFWFC 1996).

STUDIES WITHOUT ASSOCIATED RESTORATION EXPECTATIONS

I. FLOODPLAIN FISH ASSEMBLAGE STRUCTURE

Baseline Condition

Methods

Floodplain fish assemblage metrics included species richness (S = the total number of species present), species diversity (H'), where H' = $-\Sigma\, p_i \ln p_i$ and p_i is the proportional abundance of the ith species, and community evenness (J'), where J = H'/lnS (Price 1984). Species richness was calculated for each habitat seasonally and for the entire baseline period. Seasons were defined as wet (June through November) and dry (December through May). Values of evenness were compared between like habitats to better understand results of the Shannon index, as it evaluates both species richness and evenness within a community. Mean species diversity and evenness were calculated for the baseline period and seasonally by summing monthly values and dividing by the number of months sampled over each period. Mean species diversity was compared between Control and Impact sites for all habitats using General Linear Models (ANOVA; SAS Institute 1990). These metrics were not used in restoration expectation development.

Results

Species richness was highest within Pool C S.CMF (7), followed by Pool D S.CMF and both BLMsites (6), Pool A UP (4), and Pool C UP (3) (Table 13-8). A similar trend was observed for species richness during wet and dry seasons; however, no species were collected within UP sites during the wet season. Species diversity (H') was low in all habitats over the baseline period and ranged from 0.64 in Pool C UP to 0.77 in Pool C C.MCF (Table 13-8). Mean diversity of all floodplain samples during the baseline period was not significantly different between Control and Impact sites in any habitat (ANOVA; $p > 0.05$). Seasonal species diversity showed similar ranges (wet: 0.00–0.77; dry: 0.00–0.86). Mean wet and dry season diversity in similar habitats also was not significantly different ($p > 0.05$). Community evenness (J') was low to moderate in all habitats (range: 0.00–0.57) (Table 13-8). Evenness showed greater seasonal variability and was higher during the dry season for both S.CMF sites and Pool C UP.

Table 13-8. Community structure indices for baseline floodplain fish assemblages. Results for the entire study period are summarized in Section A. Section B lists indices calculated for each habitat during wet (w) and dry (d) seasons. (S=Species Richness, H'=Shannon index, J'=Evenness). Habitats sampled included Broadleaf Marsh (BLM). Woody Shrub (S.CMF), and Pasture (UP).

		BLM		S.CMF		UP	
A.	Index	Pool A	Pool C	Pool C	Pool D	Pool A	Pool C
	S	6	6	7	6	4	3
	H'	0.43 ± 0.11	0.29 ± 0.09	0.77 ± 0.08	0.73 ± 0.11	0.17 ± 0.12	0.06 ± 0.06
	J'	0.39 ± 0.09	0.25 ± 0.08	0.56 ± 0.05	0.57 ± 0.08	0.17 ± 0.11	0
B.				Control			
		Pool A (w)	Pool A (d)	Pool D (w)	Pool D (d)	Pool A (w)	Pool A (d)
	S	5	5	6	5	0	4
	H'	0.42 ± 0.16	0.43 ± 0.15	0.62 ± 0.15	0.86 ± 0.15	0	0.37 ± 0.24
	J'	0.40 ± 0.14	0.37 ± 0.11	0.47 ± 0.12	0.68 ± 0.12	0	0.37 ± 0.23
				Impact			
		Pool C (w)	Pool C (d)	Pool C (w)	Pool C (d)	Pool C (w)	Pool C (d)
	S	6	4	7	6	0	3
	H'	0.32 ± 0.15	0.25 ± 0.13	0.77 ± 0.13	0.77 ± 0.10	0	0.14 ± 0.14
	J'	0.28 ± 0.13	0.22 ± 0.10	0.51 ± 0.08	0.61 ± 0.05	0	0

Reference Condition

Methods

Between 1956 and 1957 the Florida Game and Fresh Water Fish Commission (FGFWFC) sampled fish assemblages of the pre-channelized Kissimmee River to provide consideration and guidance to the U. S. Army Corps of Engineers (ACOE) for the planned channelization of the river. The sampling method employed and habitat characteristics of the sample area are unclear. Fishes were collected from a single 0.1 ha sample of floodplain marsh to which rotenone was applied. Water depths in the sample area ranged from "shallow" to 1.0 m (FGFWFC 1957). Sampling was conducted in June 1957, one year following an extreme drought.

Results

Pre-channelized floodplain marsh supported 24 species (Table 13-2) and was reasonably diverse (H' = 2.53). Community evenness was 1.86.

Discussion

Although reference data on floodplain fish assemblage structure come from a single sample, it is evident that pre-channelization floodplains supported, at least periodically, a relatively diverse fish community. Because species richness within Control and Impact sites was similar under channelized conditions, any increases within Impact sites will be clearly linked to restoration, if species richness within Control sites remain similar to baseline values following restoration. Pool A BLM is expected to be inundated a greater portion of the year as a result of elimination of a secondary drainage ditch in this region; however, this change in hydrology will not be as great as in Pool C, and the Pool A BLM will not be adjacent to a restored river reach. The degree of change in species richness from the baseline condition within Impact sites will be significant (>300%), if species richness approximates that found prior to channelization (24 species).

Species diversity for the baseline period was low (see Margalef 1972, Magurran 1988) and varied little seasonally among habitats. Increased fish species diversity within floodplain habitats following restoration will require reestablishment of specific system functions and microhabitats. Reestablishment of a fluctuating hydrograph and spatial and temporal variability in inundation depth across the floodplain will lead to restoration of backwater lakes and ponds (for supporting large-bodied species), deep and shallow marsh, and a peripheral, shallow wet prairie (nursery and refuge areas for small-bodied fish, and young-of-the-year and juvenile large-bodied species). Diversity will increase significantly (>100%) following restoration of these floodplain habitats if it approximates that found in pre-channelized marsh (FGFWFC 1957). Species diversity is likely to exhibit seasonal trends following restoration of a seasonal flood pulse. Diversity values are likely to be higher during the wet season when hydrologic conditions favor use by the majority of fish taxa in the system.

II. RIVER CHANNEL FISH ASSEMBLAGE STRUCTURE

Baseline Conditions

Methods

Attributes of river channel fish assemblage structure were studied using block nets coupled with a fish toxicant (5% emulsified rotenone) and hoopnets. Each sampling method evaluated specific metrics of fish assemblage structure, because neither method was free of bias for all metrics. These metrics were not used in restoration expectation development.

Block Net Sampling. Block net samples are one of the few sampling methods that estimate fish density directly (Bettoli and Maceina 1996). Collection sites for single block net samples (0.4 acre) were selected randomly within three remnant river channels of each pool (Pool A: Ice Cream Slough Run, Rattlesnake Hammock Run, Persimmon Mound Run, Figure 13-1; Pool C: Oxbow 13, Micco Bluff Run, MacArthur Run, Figure 13-1) by driving a randomly determined number of seconds (obtained by random number generator) at 2000 rpm into each river run. The three samples in each pool were considered replicates.

Sampling was conducted in late spring months (May, June) to maximize toxicant effectiveness (Bettoli and Maceina 1996) while minimizing its persistence in the environment (Gilderhus et al. 1988), and to coincide with time periods exhibiting minimum to no flow within the pre-channelized system. Block net sampling was conducted over a ten-day (five days per pool) period in 1997 and 1998.

> Day One - On the first day of sampling, paired block nets (60.9 m x 6.9 m) were deployed perpendicular to the river bank, spaced 50 meters apart, and extending across the river channel. Nets extended from the water surface to the river channel substratum by floats and lead lines. Water column depth was recorded at 10 m intervals along three transects (center of channel and half distance to shore on each side of centerline) within each enclosure. Mean water depth within the sample area was calculated using depths obtained from each transect. Total volume of aqueous habitat sampled was estimated as: Volume = shoreline length X river width X mean water depth. Sample locations were recorded in space and time using GPS (Trimble® GPS Pathfinder Pro XL).
>
> Day Two - Prior to rotenone application, 50 fishes of various species were captured within the enclosed area using electroshocking gear (Coffelt® model VVP-15), marked by fin clipping or fine fabric Floy® tags (depending upon fish size), and released into the enclosed area for recapture to determine sampling efficiency. Five percent emulsified rotenone was applied within each block net to achieve concentration levels of 2–3 ppm (e.g., 2.466 l rotenone/ 4046.854 m^2 = 2 ppm).
>
> Days Three through Five - Poisoned fish were collected, identified to species, measured (standard and total lengths; mm), weighed (grams), and counted. Weights for fish collected subsequent to Day Three were assigned from length-weight regressions developed from first day collections. Three 15 minute electroshocking episodes were conducted (subsequent to collection of poisoned fishes) within each blocked off area to collect any fish unaffected by rotenone application (Day Five only). Shocked fishes also were identified to species, measured, weighed and counted. Shocked fishes were included in all analyses. Dense littoral vegetation precluded efficient collection of small-bodied fishes; therefore, analyses of block net samples were limited to large-bodied species.

Fish species richness, relative abundance, density, and biomass were calculated for block net samples. Mean sample abundance was calculated for each pool by summing relative abundance for each run and dividing by three. Mean annual relative abundance was generated by averaging sample means for each year. Mean annual relative abundance was compared between sites using ANOVA (SAS Institute 1990) and associated means separation test. Mean annual sample density (#/0.4 acre) and mean annual sample biomass (g/0.4 acre) were calculated and similarly compared. Mean annual sample density and mean annual sample biomass were converted to #/acre and g/acre for comparisons with pre-channelization data.

Hoopnet Sampling. Hoopnet gear is selective for centrarchid species and yields relative species composition, abundance, and biomass for most game fish species. This sampling method is not as resource-consumptive as block netting and is easily replicated throughout the year without negatively affecting local fish populations because collected fishes are released. Thus, it provides data on temporal distributions.

Three hoopnets were deployed at random locations within C-38 and each of three remnant river runs (Pool A: Ice Cream Slough Run, Rattlesnake Hammock Run, Persimmon Mound Run; Pool C: Oxbow #13, Micco Bluff Run, MacArthur Run) in each pool (n = 12 nets/pool) (Figure 13-1). Sample sites in remnant river runs were selected by driving a randomly determined number of seconds at 2000 rpm from the entrance of the run being sampled (first net), and from each previously deployed net (second and third nets). Sample sites for hoopnets deployed in C-38 were similarly selected by driving a randomly determined number of seconds from the entrance of the previously sampled remnant river run. Direction traveled in C-38 from the remnant river run entrance (North, South) was determined using a random number generator (odd numbers = North, even numbers = South). All hoopnets were set at a distance greater than 50 m from the entrance of remnant river runs to minimize any bias associated with fish populations using the area near the confluence of C-38 and remnant river runs. The side of the river on which each hoopnet was deployed was determined using a random number generator (odd numbers = East, even numbers = West). At each sample location, nets were placed along the deep-water edge of the littoral zone, at a depth <300 centimeters. Hoopnets were deployed perpendicular to the riverbank, with the mouth (opening) facing downstream, and supported with 5 cm diameter PVC poles anchored to the substrate.

Hoopnet sampling was conducted monthly from September 1997 to August 1998. Hoopnets were deployed for approximately seven hours (0900–1600 hrs.) during each sampling event. Captured fish were identified to species, measured (mm; standard length, total length), weighed, and released. Analyses of hoopnet samples were limited to large-bodied species due to sampling net mesh size (5 cm x 5 cm).

Species richness was calculated for each pool. Catch per unit effort (CPUE) was calculated for abundance and biomass data. Mean seasonal abundance and biomass were calculated for each run based on each replicate taken over each season. Seasons were defined as wet (June through November) and dry (December through May). Mean seasonal abundance and biomass were compared between runs and sites using ANOVA (SAS Institute 1990) and associated means separation test.

Catch per unit effort (CPUE) was calculated for biomass data. Mean annual CPUE for biomass was calculated similarly for groups and all species combined. Mean annual biomass CPUE was compared between years and sites using ANOVA (SAS Institute 1990) and associated means separation test.

Results

Block Net Sampling. A total of 2242 fishes representing 18 species were collected during block net sampling (Table 13-9). Species richness for the baseline period was identical at Control (16) and Impact (16) sites. Total numbers of fishes sampled also was similar at Control (1112) and Impact sites (1130). Mean sample density did not differ significantly between years within Control (ANOVA; $p = 0.6898$) or Impact (ANOVA; $p = 0.0700$) sites (Table 13-10). Mean density also did not differ significantly between Control and Impact sites within Year 1 (ANOVA; $p = 0.9352$) or Year 2 (ANOVA; $p = 0.9230$). No significant difference ($p > 0.05$) was found in mean annual density between Control and Impact sites.

Mean annual relative abundance within Control sites was dominated by centrarchids (69%), followed by lepisostids (14.7%), ictalurids (6%), and exotic fish (1%). Similar mean annual relative abundance was found at Impact sites (centrarchids 75%, lepisostids 8%, ictalurids 9%, exotic fish 0.3%). Mean annual relative abundance was not significantly different between years within sites or between Control and Impact sites within years ($p > 0.05$). However, these relative abundance values may not accurately reflect river channel fish community composition because small-bodied fishes were not collected, and therefore not included in analyses.

A total of 293,011 g (live mass) of fish biomass was collected during block net sampling (Table 13-11). Total sample biomass was similar at Control (166,084 g) and Impact (126,927 g) sites. Mean sample biomass was not significantly different between years at Control (ANOVA; $p = 0.9801$) or Impact sites (Impact; $p = 0.3078$; Table 13-12). Mean sample biomass also did not differ between Control and Impact sites during Year 1 (ANOVA; $p = 0.6150$) or Year 2 (ANOVA; $p = 0.6304$). However, mean annual biomass was greater at Control sites (ANOVA; $p = 0.0504$). Lepisostids and amiids had the highest mean biomass within Control sites (38.4%), followed by centrarchids (33.4%), ictalurids (17.5%), and exotics (3.2%). Within Impact sites, mean biomass of centrarchids (38.6%) was greater than lepisostids and amiids (30.2%), ictalurids (25.4%), and exotics (2.1%).

Hoopnet Sampling. A total of 1099 fishes representing 16 species were collected by hoopnet sampling (Table 13-13). Species richness for the baseline period was similar at Control (16) and Impact (14) sites. Total numbers were similar between sites (Control 518, Impact 581). Total numbers of fishes collected within canal and remnant river runs also were similar for both Control (canal 274; river channel 244) and Impact (canal 260; river channel 321) sites; however, sampling effort was three times greater in river channels. Species composition was similar between pools, and was dominated by centrarchids.

Mean seasonal abundance was not significantly different among river runs within each pool during dry (ANOVA; Control $p = 0.3631$, Impact $p = 0.6061$) and wet (ANOVA; Control $p = 0.1115$, Impact $p = 0.0935$) seasons. Therefore, mean seasonal abundance data for all river runs within Control and Impact sites were pooled for each season, and was not significantly different between Control and Impact sites during dry (ANOVA; $p = 0.9049$) and wet (ANOVA; $p = 0.6909$) seasons (Table 13-14).

Total numbers of fishes collected differed between seasons at Control sites (dry = 355, wet = 163) and Impact sites (dry = 399, wet = 182). Mean seasonal abundance was significantly different between seasons in Ice Cream Slough (p = 0.0020), Rattlesnake Hammock (p = 0.0460), and MacArthur (p = 0.0175) river runs (Table 13-13). Therefore, data could not be pooled for comparisons between sites.

Over 440,000 g (live mass) of fish biomass was collected during hoopnet studies. Total sample biomass was similar at Control (204,788 g) and Impact sites (239,265 g). Mean seasonal biomass was not significantly different among river runs (excluding C-38 canal) within each site during dry (ANOVA; Control p = 0.3282, Impact p = 0.6826) and wet (ANOVA; Control p = 0.2397, Impact p = 0.2464) seasons (Table 13-15). Therefore, mean seasonal biomass data for all river runs within Control and Impact sites were pooled for each season, and was not significantly different between Control and Impact sites during dry (ANOVA; p = 0.6160) and wet (ANOVA; p = 0.0700) seasons (Table 13-16).

Table 13-9. Total numbers of fishes collected per river run during 1997 (A) and 1998 (B) baseline block net sampling.

SPECIES	CONTROL			IMPACT		
1997	ICS	PM	RSH	OX13	MB	MAC
Amia calva	8	7	4	4	1	2
Ameiurus natalis	1	0	0	0	0	0
Ameiurus nebulosus	7	1	0	7	4	7
Dorosoma petenense	0	0	0	0	0	3
Erimyzon sucetta	12	11	1	1	11	23
Esox niger	0	1	0	0	0	0
Ictalurus punctatus	0	0	0	0	1	0
Lepisosteus platyrhincus	14	3	7	19	30	18
Lepomis gulosus	151	12	4	90	166	58
Lepomis macrochirus	181	16	16	57	34	69
Lepomis microlophus	57	1	1	10	2	1
Lepomis punctatus	10	0	0	5	16	4
Micropterus salmoides	12	1	9	2	16	25
Oreochromis aureus	3	0	0	0	0	2
Pomoxis nigromaculatus	110	17	0	7	25	3
TOTAL	566	70	42	202	306	215
1998	ICS	PM	RSH	OX13	MB	MAC
Amia calva	3	3	5	6	5	3
Ameirus nebulosus	20	15	2	21	12	23
Clarias batrachus	0	1	0	2	5	4
Erimyzon sucetta	1	0	1	3	2	1
Esox niger	0	0	0	0	1	0
Hoplosternum littorale	0	0	2	0	0	1
Lepisosteus osseus	1	0	0	0	0	0
Lepisosteus platyrhincus	35	17	16	3	5	19
Lepomis guolosis	76	11	15	11	53	46
Lepomis macrochirus	120	14	11	34	57	58
Lepomis microlophus	12	2	5	1	10	4
Lepomis punctatus	18	0	0	0	1	4
Micropterus salmoides	5	0	0	0	2	1
Pomoxis nigromaculatus	20	2	1	1	3	5
TOTAL	311	65	58	82	156	169

ICS = Ice Cream Slough, RSH = Rattlesnake Hammock Run, PM = Persimmon Mound Run, Ox-13 = Oxbow 13 Run, MB = Micco Bluff Run, MAC = MacArthur Run.

Mean seasonal biomass was not significantly different between wet and dry seasons in each run except Ice Cream Slough, which was significantly higher during the dry season (ANOVA; p = 0.0146). Therefore, data could not be pooled for comparisons between sites. Total biomass estimates were distributed similarly among fisheries categories at both Impact (centrarchids 31%, lepisostids and amiids 28%, and ictalurids 38%) and Control (centrarchids 28%, lepisostids and amiids 36%, ictalurids 36%) sites.

Table 13-10. Mean densities of fishes collected in block net samples at Control and Impact sites. (A) Mean sample densities (± SE) of fishes collected at Control and Impact sites (n=3 at both sites for both years). (B) Mean annual density (± SE) at Control and Impact sites.

	YEAR	Mean Sample Density (± SE)	
A.			
		Control	Impact
	Year 1	565 ± 425	602 ± 80
	Year 2	360 ± 207	338 ± 68
B.			
		Control	Impact
		462 ± 144	470 ± 186

Reference Conditions

Methods

A single 0.38 acre fish sample was collected by FGFWFC in July of 1957 using block nets and 5% emulsified rotenone. The exact methods used by FGFWFC are unclear. The sample area was chosen “to include boils, whirlpools, and eddies” found in the center of a river bend (FGFWFC 1957). The sample location also included a shallow beach area in which there was a backward movement of current.

Results

Pre-channelized river channels contained 26 freshwater fish taxa belonging to 12 families (Table 13-17). Ictalurids (61.1%) dominated community composition, but small-bodied species (28.9%), centrarchids (8.8%), and catostomids (1.2%) also were present. Density of fishes within the river channel was 937 fish/0.2 ha; however, severe drought conditions occurring the previous year may have affected fish density through stress-related mortality, or alternatively, by leading to downstream emigration into Lake Okeechobee.

Discussion

Based on results of baseline block net sampling, density of river channel fishes appears to have declined by approximately 50% since channelization. Pre-channelization data indicate a density of 937 fish per 0.2 ha (FGFWFC 1957), while samples from Impact and Control sites yielded a mean of 462 and 470 fish per 0.2 ha, respectively. Results of hoopnet sampling suggest fish density and biomass vary seasonally with greater mean abundance and biomass during the dry season. This trend might be expected in river systems with a seasonal river channel-floodplain linkage because densities within the main channel are likely to decrease during the wet season as riverine species migrate onto inundated floodplain to exploit temporarily abundant resources (Welcomme 1979, Bayley 1991). However, floodplain habitats on the channelized Kissimmee were not available (or available on a very limited basis) to fishes during the baseline period. Instead, fishes may have responded to seasonal differences in dissolved oxygen concentrations within river channels. Mean dissolved oxygen concentrations in river channels were greater during the dry season (1.9–3.7 mg/L) than the wet season (0.8–1.4 mg/L) (Colangelo and Jones 2001). Fishes may have dispersed throughout the system during the wet season to minimize constraints of low levels of dissolved oxygen.

Mean annual density of river channel fish is expected to increase following restoration. Increases in densities of large-bodied fishes require restoration and maintenance of riverine habitats that match the habitat requirements of the pre-channelized community (Sheldon and Meffe 1995). Restoration of pre-channelization hydrologic characteristics, especially river channel/floodplain connectivity, will be the mechanism driving restoration of the river channel fish community. Increased export of vertebrate and invertebrate biomass from the floodplain to the river channel during the receding hydrograph should supplement fish diets (Welcomme 1979, Harris et al. 1995), thereby increasing growth and reproductive rates of most river channel species. The availability of protective floodplain habitats should lead to increased survivorship and recruitment of juveniles into breeding populations.

Table 13-11. Total biomass (g/0.4 acre) of fishes collected per river run during 1997 and 1998 baseline block net sampling.

SPECIES	CONTROL			IMPACT		
1997	ICS	PM	RSH	OX13	MB	MAC
Amia calva	11889	9017	3121	4130	1456	823
Ameiurus natalis	260	0	0	0	0	0
Ameiurus nebulosus	4341	755	0	2752	1008	3684
Dorosoma cepedianum	0	0	0	0	0	806
Erimyzon sucetta	3458	577	455	595	83	218
Esox niger	0	474	0	0	0	0
Ictalurus punctatus	0	0	0	0	838	0
Lepisosteus platyrhincus	9116	1328	1885	3121	7611	4848
Lepomis gulosus	1590	135	42	2032	2116	1326
Lepomis macrochirus	10028	2468	1203	3916	2370	5162
Lepomis microlophus	2002	258	67	1018	120	210
Lepomis punctatus	415	0	0	205	865	308
Micropterus salmoides	2075	429	3412	367	3321	722
Oreochromis aureus	5045	0	0	0	0	2383
Pomoxis nigromaculatus	7851	37	0	31	254	244
TOTAL	58078	15478	10185	18167	20042	20734
1998	ICS	PM	RSH	OX13	MB	MAC
Amia calva	4167	3743	4544	2690	2840	1911
Ameirus nebulosus	14257	7911	1475	9559	3494	10950
Clarias batrachus	0	200	0	848	694	1108
Erimyzon sucetta	104	0	460	1970	1101	159
Esox niger	0	0	0	0	524	0
Hoplosternum littorale	0	0	230	0	0	63
Lepisosteus osseus	4	0	0	0	0	0
Lepisosteus platyrhincus	7222	7037	7606	567	2354	6021
Lepomis guolosis	2677	601	567	352	2064	1408
Lepomis macrochirus	10812	1671	440	2841	6699	3682
Lepomis microlophus	591	575	98	119	519	371
Lepomis punctatus	941	0	0	0	39	158
Micropterus salmoides	1837	0	0	0	893	181
Pomoxis nigromaculatus	2144	396	33	75	383	1275
TOTAL	44756	22134	15453	19021	21604	27359

ICS = Ice Cream Slough, RSH = Rattlesnake Hammock Run, PM = Persimmon Mound Run, Ox-13 = Oxbow 13 Run, MB = Micco Bluff Run, MAC = MacArthur Run.

Relative abundance of small-bodied fishes was similar under baseline and pre-channelization conditions (pre-channelization 28.9%; baseline - Control 20.4%, Impact 33.1%). In the pre-channelization system, small-bodied fish composition was dominated by taillight shinner *Notropis maculatus* (73.8%), eastern mosquitofish *Gambusia holbrook* (8.2%), tadpole madtom *Noturus gyrinus* (5.5%), and golden shinner *Notemigonous crysoleucas* (4.8%). Mosquitofish was dominant at Impact sites (51.1%), while golden shiner was dominant at Control sites (70.6%) and abundant at Impact sites (35.6%) during baseline conditions. Dominance of mosquitofish at Impact sites likely is attributable to increased vegetative cover and decreased water quality within the channelized system.

Table 13-12. Mean biomass of fishes collected in block net samples at Control and Impact sites. (A) Annual mean biomass (± SE) of fishes collected at Control and Impact sites with block net sampling (n=3 at both sites for both years). (B) Mean biomass (± SE) at Control and Impact sites over both years of block net sampling (n=6 at both sites).

A.	Mean Sample Biomass (± SE)	
	Control	Impact
Year 1	27,918 ± 15,160	19,652 ± 767
Year 2	27,452 ± 8866	25,852 ± 3565

B.	Mean Annual Biomass (± SE)	
	Control	Impact
	27,685 ± 7854	22,738 ± 2136

Although relative abundance of small-bodied fishes was similar between baseline and pre-channelization conditions, differences in density could not be evaluated due to different sampling methods. Electrofishing does not estimate the number of fish per unit area, but provides an estimate of catch per unit effort. Electrofishing also has an inherent bias for larger fishes, and may not have provided a complete inventory of smaller individuals, including small-bodied fish species.

Species richness is not expected to change significantly following restoration. The number of large-bodied species inhabiting the Kissimmee is consistent with distributions of fishes occurring in other rivers of pennisular Florida, including the Peace, Caloosahatchie, Manatee, Alafia, Hillsborough, and Withlacoochee Rivers (Trexler 1995). Species richness of large-bodied fishes has increased since channelization due to the introduction of the following exotic species: walking catfish *Clarias batrachus*, Oscar *Astronotus ocellatus*, blue tilapia *Oreochromis aureus*, and most recently, brown hoplo *Hoplosternum littorale* in 1997 and suckermouth catfish *Pterygoplichthyes disjunctivus* in 2001. New exotic species may become established within the Kissimmee River over the next 20 years, as they work their way through the interconnected waterways of south and central Florida. Recolonization by species believed to be extirpated from the system (Perrin et al. 1982) may occur if restored conditions are amenable and a source population has access to the basin.

Small-bodied fish relative abundance likely will be higher in restored river channels due to increased production on the floodplain and subsequent transport to the river channel. Forage fish inhabited both river channel and floodplain habitats in the pre-channelization system. Forage fishes are particularly important components in the piscine food web and are a primary food item of large piscivorous species. Most piscivorous fishes undergo an ontogenetic shift from a diet of invertebrates to fishes. Fishes able to make this shift earlier exhibit faster growth rates, higher overwinter survival, and greater reproductive success (Mittlebach and Persson 1998).

Post-restoration evaluation of river channel fishes will be conducted using electrofish and hoopnet sampling; however, block net sampling will be eliminated. Block net sampling is costly, time consuming, and not amenable to high temporal and spatial replication without negatively impacting the fish population. Also, this method is permitted by the Florida Fish and Wildlife Conservation Commission only in non-flowing waters. No-flow conditions are not likely to occur within the river following restoration. All sampling will be initiated two years subsequent to initiation of the revised headwaters regulation schedule, which will provide continuous flow through the restored river channel. Electrofishing will be conducted annually for three consecutive years following two years of continuous flow within Impact sites, and will begin on the third and eighth 8^{th} years.

Table 13-13. Total number of fishes collected during baseline hoopnet collections at Control and Impact sites in the Kissimmee River under channelized conditions.

			Control	Impact
Lepisosteidae (gars)				
Lepisosteus platyrhincus	Florida gar		8	6
Amiidae (bowfins)				
Amia calva	bowfin		4	15
Clupeidae (herrings)				
Dorosoma cepedianum	gizzard shad		14	4
Esocidae (pikes)				
Esox niger	chain pickerel		1	2
Catostomidae (suckers)				
Erimyzon sucetta	lake chubsucker		31	39
Iclaluridae (catfishes)				
Ameiurus natalis	yellow bullhead		2	1
Ameiurus nebulosus	brown bullhead		112	130
Callichthyidae (armored catfishes)				
Hoplosternum littorale	brown hoplo		4	4
Centrarchidae (sunfishes and basses)				
Lepomis gulosus	warmouth		3	3
Lepomis macrochirus	bluegill		199	212
Lepomis microlophus	redear sunfish		37	59
Lepomis punctatus	spotted sunfish		3	4
Micropterus salmoides	largemouth bass		18	20
Pomoxis nigromaculatus	black crappie		79	85
Cichlidae (cichlids)				
Oreochromis aureus	blue tilapia		3	3
		TOTAL	518	581

Analysis of condition indices and growth rates might be useful in detecting restoration associated change and may be incorporated as a metrics for post-restoration evaluation. Growth exponent b and growth rates were determined under baseline conditions by Arrington and Jepsen (2001). Growth exponent b measures length-weight relationships in fishes and provides information on the relative health or "plumpness" of fishes. Growth rates were determined using linear and von Bertalanffy growth function relationships of standard length on age (determined through otolith analysis).

III. CREEL SURVEYS

Baseline Condition

Methods

Estimates of angler effort and success were evaluated for the Kissmmee River/C-38 system by the Florida Game and Fresh Water Fish Commission (FGFWFC) for the period of March 1992 through March 1994 via stratified roving creel surveys with non-uniform sampling probabilities (FGFWFC 1994). The Kissimmee River was divided into three units on the basis of access and time required to survey each unit. Pools A and B, Pools C and D, and Pool E were treated as individual units (Figure 13-1). Fishing success in each unit was assumed to be equal. Proportional fishing effort in each unit, and for each month, were estimated from a year-round aerial survey of boats in the channelized system.

Table 13-14. Mean seasonal abundances (± SE) of fishes in hoopnet samples from river channels at Control and Impact sites.

Control		Impact	
	Dry Season		
ICS (n = 12)	3.25 ± 0.69	Ox-13 (n = 15)	4.86 ± 1.01
RSH (n = 12)	3.40 ± 0.83	MB (n = 15)	6.53 ± 3.28
PM (n = 12)	7.40 ± 3.63	Mac (n = 15)	3.60 ± 1.00
	Wet Season		
ICS (n = 18)	1.05 ± 0.24	Ox-13 (n = 18)	3.16 ± 0.88
RSH (n = 18)	1.72 ± 0.23	MB (n = 18)	2.61 ± 0.58
PM (n = 18)	1.27 ± 0.17	Mac (n = 18)	1.27 ± 0.13

ICS = Ice Cream Slough, RSH = Rattlesnake Hammock Run, PM = Persimmon Mound Run, Ox-13 = Oxbow 13 Run, MB = Micco Bluff Run, MAC = MacArthur Run.

Table 13-15. Mean seasonal abundances (± SE) of fishes in hoopnet samples from river channels (pooled data) at Control and Impact sites.

	Control	Impact
Dry Season	4.78 ± 1.35 n = 42	5.00 ± 1.17 n = 45
Wet Season	1.35 ± 0.13 n = 54	2.35 ± 0.36 n = 54

Twenty-six contiguous periods consisting of one weekday sample and one weekend sample were scheduled during each year. Peak sampling intensity was scheduled during the months of peak fishing effort (June through November) and a minimum of two samples per month were scheduled during the months of least fishing pressure. The starting point of each sample (north and south end of the unit), the order of creel tasks (instantaneous count of anglers or angler interviews), and the actual date of sampling (one weekday sample and one weekend sample per period) were chosen with uniform probability. The unit and time of date to be sampled were chosen randomly with non-uniform probability. The probability of selecting a unit to be sampled was based on the estimated proportional fishing effort in that unit, and the probability of selecting the time of day was 0.60 for the a.m. period and 0.40 for the p.m. period based on the proportional amount of fishing expected during each time period. The sample units were divided into two areas, remnant river channels and the C-38 canal. Instantaneous angler counts were conducted by boat within C-38 and remnant river runs longer than 0.8 km. For angler interviews, hours fished for all species, hours fished for particular species, and catch were recorded.

Reported values for fishing effort and success come directly from a FGFWFC completion report (FGFWFC 1994), which did not provide raw data. Fishing effort and success were determined for C-38 canal, remnant river runs, and both areas combined. Also, results by sampling unit were not provided, but instead were reported as overall values. Therefore, differences between units could not be determined. Annual estimates of effort and success are presented with corresponding percent coefficient of variation as compiled in FGFWFC (1994). Species categories include largemouth bass, black crappie, sunfishes (*L. gulosus, L. macrochirus, L. microlophus, and L. punctatus*), catfish (*A. catus, A. natalis, A. nebulosus, and I. punctatus*), and general fish.

Table 13-16. Mean seasonal biomass ($\pm$ SE) of fishes in hoopnet samples from river channels at Control and Impact sites.

Control		Impact	
	Dry Season		
ICS (n = 12)	1377 ± 355	Ox-13 (n = 15)	2135 ± 488
RSH (n = 12)	983 ± 294	MB (n = 15)	2837 ± 1742
PM (n = 12)	2773 ± 1398	Mac (n = 15)	1473 ± 590
	Wet Season		
ICS (n = 18)	462 ± 1623	Ox-13 (n = 18)	1214 ± 495
RSH (n = 18)	694 ± 242	MB (n = 18)	972 ± 223
PM (n = 18)	268 ± 84	Mac (n = 18)	442 ± 171

ICS = Ice Cream Slough, RSH = Rattlesnake Hammock Run, PM = Persimmon Mound Run, Ox-13 = Oxbow 13 Run, MB = Micco Bluff Run, MAC = MacArthur Run.

Results

Total fishing effort over the period of study was 284,160 hours, and 292,188 fish were caught. Largemouth bass was the most sought after species. Total estimated effort for largemouth bass was 101,527 hours and comprised 35.7% of fishing effort. Sunfishes (30.8%) were the next most sought after group, followed by black crappie (18.5%), and catfish (5.6%). The remainder of effort (9.4%) targeted general fish. Catch rate was highest for sunfishes (1.86 fish/hour), followed by black crappie (0.79 fish/hour) and catfish (0.48 fish/hour), and was lowest for largemouth bass (0.36 fish/hour) (Table 13-18).

Table 13-17. Numbers and percent composition of fishes collected by GFC (1957) in the historic river channel using block nets and 5% emulsified rotenone. Fishes present in historic river channel, but collected using other methods, are represented with an asterisk (*) and were not used to generate percent composition.

		Number Collected	Percent Composition
Large-bodied Taxa			
Catostomidae			
Erimyzon sucetta	lake chubsucker	6	1.2
Centrarchidae			
Micropterus salmoides	largemouth bass	6	1.2
**Lepomis auritus*	redreast		
Lepomis macrochirus	bluegill	26	5.2
Lepomis gulosus	warmouth	3	0.6
Lepomis microlophus	readear sunfish	9	1.8
**Pomoxis nigromaculatus*	black crappie		
Clupeidae			
**Dorosoma cepedianum*	gizzard shad		
Ictaluridae			
Ameiurus catus	white catfish	3	0.6
**Ameiurus natalis*	yellow bullhead		
Ameiurus nebulosus	brown bullhead	3	0.6
Ictalurus punctatus	channel catfish	300	59.9
Small-bodied Taxa			
Atherinidae			
Labidesthes sp.	silverside	3	0.6
**Menidia beryllina*	inland silverside		
Clupeidae			
Dorosoma petenense	threadfin shad	1	0.2
Cyprinidae			
Notemigonous crysoleucas	golden shiner	7	1.4
Notropis maculatus	tailight shiner	107	21.3
**Notropis petersoni*	coastal shiner		
**Opsopoedus emilidae*	pugnose minnow		
Cyprinodontidae			
**Jordanella floridae*	flagfish		
Fundulidae			
**Fundulus chrysostus*	golden topminnow		
Fundulus seminolis	seminole killifish	3	0.6
Lucania goodei	bluefin killifish	1	0.2
Ictaluridae			
Noturus gyrinus	tadpole madtom	8	1.6
Percidae			
Etheostoma fusiforme	swamp darter	3	0.6
Poeciliidae			
Gambusia holbrooki	eastern mosquitofish	12	2.4
	TOTAL	500	100

Reference Condition

Methods

Estimates of angler effort and success were evaluated for the Kissimmee River by the U. S. Fish and Wildlife Service (USFWS) from June 1955 through May 1956. Fishing pressure on the Kissimmee River was determined from fishing camp records of boats rented and private boats launched. For survey purposes, the river was divided into three areas. The "upper river" included the stretch north of Dougherty Dike (exact location unknown) and Lake Kissimmee. The "middle river" included the stretch from Dougherty Dike south to Highway 70. The "lower river" included Highway 70 south to Lake Okeechobee, excluding Government Cut. Several stations in each river section were to be creeled one day each month over the study period. However, due to manpower limitations and extremely low water levels caused by severe drought, surveys were conducted on only nine dates. Survey stations were creeled by boat and each fisherman was interviewed by asking the following questions: (1) catch, (2) time fished, (3) target species, and (4) reason for choosing fishing location. Survey data is reported as percentage of total fishing effort by taxa and catch rate (number of fish/hour) by taxa.

Results

An estimated 17,066 anglers fished the lower and middle Kissimmee River during the survey period. This estimate accounted for 22% of the total fishing effort in the Kissimmee Basin. Also, the observed estimate is considered to be conservative due to limited angler access and negative angler success resulting from severe drought. Interviews with camp operators indicated that fishing pressure was off approximately 50% from the previous year.

Table 13-18. Recent fishing effort and catch rates from creel surveys conducted under pre-channelization and baseline conditions. (* Denotes estimated angler effort for largemouth bass under pre-channelization conditions when not affected by severe drought).

	Pre-channelization Survey		Baseline Survey	
	% effort	Catch rate	% effort	Catch rate
Largemouth bass	56 (75*)	0.21	35.7	0.36
Sunfish	17	0.79	31.8	1.96
Black crappie	11	0.95	18.5	0.79
Catfish	---	---	5.6	0.48
General fish	16	0.66	9.4	---

Creel data indicated that largemouth bass (56%) was the taxa most targeted by anglers (Table 13-18). Sunfishes (17%) were the next most sought after group, followed by black crappie (11%). The remainder of effort (16%) did not target individual taxa and is described as general fish. Catch rate was highest for black crappie (0.95 fish/hour), followed by sunfishes (0.79 fish/hour) and general fish (0.66 fish/hour), and was lowest for largemouth bass (0.21 fish/hour)(Table 13-18).

Comparisons and Discussion

Largemouth bass *Micropterus salmoides* was the most sought after species (56%) prior to channelization (USFWS 1959). However, this estimate is considered conservative because the survey was conducted during a severe drought, when fishing pressure was reduced by 50% from the previous year (USFWS 1959). It was estimated that greater than 75% of the total fishing effort would be directed at largemouth bass during normal water conditions. Actual fishing pressure on the river is underestimated because fishing effort in the river portion of the upper river segment could not be separated from fishing effort on Lake Kissimmee. The catch rate for largemouth bass was considered to be an all time low for the river during the pre-channelization survey period because of the severity of the drought. Because largemouth bass catch rates declined during the drought, it is believed many anglers switched their effort to more easily caught sunfishes. Most effort for black crappie was expended in the lower portion of the river during the spawning migration when large concentrations of crappie moved from Lake Okeechobee into the Kissimmee River.

Comparisons of pre-channelization and baseline creel data suggest that the focus of angling effort has changed dramatically. Most angling effort expended in the channelized Kissimmee River system was equally focused on largemouth bass and sunfishes, whereas over 50% (and possibly as much as 75%) of effort was directed at largemouth bass prior to channelization. The primary focus on largemouth bass prior to channelization is believed to be a result of the river's reputation for producing many exceptionally large individuals (Miller 1988). Comparisons of catch rates for bass under pre-channelized and baseline conditions are suspect, since catch rates in the pre-channelization study were greatly reduced as a result of extreme drought conditions. The trend of increased angler success for sunfishes following channelization reflects their concurrent increase in relative abundance, and demonstrates increased populations of adult, harvestable fish. Conversely, low catch rates of largemouth bass and black crappie indicate decreased populations of adult, harvestable fish.

A restoration expectation was not derived for angler effort and success since pre-channelization data were negatively impacted by extreme drought and do not reflect typical conditions. Also, angler effort is contingent on numerous factors other than reestablishment of ecological integrity to the river system and, therefore, is not suitable for use as an indicator of restoration success. Post-restoration evaluation of angler effort and success will be conducted using baseline methods. A three-year creel investigation will commence on the second year following implementation of the Headwaters Revitalization Schedule.

IV. FISH DIETS

Methods

Fish feeding habits were studied by examining gut contents of nine fish taxa that were selected based on trophic categories and included *Micropterus salmoides* (piscivore), *Lepomis gulosus* (invertivore/piscivore), *Lepisosteus platyrhincus* (piscivore), *L. machrochirus* (omnivore), *L. microlophus* (invertivore), *Pomoxis nigromaculatus* (invertivore/piscivore), *Erimyzon sucetta* (invertivore), *Notemigonus chrysoleucas* (omnivore), and *Gambusia holbrooki* (omnivore). Fishes were collected in and around littoral vegetation of remnant channels and C-38 canal in Pools A, B, and C using boat-mounted electrofishing gear to determine if location affected fish diets. Sample locations within each pool were selected by driving a randomly determined number of seconds at 2000 rpm from a randomly chosen point on C-38 or remnant river run. Fishes were collected during daytime hours in both winter (December 1996 and January 1997) and summer (June 1997) to include a range of environmental conditions. Fishes were placed in a mixture of ice and fresh water to arrest metabolism. In the field, standard length of fishes >100 mm was measured to the nearest mm. Stomachs were removed and preserved in buffered formalin. Smaller fishes were preserved whole in buffered formalin and returned to the laboratory for identification, measuring, and removal of stomach contents.

For large fishes, stomach contents were rinsed, separated by prey type into individual aluminum tins, and dried for 24 h at 100°C. For small fishes, stomach contents were rinsed through a series of nested sieves (0.500, 0.250, 0.150, 0.075 mm) with distilled water to sort prey items into different size categories (Livingston 1982, 1984, 1988). A 0.5 ml sub-sample from each sieve fraction was then examined under a dissecting microscope to identify prey to the lowest taxonomic level possible, and calculate relative abundance. The contents of each sieve fraction were placed into separate aluminum tins and dried for 24 h at 100°C.

For large fishes, relative prey biomass was calculated by dividing dry weights of prey species by the total dry weight of stomach contents. For small fishes, dry weights were multiplied by prey relative abundance to calculate relative prey weight for each sieve fraction. Analysis of prey data by absolute or relative weight is preferred (Wallace 1981). Data for large predators were categorized in the 0.500 mm sieve fraction. However, data across size classes were pooled to simplify interpretation.

The metrics analyzed were number of prey types eaten (prey richness) and prey weight. Analysis of variance (ANOVA) was used to test for the effect of sample location on species prey weight and relative abundance of fish prey. Tukey-Kramer post hoc tests were used for pairwise comparisons. Similarity in relative abundance of prey categories between pools was determined using cluster analysis. Prey categories included filamentous algae, annelids, microcrustaceans (e.g., ostracods, copepods), detritus, fishes, herpetofauna, aquatic insects, mollusks, grass shrimp, plant remains, crayfish, sand, and terrestrial arthropods. Simple linear regression was used to determine the relationship between length of predatory fishes and prey length.

Results

Number and relative abundance (%) of different prey items and prey richness for each taxa studied are listed in Appendices 13-1A to 13-9A. Predator length explained 22% of the variation in prey length (ANOVA, $F1,1879 = 5333.3$, $p = 0.0001$) (Figure 13-9). Prey weight (ANOVA; $F2,1923=45.4$, $p=0.0001$) varied significantly among pools (Figure 13-10). Post-hoc comparisons indicate that weights of prey taken by Pool B predators were greater than prey items of predators from Pools A and C. Prey quantity (number of prey items) was higher in Pool B predators after adjusting for differences in body length. Cluster analysis indicates that major prey groupings in Pools A and C are more closely related to one another than to Pool B prey (Figure 13-11). Relative abundance (%) of fish prey was not significantly different in fishes collected from Pools A, B, and C (ANOVA; $p >0.05$). Fish prey comprised the greatest percentage of food items in the diets of Florida gar, bowfin, and largemouth bass (Figure 13-12). Grass shrimp (*Palaemonetes* sp.) dominated the diet of warmouth in Pool A, while crayfish was the dominant food item in Pool C (Figure 13-12).

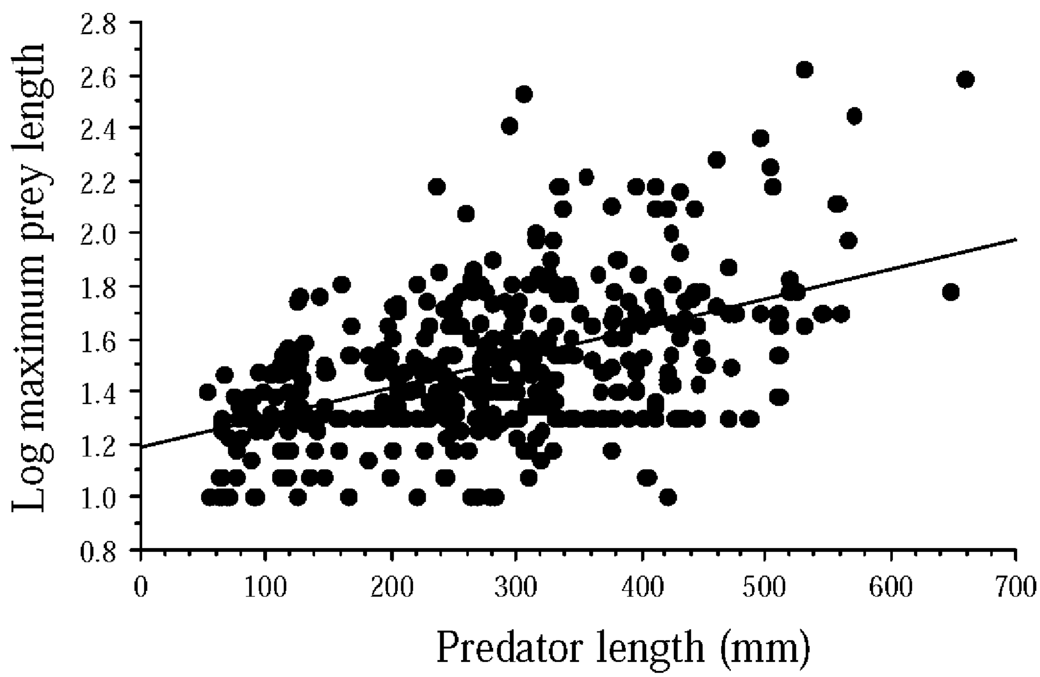


Figure 13-9. Scatterplot indicating relationship between length of fish predator and log maximum prey length. Predator length explains 24% of the variation in prey length ($F_{1,423}$= 132.1, $p<0.05$).

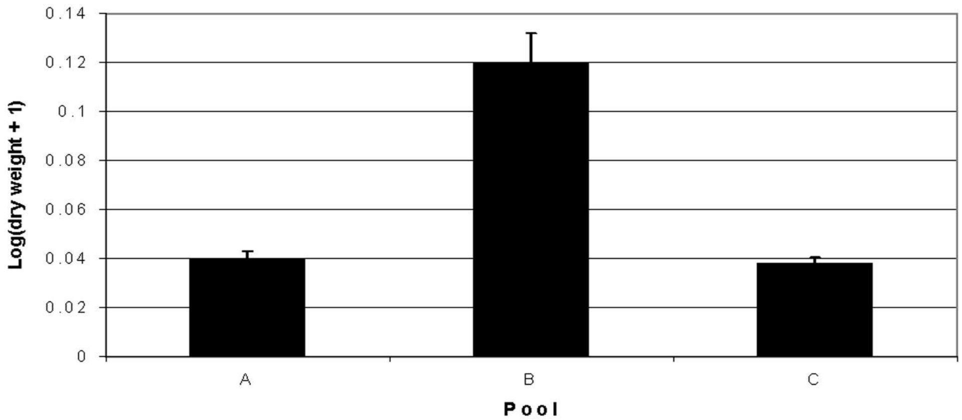


Figure 13-10. Mean (± 1 standard error) log transformed dry weight of stomach contents of predators collected from Pools A, B, and C of the Kissimmee River.

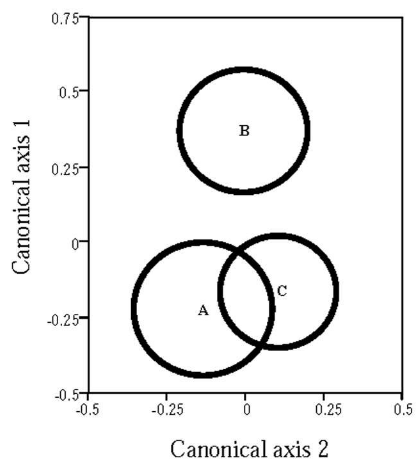


Figure 13-11. Multivariate least-squares means and 95% confidence ellipses of dominant prey types (see Table 13-4) from Pools A, B, and C of the Kissimmee River.

Discussion

Flow through three remnant river runs in Pool B was enhanced by placement of notched weirs in C-38, asscociated with the Kissimmee River Demonstration Project, located immediately downstream of the northern confluence of each river run with C-38 canal. The weirs functioned to back up water during periods of high flow, forcing water through remnant river runs and occasionally out on to limited portions of adjacent floodplain. Reintroduced flow flushed accumulated organic sediments and reduced the width of emergent vegetation along the littoral edge. Dissolved oxygen levels increased in these runs as a result of decreased sediment oxygen demand and reparation through turbulent mixing (Toth 1991). Sampled fishes in enhanced Pool B were significantly longer than their counterparts in Pools A and C, so it was not unexpected that the total weight of food in their stomachs also was greater. However, prey quantity (total number of prey items) was still higher in Pool B predators even after adjusting for differences in body length. Increased food quantity may reflect enhanced foraging opportunities that have arisen since enhancement of Pool B (Jordan and Arrington 2001).

Although the amount of prey is an important indicator of habitat quality, the types of prey available may be even more important. For example, most piscivorous fishes start life feeding on invertebrates and later undergo ontogenetic shifts to piscivory (Gerking 1994). Fish prey are apparently more energetically profitable than invertebrate prey, and fishes that switch to piscivory have faster growth rates, higher overwinter survival, and potentially greater reproductive success (e.g., Mittlebach and Persson 1998). Restoration of the Kissimmee River will result in increased connectivity between river channel and floodplain habitats and may result in more fish prey becoming available (Trexler 1995). Jordan and Arrington (In review) found that large predatory fishes in enhanced Pool B consumed greater proportions of fish prey. Although piscivory was mostly limited to large-bodied fishes, smaller fishes fed on scales and larvae. The amount of fish in a predator's diet reflected both taxonomy and foraging location. At least 90% of the diet of Florida gar was comprised of fishes, whereas the proportion of fishes in the diets of bowfin, warmouth, and largemouth bass varied considerably among Pools A, B, and C (Figure 13-12). Similarly, the relative importance of crayfish and grass shrimp also varied with fish species and foraging location. However, the similarity in prey community composition between Pools A and C indicates they should serve as good Control and Impact sites.

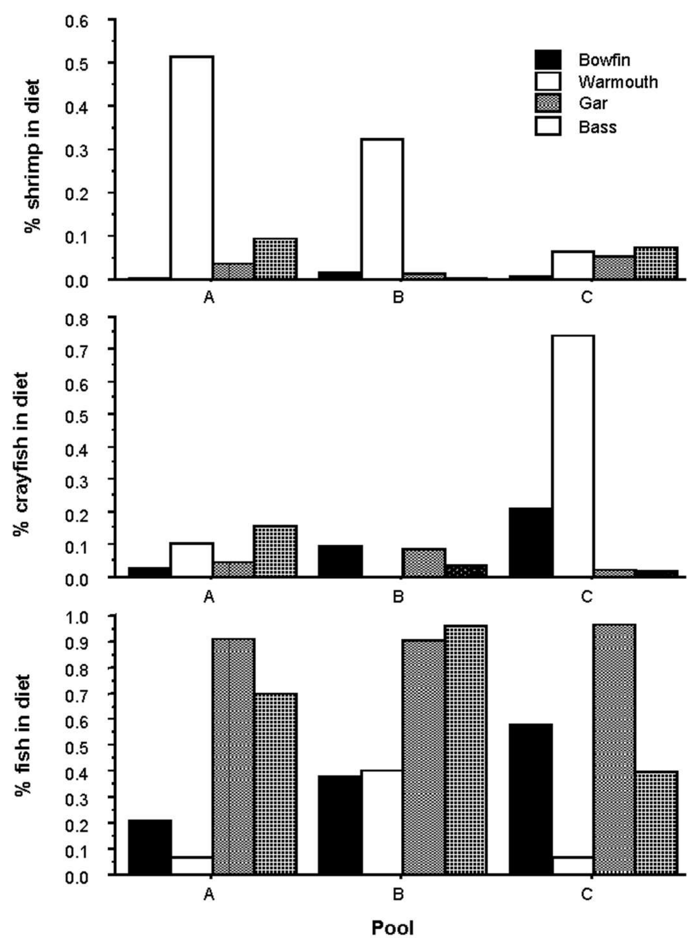


Figure 13-12. Relative abundance (%) of (a) grass shrimp, (b) crayfish, and (c) fishes in the diets of large predatory fishes collected from Pools A, B, and C of the Kissimmee River.

Analysis of fish feeding habits will be repeated during post-construction evaluations. It is expected that large predatory fishes in Pool C will consume greater proportions of fish prey than similar taxa in Pool A. Sampling should be initiated no sooner than three years following initiation of the revised Headwaters

Regulation Schedule to allow for sufficient change in river channel fish community structure and reestablishment of the aquatic food web. Ontogenetic changes in feeding habits may be an important metric to include in post-restoration analyses. A study of fish feeding habits using stable isotopes to identify energy pathways within the aquatic food web also may be incorporated during post-restoration evaluations.

V. REPRODUCTIVE EFFORT AND LARVAL FISH ASSEMBLAGE STRUCTURE

Methods

Fish larvae were sampled bi-weekly between March 11, 1997 and June 26, 1997 (eight sampling events) and between January 13, 1998 and April 8, 1998 (seven sampling events) to evaluate baseline larval fish assemblage structure within the channelized Kissimmee River. Push net sampling was conducted at fixed sites using paired, bow-mounted 505-micron plankton nets pushed just below the water's surface. Sampling effort was stratified within lower, middle, and upper zones within each pool to address the hypothesis that spawning occurs only in southern (lower) reaches of the channelized system (Trexler 1995). Two replicate samples were collected in lower, middle, and upper reaches of three remnant river runs of Pools A (Persimmon Mound Run, Kicco Run, and Ice Cream Slough Run; Figure 13-1) and C (MacArthur Run, Micco Bluff Run, and Oxbow 13; Figure 13-1). Only middle reaches of Ice Cream Slough Run and Oxbow 13 were sampled due to limited sampling area by encroachment of emergent vegetation into the center of the channel. Two replicate samples also were collected from mid-channel and littoral zones of C-38 in lower, middle, and upper regions of each pool, resulting in a total of 26 samples per pool. Mid-channel zones were sampled using replicate, side-by-side plankton nets, while littoral zones were sampled using two consecutive single net pushes.

Water quality data was collected at each site prior to sampling. Dissolved oxygen, specific conductance, pH, turbidity, and water temperature were measured using a Hydrolab™ multiprobe water quality instrument. Current velocity was measured using a Marsh-McBirney Flowmate 2000 portable flow meter. Mechanical flow meters were suspended inside each plankton net to calculate total water volume sampled. All samples were preserved in the field with 10% buffered formalin.

Fishes without a full complement of fin rays were classified as larval. Larval fish from each sample (replicate samples were not pooled) were sorted, identified to lowest possible taxonomic unit, and measured (total length) to the nearest millimeter. Species richness and relative abundance were calculated. Differences in total larval fish density within each riverine category (remnant river channel, C-38 pelagic, C-38 littoral) across three regions of each pool (lower, middle, and upper) were tested using repeated-measures analysis of variance (R-M ANOVA). Differences in longitudinal distribution of numerically significant taxa within each pool also were tested using R-M ANOVA. Differences in total larval fish density among the three riverine categories among and between pools were tested using R-M ANOVA. Correlations between larval fish density and environmental factors (levels of dissolved oxygen, pH, turbidity, water temperature, and flow rate) within each pool were tested using Pearson Correlation Coefficient (SAS Institute 1990).

Results

A total of 23 taxa were collected during the study (Table 13-19). Species richness was similar at Control ($n = 22$) and Impact sites ($n = 19$). Species richness ranged between 15 and 20 taxa in river channels and 16 and 17 taxa in C-38 (Table 13-20). However, species richness was 25% greater at river channel Control sites during 1998 than 1997 (Table 13-20). Unidentified sunfishes (*Lepomis* spp.) and shad (*Dorosoma* spp.) were numerically dominant in both pools and comprised 69.1% and 80.9% of larval fishes collected in Control and Impact pools, respectively.

Mean sample density was significantly greater during 1997 than 1998 at both Control (ANOVA; $p < 0.0001$) and Impact (ANOVA; $p = 0.0001$) sites, and was significantly greater at Control sites than Impact sites during 1997 (ANOVA; $p = 0.0059$), but not during 1998 (ANOVA; $p = 0.53$; Table 13-21). Mean sample density varied between lower, middle, and upper remnant river channels at Control (R-M ANOVA; $p = 0.0050$) and Impact (R-M ANOVA; $p = 0.0030$) sites. Densities typically were greater at lower sites (i.e., Kicco Run) in the Control pool, but consistently greater at middle sites (i.e., Micco Bluff

Run) in the Impact pool during 1997 (Figure 13-13). Mean sample density also differed between lower, middle, and upper regions for mid-channel (pelagic) C-38 sites at Control (R-M ANOVA; $p = 0.0115$) and Impact sites (R-M ANOVA; $p = 0.0092$); however, there was no clear pattern of differences among locations, nor was there any consistency between years (Figure 13-14). No difference was found in mean sample density between lower, middle, and upper regions among C-38 littoral sites at Control (R-M ANOVA; $p = 0.1631$) or Impact (R-M ANOVA; $p = 0.6595$) sites (Figure 13-15).

There also were differences in larval fish density among river channel, C-38 littoral, and C-38 pelagic sites. At lower Control sites, mean sample density was significantly greater at riverine sites (R-M ANOVA; $p = 0.0176$) (Figure 13-16). Densities also differed among site types at middle (R-M ANOVA; $p = 0.0062$) and upper (R-M ANOVA; $p = 0.0085$) locations of Control sites; however, mean sample density was lowest at riverine sites within both of these regions (Figure 13-16). Differences were not significant among site types at middle (R-M ANOVA; $p = 0.2002$) and upper (R-M ANOVA; $p = 0.1431$) regions of Impact sites. However, mean sample density was significantly greater at lower C-38 littoral sites (R-M ANOVA; $p = 0.0298$), than river channel or C-38 pelagic sites (Figure 13-17). Larval fish density was positively correlated, but not statistically significant, with water temperature; however, the degree of correlation varied among sites within pools (Table 13-22).

Table 13-19. Total numbers of larval fishes collected at Control and Impact sites.

		Control	Impact
LARGE-BODIED TAXA			
Catostomidae (suckers)			
Erimyzon sucetta	lake chubsucker	852	593
Centrarchidae (sunfishes)			
Lepomis spp.	unidentified sunfishes	24,142	24,897
Micropterus salmoides	largemouth bass	118	63
Pomoxis nigromaculatus	black crappie	2116	574
Clupeidae (herrings)			
Dorosoma spp.	unidentified shad	10,238	3974
Dorosoma cepedianum	gizzard shad	229	255
Cichlidae (cichlids)			
Oreochromis aureus	blue tilapia	14	8
Esocidae (pikes)			
Esox niger	chain pickerel	2	--
Ictaluridae (bullhead catfishes)			
Ameiurus natalis	yellow bullhead	9	1
Ameiurus nebulosus	brown bullhead	7	4
Ictalurus punctatus	channel catfish	1	--
Lepisosteidae (gars)			
Lepisosteus osseus	longnose gar	1	--
Lepisosteus platyrhincus	Florida gar	--	2
SMALL-BODIED TAXA			
Aphredoderidae (pirate perches)			
Aphredoderus sayanus	pirate perch	14	34
Atherinidae (silversides)			
Labidesthes sicculus	brook silverside	2538	1851
Belonidae (needlefishes)			
Strongylura marina	Atlantic needlefish	15	--
Cyprinidae (carps and minnows)			
Notemigonous crysoleucas	golden shiner	740	339
Notropis maculatus	tailight shiner	1992	80
Fundulidae (killifishes)			
Fundulus chrysostus	golden topminnow	31	60
Fundulus lineotus	lined topminnow	7	5
Lucania goodei	bluefin killifish	304	176
Percidae (perches)			
Etheostoma fusiforme	swamp darter	648	1443
Poeciliidae (livebearers)			
Gambusia holbrooki	eastern mosquitofish	1547	719
Heterandria formosa	least killifish	293	443
Poecilia latipinna	sailfin molly	1	9
	TOTAL	45,859	35,530

Table 13-20. Larval fish species collected in pushnet samples at Control and Impact sites. (cl = C-38 littoral, cp = C-38 pelagic, rc = river channel).

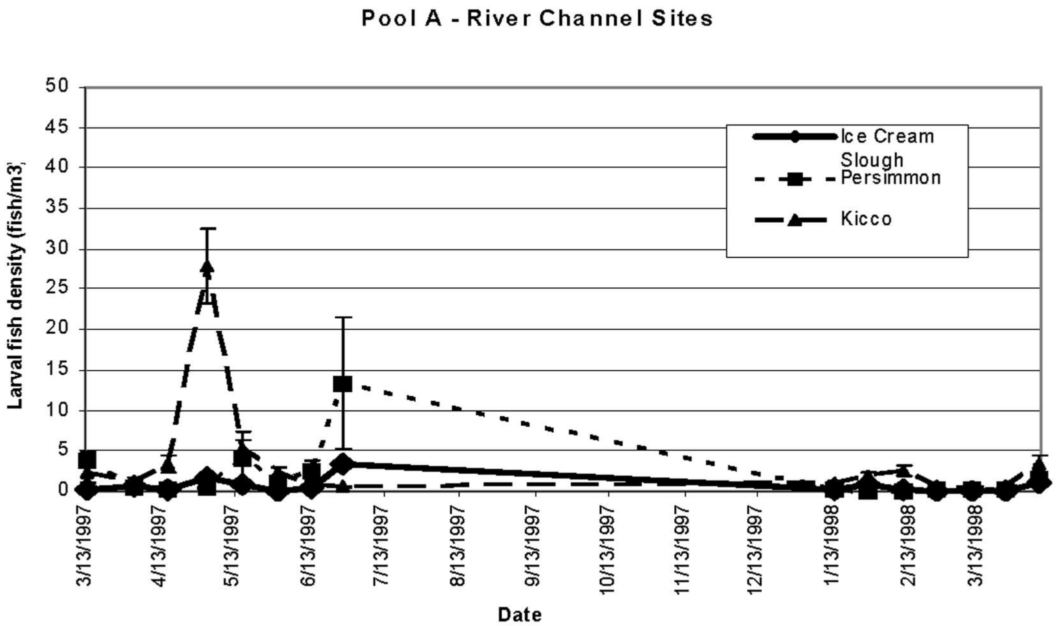
		Control Sites		Impact Sites	
		1997	1998	1997	1998
Ameiurus natalis	yellow bullhead	cl, cp, --	--, --, --	--, cp, --	--, --, --
Ameiurus nebulosus	brown bullhead	cl, cp, --	--, --, --	--, cp, rc	--, --, --
Aphredoderus sayanus	pirate perch	--, --, --	cl, cp, rc	--, --, rc	cl, cp, rc
Dorosoma spp.	unidentified shad	cl, cp, rc	--, --, --	cl, cp, rc	--, --, --
Dorosoma cepedianum	gizzard shad	--, --, --	cl, cp, rc	--, --, --	cl, cp, --
Erimyzon sucetta	lake chubsucker	--, --, --	cl, cp, rc	--, --, --	cl, cp, rc
Esox niger	chain pickerel	--, --, --	--, --, rc	--, --, --	--, --, --
Etheostoma fusiforme	swamp darter	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Fundulus chrysostus	golden topminnow	cl, cp, rc	cl, cp, rc	cl, cp, rc	--, cp, rc
Fundulus lineotus	lined topminnow	--, --, --	--, cp, rc	cl, --, --	--, --, --
Gambusia holbrooki	eastern mosquitofish	cl, cp, rc	--, cp, rc	cl, cp, rc	cl, cp, rc
Heterandria formosa	least killifish	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Ictalurus punctatus	channel catfish	--, cp, --	--, --, --	--, --, --	--, --, --
Labidesthes sicculus	brook silverside	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Lucania goodei	bluefin killifish	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Lepisosteus osseus	longnose gar	--, --, --	--, --, rc	--, --, --	--, --, --
Lepisosteus platyrhincus	Florida gar	--, --, --	--, cp, --	--, --, --	--, --, --
Lepomis spp.	unidentified sunfish	cl, cp, rc	--, cp, rc	cl, cp, rc	cl, cp, rc
Menidia berylina	inland silverside	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Micropterus salmoides	largemouth bass	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Notemigonous crysoleucas	golden shiner	cl, cp, --	cl, cp, rc	cl, cp, rc	cl, cp, rc
Notropis maculatus	tailight shiner	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Oreochromis aureus	blue tilapia	--, --, rc	--, --, rc	--, --, --	cl, cp, rc
Poecilia latipinna	sailfin molly	--, --, rc	--, --, --	cl, --, rc	--, --, --
Pomoxis nigromaculatus	black crappie	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Strongylura marina	Atlantic needlefish	--, cp, rc	--, --, rc	--, --, --	--, --, --

Table 13-21. Mean annual density (± SE) of larval fishes in pushnet samples (all habitats combined) at Control and Impact sites. Values are expressed as fish/m^3.

	Control	Impact
Year 1	5.63 ± 0.71	3.03 ± 0.59
Year 2	0.60 ± 0.20	0.46 ± 0.12

Discussion

Studies on larval fish assemblages have shown that the number of species and their relative composition generally do not reflect similar attributes of adult fish communities within the same system (Holland and Sylvester 1983, Holland 1986, Turner et al. 1994, Scheidegger and Bain 1995). Early life stages that are buoyant are collected more easily by most widely used sampling methods (e.g. push nets, seines, and towed plankton nets), and dominance of these taxa may result in misrepresentation of community structure of larval fish assemblages (Holland 1986). However, of the taxa generally collected by common sampling methods, dominance of specific taxa can be used to characterize aquatic systems as either lentic or lotic, based primarily on habitat requirements of larvae (Scheidegger and Bain 1995).



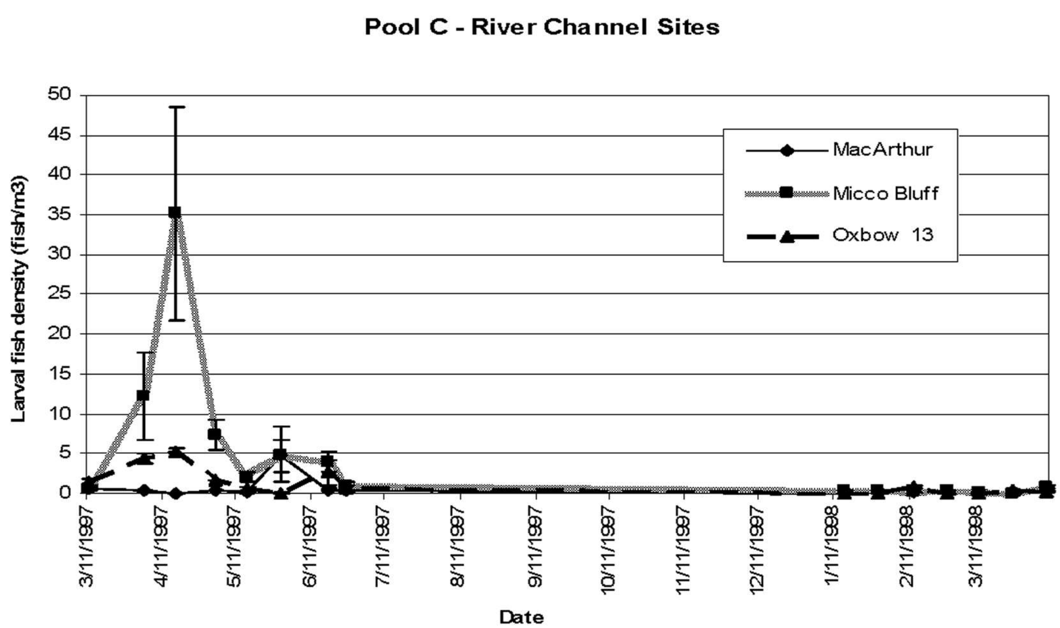
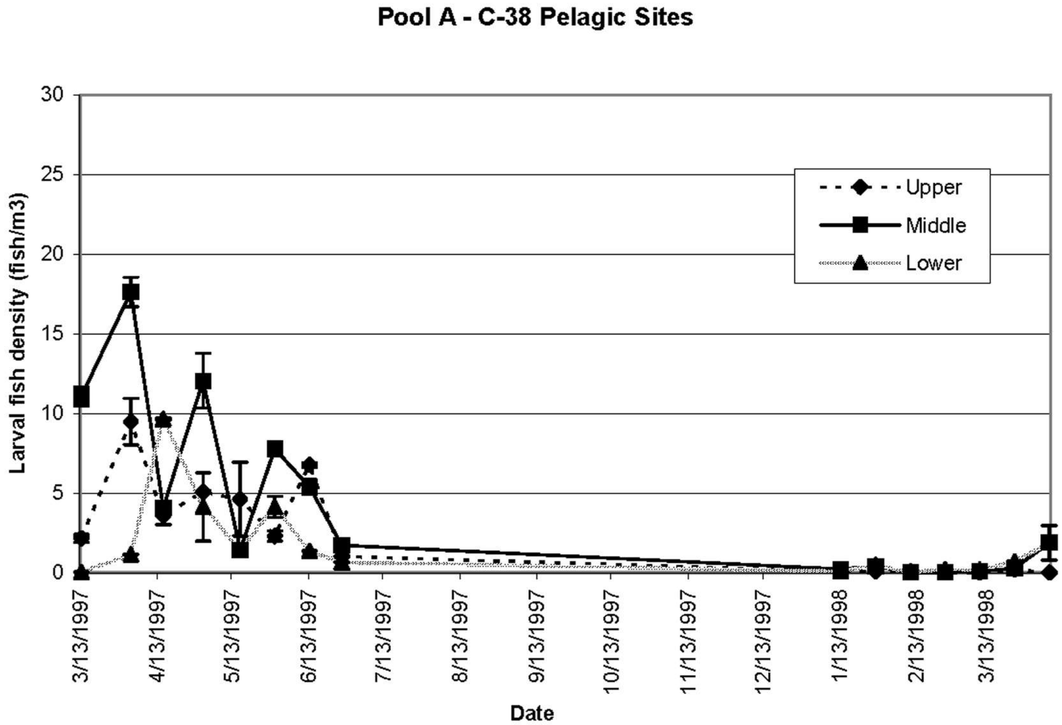


Figure 13-13. Mean larval fish density for each sampling date within remnant river channels at Control (Pool A) and Impact (Pool C) sites under channelized conditions.



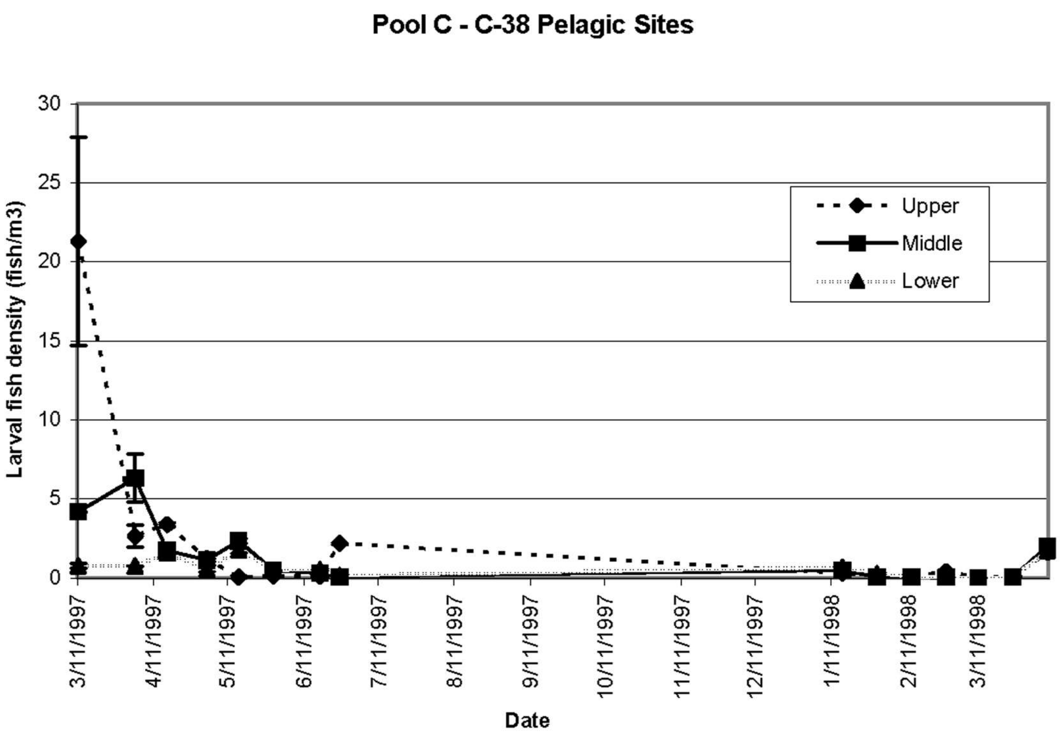
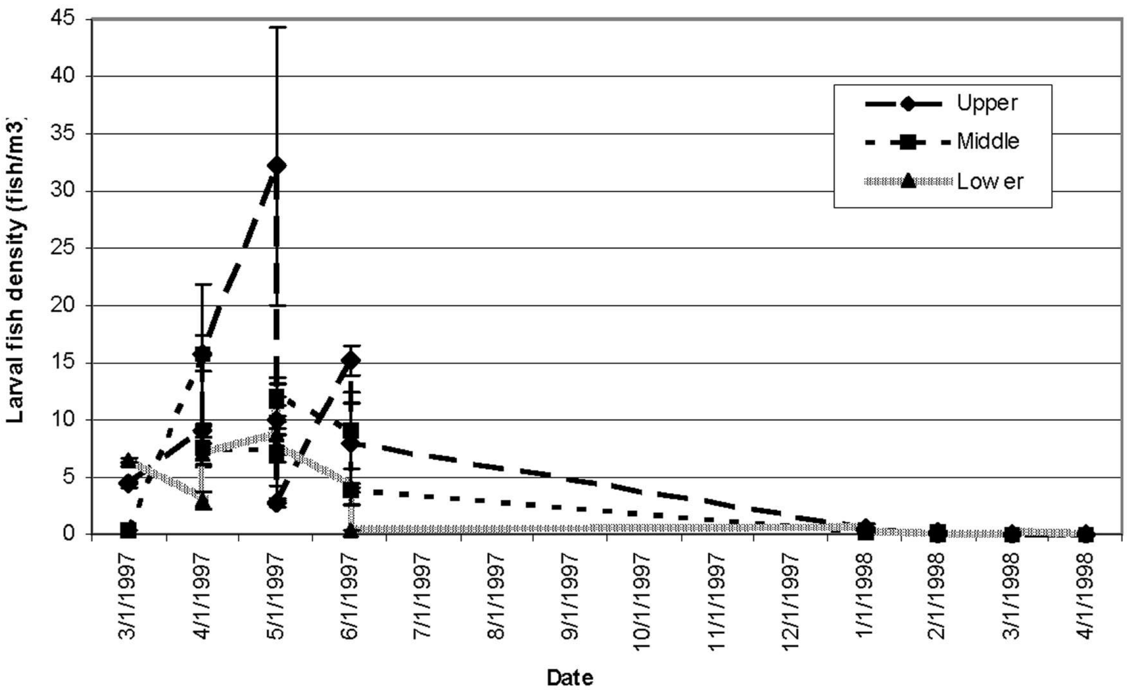


Figure 13-14. Mean larval fish density for each sampling date in upper, middle, and lower C-38 pelagic zones at Control (Pool A) and Impact (Pool C) sites under channelized conditions.

Pool A - C-38 Littoral Sites



Pool C - C-38 Littoral Sites

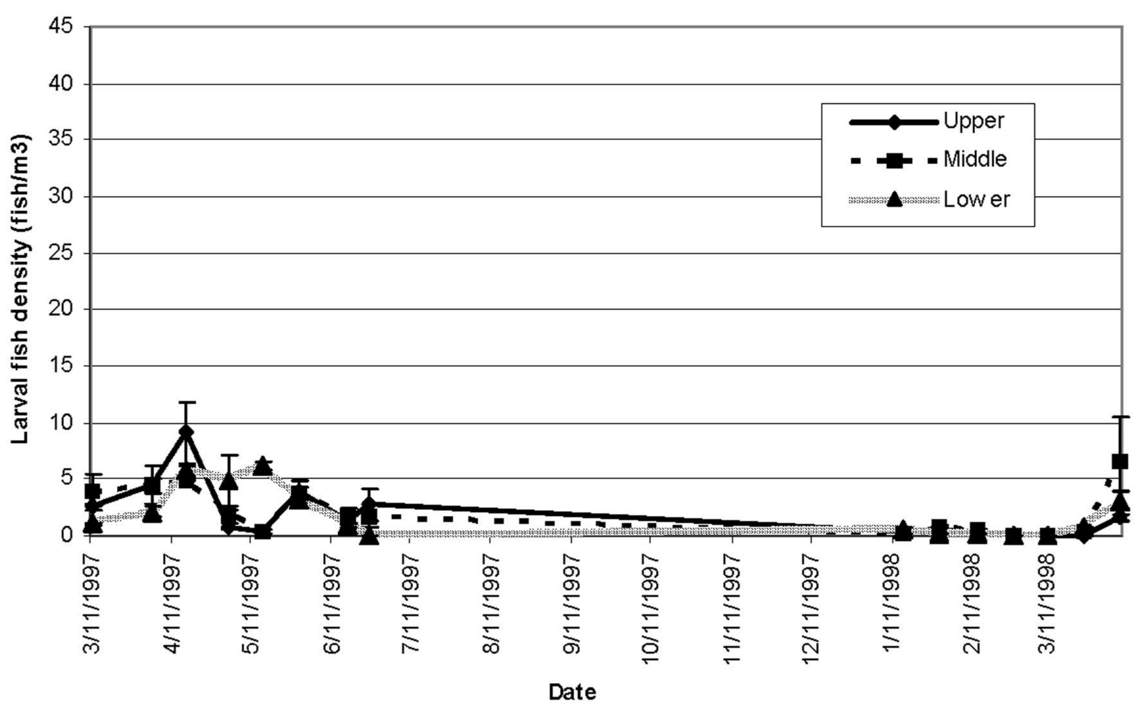


Figure 13-15. Mean larval fish density for upper, middle, and lower C-38 littoral zones at Control (Pool A) and Impact (Pool C) sites under channelized conditions.

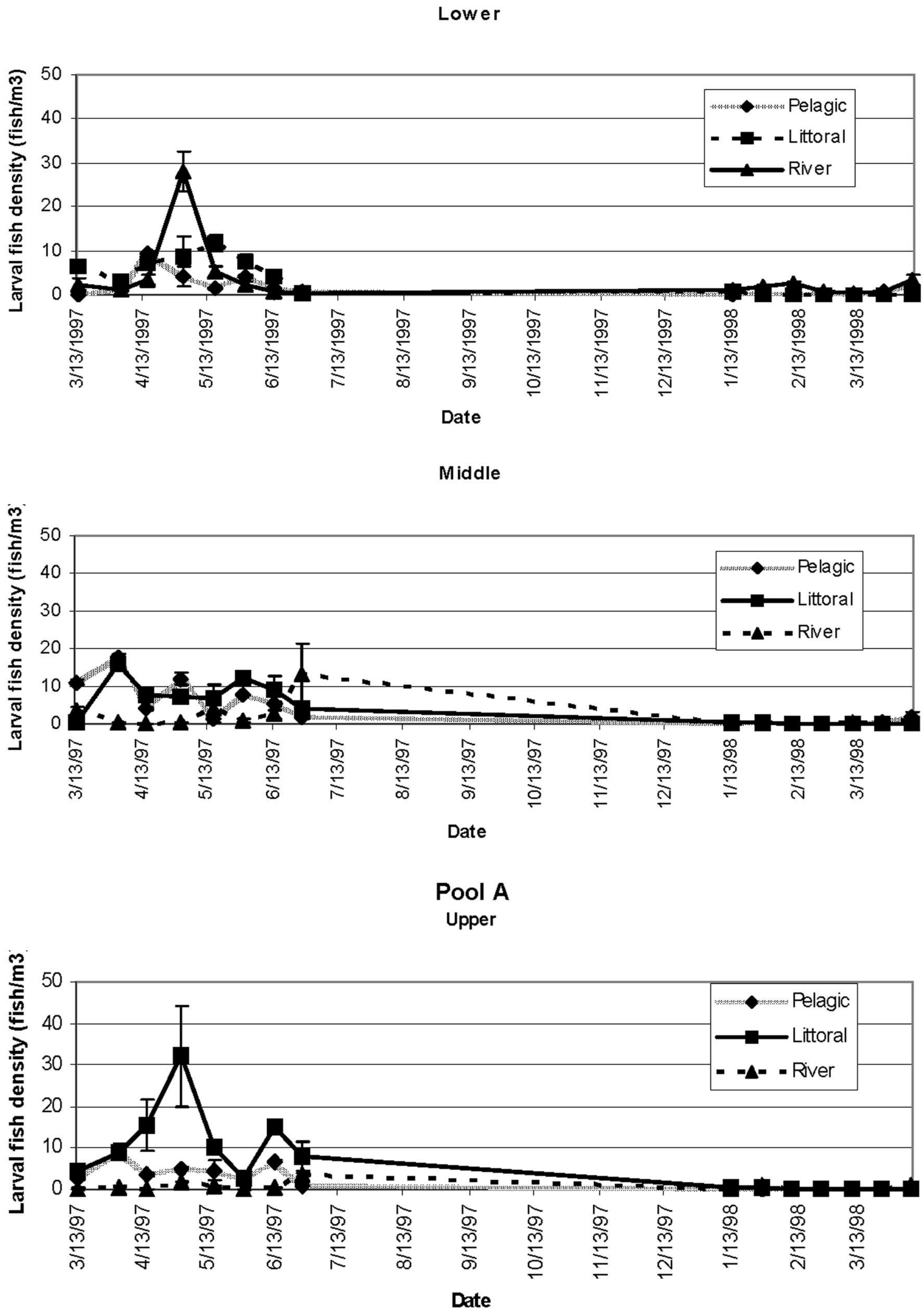


Figure 13-16. Mean larval fish density in upper, middle, and lower regions at C-38 pelagic, C-38 littoral, and river channel sites in Pool A (Control Site) under channelized conditions.