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Environmental Responses to the Kissimmee River Demonstration Project

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by

Louis A. Toth

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Environmental Sciences Division
Research and Evaluation Department
South Florida Water Management District
West Palm Beach, Florida

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EXECUTIVE SUMMARY

Between 1962 and 1970, the Kissimmee River was channelized as part of a flood control project for the Kissimmee chain of lakes region in central Florida. As a result of channelization, flow that formerly was carried by a meandering river channel and floodplain has been confined to an excavated canal (C-38).

Channelization caused numerous environmental impacts in the Kissimmee River ecosystem. These impacts stem from alteration of key determinants of ecological integrity of the river/floodplain ecosystem. In addition to direct physical destruction of river and floodplain habitat which resulted from canal excavation and deposition of spoil, channelization impacted the Kissimmee River ecosystem through altered hydrologic regimes. Ecological consequences of altered floodplain hydrology and drainage of former swamps and marshes include diminished floodplain habitat diversity, reduction of waterfowl and wading bird usage of the floodplain, and loss of habitat for both forage fish and larger riverine fish species. The nature and rate of energy exchange between the floodplain and river also has been disturbed. Elimination or modification of river/floodplain interactions has affected the functional integrity of both the river and floodplain. Other river impacts, including degraded water quality, sedimentation of river substrates, diminished habitat quality and diversity, and degradation of river biological communities have resulted from interruption of flow.

In August 1983, the legislatively created Kissimmee River Coordinating Council culminated seven years of restoration studies with a recommendation that advocated dechannelization of the Kissimmee River. In response, the South Florida Water Management District (SFWMD) took the lead role in the restoration effort by designing a Demonstration Project to provide the necessary information to implement dechannelization. The Demonstration Project was conducted in Pool B, a 19.5 km section of canal, remnant river and floodplain, between water control structures S-65A and S-65B. The project had four major components: construction of three notched weirs across C-38, implementation of a pool stage fluctuation schedule, creation of a "flow-through" marsh, and hydrologic and hydraulic modeling studies.

Prior to initiation of the Demonstration Project, the SFWMD, Florida Game and Fresh Water Fish Commission (GFC) and Florida Department of Environmental Regulation (DER) endorsed a multi-agency Memorandum of Agreement and assumed joint responsibility to monitor and evaluate environmental effects of the Demonstration Project. Staff scientists from the SFWMD, GFC and DER collaborated in the development of this joint monitoring program. The SFWMD assumed responsibility for monitoring (1) effects of Demonstration Project-related changes in the Pool B hydrologic regime on floodplain vegetation and secondary productivity and (2) effects of reintroduced flow on benthic invertebrate communities and river channel habitat characteristics. This report presents results of these studies.

Environmental monitoring data were collected between July 1984 and November 1988. To evaluate effects of hydrologic changes on floodplain vegetation, seven transects were monitored on the Pool B floodplain. These transects represented the range of Pool B floodplain vegetation communities that had developed in response to post-channelization hydrologic conditions. Effects of changes in floodplain hydrologic regimes on secondary productivity were evaluated by sampling aquatic invertebrates along six of these transects. Effects of pool stage fluctuation on fish utilization of an existing floodplain marsh also were evaluated. To assess effects of reintroduced flow on river channel habitat, bottom profiles and sediment characteristics were monitored along 31 river cross-sections. Benthic invertebrate densities and community structure were evaluated at 21 river channel sampling locations. Environmental monitoring was complemented by a network of stage recorders which provided hydroperiod and flow data during the monitoring period.

Because channelization impacted the Kissimmee River ecosystem largely by altering hydrologic regimes, an effective restoration program must re-establish hydrologic characteristics that formerly shaped the ecological structure and function of the river. Effects of the Demonstration Project on floodplain hydrology varied along the length of the pool. The greatest inundation occurred during 1987-88 when 80-90% of the floodplain in the lower 20% of the pool had inundation frequencies comparable to pre-channelization records; however, pool stage manipulations did not reproduce water depths that typically occurred on most of the floodplain prior to channelization. In the middle 40% of the pool, the combined influence of controlled stage fluctuations and backwater effects of project weirs resulted in prolonged inundation of only 20% of the floodplain, but at least 75% of the floodplain in this portion of the pool was subjected to periodic flooding. Backwater effects only slightly increased the range of stage variability, which remained considerably lower than pre-channelization ranges during all months except March and April. The influence of backwater effects on floodplain inundation was limited by the rapid rate at which water drained off the floodplain. Resultant spiked hydrographs contrasted sharply with the gradual rates at which water levels on the floodplain typically receded prior to channelization. In the northern 40% of Pool B, backwater effects of weirs periodically reflooded about 30-35% of the completely drained floodplain; however, in the Flow-Through Marsh, pre-channelization inundation frequencies were restored on approximately 35% of the floodplain, and at least 55% of this area was inundated seasonally.

River flow regimes produced by the Demonstration Project were a function of upper basin discharge characteristics and the flow diversion efficiency of notched weirs. Weirs diverted up to 60% of C-38 flow through adjacent floodplain and river channels during high discharge periods, but diverted considerably lower proportions of C-38 flow when discharges were < 28 m^3/s. This inefficiency, coupled with upper basin regulation schedules and operation rules, produced river flow regimes that contrasted greatly with key pre-channelization discharge characteristics. Highest discharges occurred during January - April, rather than during wet season months. Extended no-flow periods were common from June - December, and typical pre-channelization base flows were generated only half as frequently during the Demonstration Project.

Plant community responses to Demonstration Project components showed that restoration of wetland communities on the Kissimmee River floodplain is feasible. Monitoring data indicate that plant community composition on both drained and inundated floodplain responded to subtle, as well as, major changes in hydrologic factors, including water depths, inundation frequencies, and temporal inundation patterns. In general, hydrologic modifications produced by the Demonstration Project led to expanded distributions of hydrophytic species, particularly Alternanthera philoxeroides, Panicum hemitomon, and Polygonum punctatum, and decreased frequencies of mesophytic and xerophytic species. Some weedy species, such as Ambrosia artemisiifolia, Eupatorium capillifolium, Myrica cerifera, Sesbania punicea, and Urena lobata, persisted in areas subject to only periodic or seasonal inundation. These results indicate that many of the remaining complement of species on the channelized floodplain are sensitive to hydrologic change, and have the reproductive potential, including a viable seed bank, to rapidly colonize and expand their distribution into habitats with favorable hydrology.

The Demonstration Project also provided evidence of the feasibility of restoring several components of floodplain function, including waterfowl and wading bird utilization, small fish and invertebrate productivity, and processes that could enhance water quality. However, because key hydrologic characteristics were not adequately reestablished, most functional aspects of floodplain ecosystem integrity were affected temporarily and/or only partially restored. Pool stage manipulations led to expansion by resident, forage fish populations, but water levels in the marsh apparently did not get deep enough, or were not deep long enough, to accomodate utilization of the marsh by larger, riverine fish species. Invertebrate colonization of reflooded portions of drained floodplain, as well as existing marsh habitat, was rapid, but appeared to produce an incomplete complement of trophic guilds. Recolonization was most successful in reflooded areas that were in close proximity to, or hydraulically connected with, aquatic habitats. Highest floodplain invertebrate densities were found in a periodically reflooded section of floodplain which was surrounded by broadleaf marsh. High invertebrate densities also were found in floodplain habitats where colonization was facilitated by overbank flow.

Results of Demonstration Project monitoring also indicate that restoration of ecological integrity of the river channel is possible. Reintroduction of flow and associated fluvial processes enhanced diversity and quality of degraded river habitat by restoring natural substrate characteristics and channel morphology. A predominantly sand substrate was restored through gradual flushing and/or covering of organic deposits, without any detectable impacts on water quality. Reintroduced flow also led to reestablishment of benthic invertebrate species composition with at least rudimentary characteristics of a natural (pre-channelization) river invertebrate community. Chaoborus punctipennis temporarily was eliminated as the dominant mid-channel invertebrate species, and runs with restored flow were colonized by Sphaeriacean clams. Density and diversity of benthic invertebrates were greater in samples from the littoral zone than in the center of the channel, and were enhanced by reintroduced flow, particularly during spring months. Although reintroduced flows clearly provided precursors of ecological integrity, the inadequacy of current river inflows precluded more meaningful and complete restoration. No-flow periods continued to contribute to low dissolved oxygen regimes in river runs adjacent weirs and prolonged periods of no flow tended to reverse any restoration progress. Densities of Sphaeriacean clams declined and high densities of Chaoborus reappeared following a three-month period of no flow during summer 1987.

Monitoring results indicate that ecological integrity of the Kissimmee River can be restored only with a holistic approach which succeeds in restoring both the form and function of the former ecosystem. Integration of these monitoring results with hydrologic modeling studies firmly establishes that restoration of the Kissimmee River ecosystem requires backfilling of long, continuous reaches of C-38. The primary recommendation of this study is to implement the Level II Backfilling Plan endorsed by Governor Martinez and the SFWMD Governing Board.

Kissimmee River restoration could be an unprecedented project of global significance. The scope of its value will be determined largely by the quality and rigor of the ecological studies and monitoring program that are conducted in association with the restoration project. Ecological monitoring studies must be the "heart" of this model restoration program. Demonstration Project results provided refined direction and guidelines for required ecosystem monitoring studies. In preparation for implementation of Part 1 of the restoration program, baseline data on all components of the ecosystem, including wading birds, waterfowl, fisheries, fish communities, habitat, water quality, and ecosystem function, should begin immediately. Two to three years of pre-construction studies, followed by a five year, post-construction evaluation phase are recommended.

Interim measures should include development of a modified pool stage fluctuation plan, and floodplain levee degradation and ditch filling projects.

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ACKNOWLEDGEMENTS

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INTRODUCTION

Between 1962 and 1970, the Kissimmee River was channelized as part of a flood control project for the Kissimmee chain of lakes region in central Florida (Figure 1). As a result of channelization, flow formerly carried by 166 km of meandering river channel and a 1.5 - 3 km wide floodplain, has been confined to a 90 km canal (C-38), which has average depth of 9 m and surface width of 64 - 105 m (U.S. Army Corps of Engineers, 1985). Beginning at the outlet of Lake Kissimmee, six water control structures (S-65, S-65A, S-65B, S-65C, S-65D, and S-65E) with tieback levees were constructed along the canal's length. These structures act as dams and replaced the natural slope of the river with five stair-step impoundments or pools (Figure 2). Relatively constant water levels have been maintained upstream of each structure, permanently inundating the lower end of each pool and completely draining the upper end.

Channelization caused numerous environmental impacts in the Kissimmee River ecosystem, including a tremendous loss of biological resources (Dineen et al., 1974; Perrin et al., 1982). These impacts and losses stem from modifications of key determinants of ecological integrity of the river/floodplain ecosystem.

Floodplain impacts have resulted from alterations of hydrologic regimes, as well as, physical destruction of wetland habitat. Between 12,000 - 14,000 of the original 18,000 ha of pre-channelization floodplain wetlands were either completely drained, covered with spoil, or converted to canal (Pruitt and Gatewood, 1976; U.S. Army Corps of Engineers, 1985). Most of the broadleaf marsh and wet prairie communities that once dominated the floodplain (Pierce et al., 1982) have been converted to unimproved and improved pasture (Milleson et al., 1980), while maintenance of stable water levels has reduced plant species diversity and eliminated spatial heterogeneity of wetland plant communities within remaining inundated portions of each pool (Goodrick and Milleson, 1974).

Impacts of channelization on these wetlands have had wide-ranging ecological consequences, including loss of fish and wildlife habitat and virtual destruction of a complex food web that the floodplain wetlands once supported. For example, since channelization there has been a 92-94 percent reduction in the 4000-5000 wintering and resident waterfowl which once utilized the lower Kissimmee basin (Perrin et al., 1982). This decline appears to be due to loss of deep, open water habitat adjacent the river channel, as well as, the extensive wet prairie which existed along the periphery of the floodplain (Chamberlain, 1960). Drainage of floodplain wetlands and maintenance of stable pool stages has eliminated plant community and habitat diversity that is necessary to attract and support large waterfowl populations (Wheeler and March, 1979; Murkin et al., 1982).

Loss of wetland habitat diversity also accounts for limited post-channelization usage of the floodplain by wading birds (Perrin et al., 1982). Prior to channelization, wading birds were provided accessible and concentrated forage in seasonally inundated wet prairie, which were afforded a constant source of fish and invertebrate colonists from adjoining marshes. Remaining floodplain wetlands do not provide favorable feeding habitat for wading birds because vegetation within the existing broadleaf marshes is too dense, and/or water levels are too deep, for efficient foraging activity (see Kushlan and Kushlan, 1975).

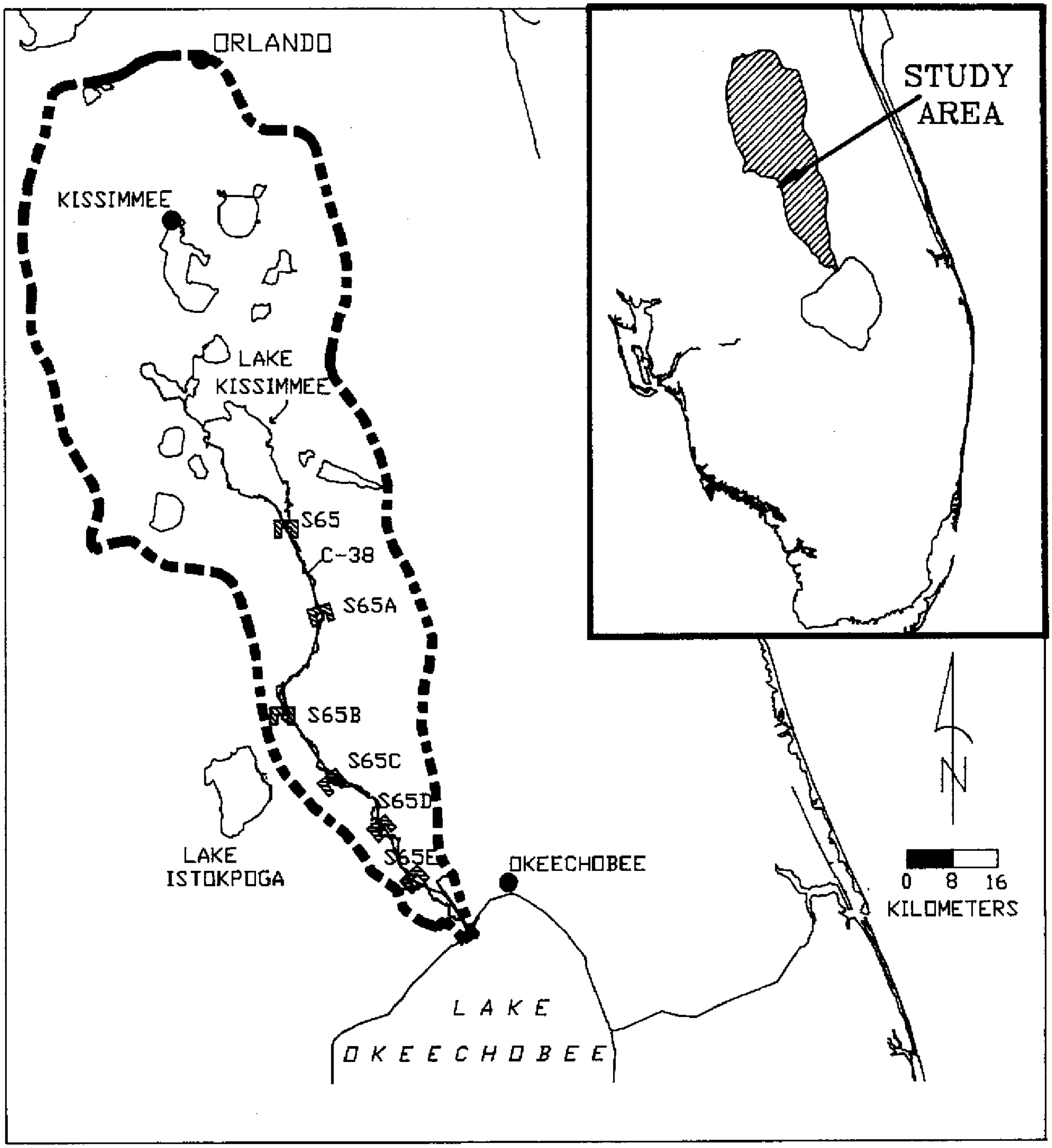

FIGURE 1. Map of study area showing C-38 and Kissimmee chain of lakes.

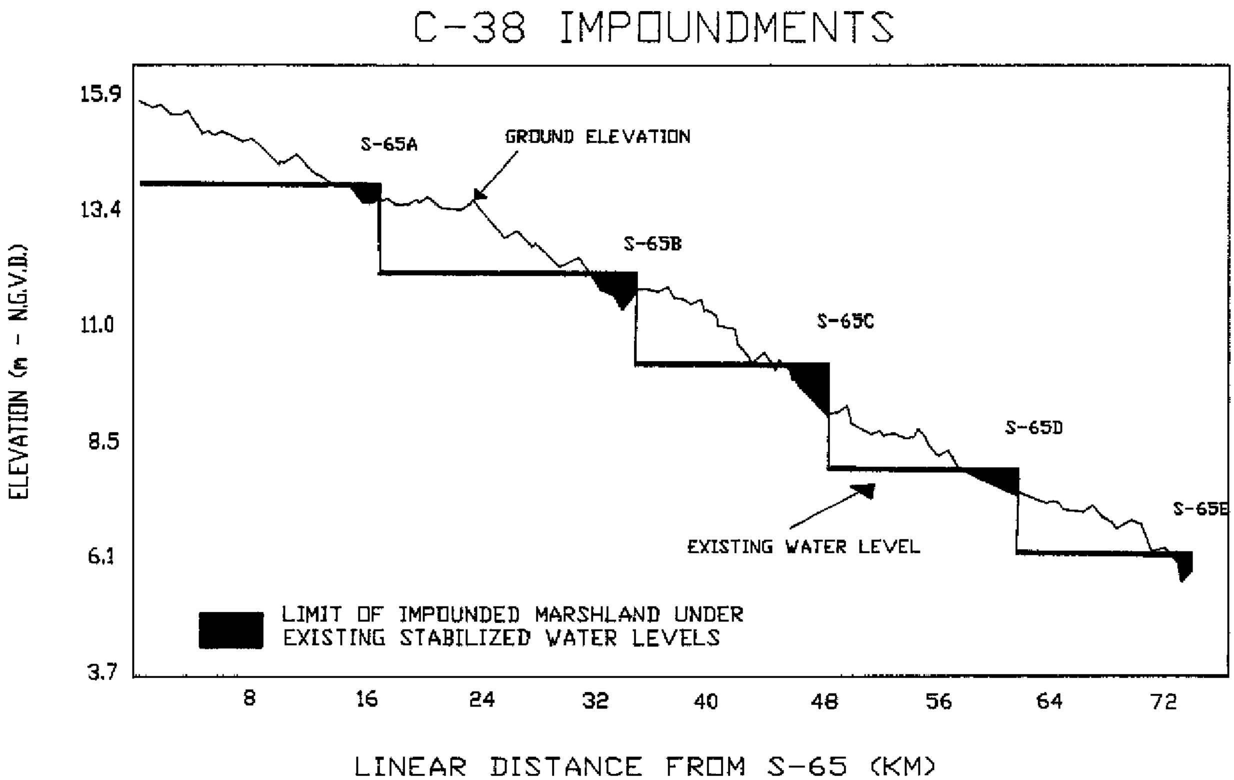

FIGURE 2. Diagram of C-38 system showing water surface profile relative to ground elevations in the Kissimmee River valley.

Drainage of floodplain wetlands also resulted in tremendous losses of fish and invertebrate production. Based upon average densities in remaining marshes (Milleson, 1976), over five billion small fish (mostly Gambusia affinis and Heterandria formosa) and six billion freshwater shrimp (Palaemonetes paludosus) occurred in the 14,000 ha of floodplain marsh that were drained by channelization. In addition to providing forage for wading birds, these small fish and invertebrates were an important food source for riverine fishes. Kissimmee River marsh samples (Florida Game and Freshwater Fish Commission, 1957; Milleson, 1976) indicate that most river fishes, including game fish species, utilized wetland resources on the floodplain during at least part of their life cycle. Moreover, when water levels receded, fish species in the river fed upon small fish and invertebrates that were imported from adjoining marsh habitats. However, because this transfer of organisms was most significant during receding stages, when water drained off the floodplain, maintenance of stable water levels has restricted this important trophic/energetic interaction between the river and floodplain.

As in the floodplain, channelization has had both direct and indirect effects on river channel habitat and associated biota. Approximately 56 km of former river were obliterated by the excavation of canal and deposition of spoil, while discontinuance of flow has resulted in severe habitat degradation in the remaining 109 km of river channel. Low dissolved oxygen regimes are indicative of effects of lack of flow on habitat quality of remnant river channels. During summer and fall months, dissolved oxygen concentrations in the river and canal frequently fall well below 3 mg/l (Federico, 1982; Perrin et al., 1982). Lack of flow-related hydrodynamic processes also has resulted in decreased depth diversity along river cross-sections and accumulations of thick deposits of decomposing organic matter on the river bottom.

These deposits have been produced by continuous sloughing of emergent and floating vegetation, and generate a high biochemical oxygen demand that contributes to prevailing low dissolved oxygen conditions in remaining river runs. Degradation of river habitat has been exacerbated by repetitive herbicide applications, which have been required to prevent emergent (e.g., Nuphar luteum) and floating (Eichornia crassipes and Pistia stratiotes) vegetation from choking old river channels.

Effects of channelization on dissolved oxygen regimes and river habitat diversity are primary causes of degradation of river biological communities. This includes a decline in the largemouth bass fishery and extirpation of six indigenous fish species from the river system (Perrin et al., 1982). The food base of river fish communities also has been impacted. Benthic invertebrate communities in the canal and remaining river sections are characteristic of a reservoir rather than riverine environment (Toth, 1988). Bottom habitat in both the canal and remnant river runs support low invertebrate densities and diversity, and are dominated by organisms that are tolerant of degraded habitat conditions. In addition to low dissolved oxygen concentrations, unsuitable substrates, and reduced habitat diversity, river invertebrate communities have been subjected to altered energy inputs. Due to hydrologic changes, wax myrtle (Myrica cerifera) replaced willow (Salix caroliniana) as a dominant riparian species and source of allochthonous organic matter inputs along much of the remaining river channel (personal observation). This represents a shift in the energy base with which the pre-channelization river invertebrate community and associated food chain co-evolved.

In summary, in addition to direct physical destruction of river and floodplain habitat which resulted from canal excavation and deposition of spoil, channelization has impacted the Kissimmee River ecosystem through altered hydrologic regimes. Ecological consequences of altered floodplain hydrology and drainage of former swamps and marshes include diminished floodplain habitat diversity, reduction of waterfowl and wading bird usage of the floodplain, and loss of habitat for forage fish and larger riverine fish species. The nature and rate of energy exchange between the floodplain and river also has been disturbed. Elimination or modification of river/floodplain interactions has affected the functional integrity of both the river and floodplain. River channel impacts, including degraded water quality, sedimentation, diminished habitat quality and diversity, and degradation of biological communities have resulted primarily from interruption of flow.

In recognition of this environmental degradation, in 1976 the Florida legislature passed the Kissimmee Restoration Act (Chapter 76-113, Florida Statutes) and established the Coordinating Council on the Restoration of the Kissimmee River Valley and Taylor Creek-Nubbins Slough Basin. This legislation directed the Coordinating Council to develop measures to alleviate the impacts of channelization and restore the environmental values of the Kissimmee River ecosystem. During the ensuing seven years, the Coordinating Council provided a forum for technical evaluation and public review of Kissimmee River restoration needs, concepts and options. In August 1983, the Coordinating Council recommended dechannelization of the Kissimmee River by partial backfill of C-38, and called for the South Florida Water Management District (SFWMD) to be the lead agency in the restoration program. In response, the SFWMD designed a Demonstration Project which was intended to provide the necessary information to implement dechannelization. The SFWMD also endorsed a multi-agency Memorandum of Agreement concerning restoration of the Kissimmee River, and along with the Florida Game and Freshwater Fish Commission (GFC) and Florida Department of Environmental

Regulation (DER), assumed joint responsibility to monitor and evaluate environmental effects of the Demonstration Project.

The Demonstration Project was conducted in Pool B, a 19.5 km long section of canal, remnant river and floodplain, between S-65A and S-65B in Osceola, Okeechobee and Highlands Counties (Figure 3). The project had four major components: construction of three notched weirs across C-38, implementation of a pool stage fluctuation schedule, creation of a "flow-through" marsh, and hydrologic and hydraulic studies. The purpose of the weirs was to simulate effects of backfilling the canal by diverting flow through adjacent remnant river runs. Installation of weirs also was expected to produce some additional floodplain inundation as water was detained upstream of these structures during discharge periods. Pool stage fluctuation was implemented as a means to counter the loss and degradation of wetland habitat that had resulted from adherence to stable pool stages. An annual 11.9 - 12.8 m (39 - 42 ft) N.G.V.D. fluctuation schedule (Figure 4) was used to seasonally inundate approximately 526 ha of the Pool B floodplain that have ground elevations between 12.2 - 12.8 m. A "flow-through" marsh concept was employed to recreate approximately 121 ha of marsh in the northeast section of the Pool B floodplain, where ground elevations are higher than the peak stage of the fluctuation schedule. This impoundment was created by installing a culvert in the S-65A tieback levee and constructing a berm along C-38 (Figure 3). The culvert was intended to deliver water into the north end of the "flow-through" marsh, where it would begin to flow overland, and eventually drain into, and provide flow through, a remnant river run. The upstream connection of this run with C-38 was blocked with an overflow weir to prevent inflows from the "flow-through" marsh and adjoining tributary from "short-circuiting" to the canal. Hydrologic and hydraulic studies evaluated engineering feasibility of dechannelization, flood control implications, and sedimentation issues (Loftin et al, 1990; Shen et al., 1990).

Prior to initiation of the Demonstration Project, staff scientists from the GFC, DER and SFWMD collaborated in the design of a joint monitoring program. The DER assumed responsibility for monitoring water quality and aquatic macroinvertebrate and periphyton responses. The GFC agreed to conduct alligator counts, waterfowl and wading bird surveys, fish population sampling, and evaluate effects on littoral plant communities and river bottom contours. The SFWMD monitoring program included studies to (1) monitor effects of changes in the Pool B hydrologic regime on floodplain vegetation and secondary production and (2) evaluate effects of reintroduced flow on benthic invertebrate communities and river channel habitat characteristics. All monitoring studies were to be conducted for two years following completion of structural components of the Demonstration Project.

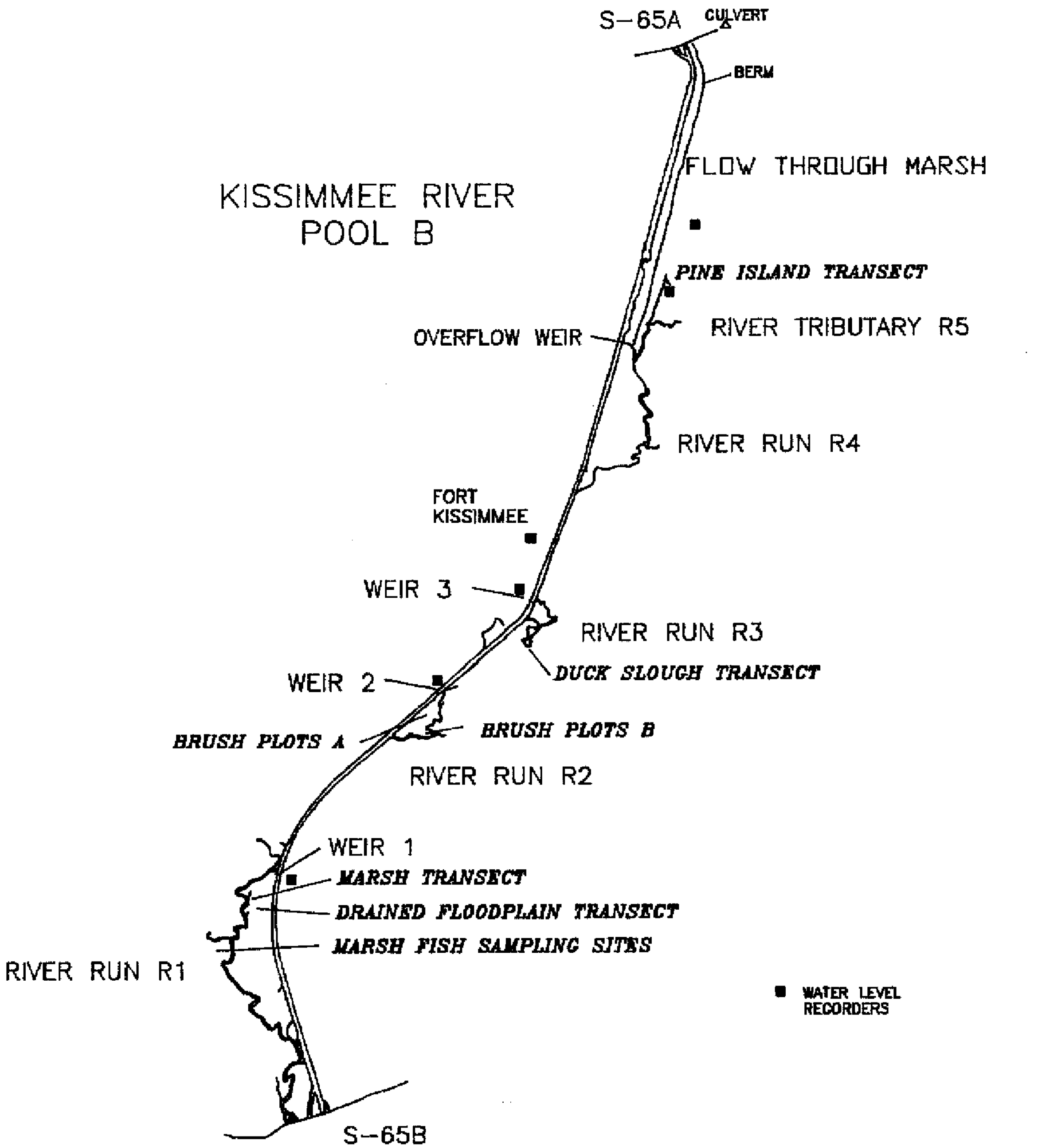

FIGURE 3. Map of Pool B showing Demonstration Project components, water level recorders, and sampling locations.

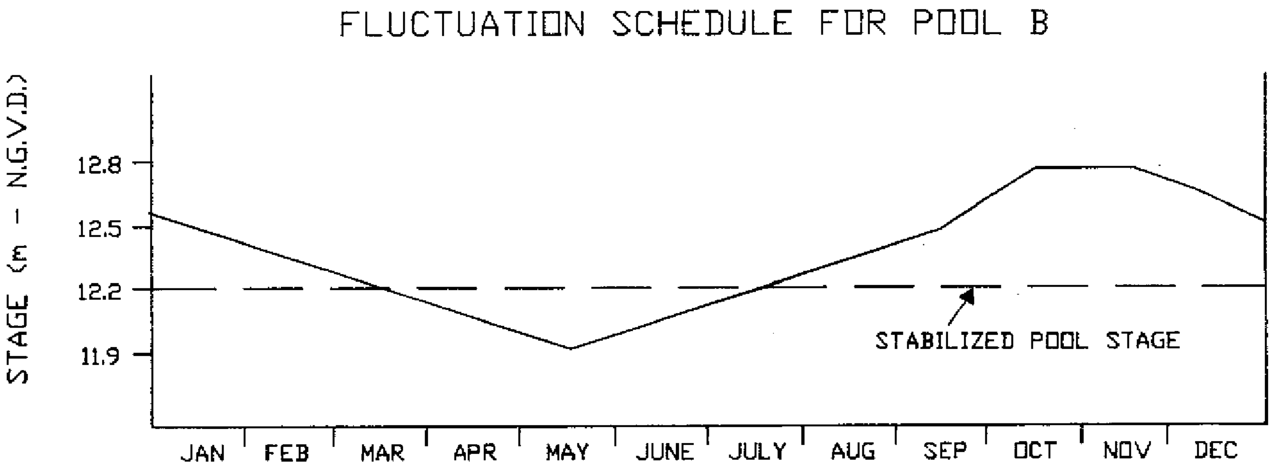

FIGURE 4. Pool stage fluctuation schedule for Pool B Demonstration Project.

METHODS

HYDROLOGY

Pre-channelization hydroperiods were based upon historic (January 1942 - September 1967) stage data from the U.S.G.S. gaging station at Fort Kissimmee (Figure 3)[1]. Discharge data (1934-60) from the U.S.G.S gaging station at the outlet of Lake Kissimmee were used in analyses of pre-channelization discharge characteristics.

Pool B stage characteristics from 1969-1988 were based primarily on data collected by continuous stage recorders located at S-65B (upstream stage) and S-65A (downstream stage). These data also were used to derive floodplain hydroperiod characteristics for the pre-demonstration project period (i.e., 1969-1984). Floodplain hydroperiod data for 1985-1988 were generated primarily from stage records taken by water level recorders that were installed during the course of the Demonstration Project (see Figure 3 for locations of Pool B stage recorders).

Following construction of weirs, discharge through adjacent river runs (R1-3) was estimated as the difference between discharge in C-38 (based upon S-65A inflow and S-65B outflow) and flow through the notch in each weir. Because these estimates do not account for overbank or floodplain flow, the data overestimate discharges through R1-3, particularly during high flow periods. Thus, broad discharge categories were used to characterize flows through these runs.

FLOODPLAIN VEGETATION

To evaluate effects of Demonstration Project-related changes in the hydrologic regime on floodplain vegetation, seven transects were established on the Pool B floodplain (Figure 3). These transects represented the range of Pool B floodplain vegetation communities which had developed in association with post-channelization hydrologic conditions. This included transects through an existing marsh at the lower end of the impoundment (Marsh Transect), marsh/wet prairie communities influenced by inflows from tributary sloughs (Pine Island Slough and Duck Slough Transects), and drained floodplain dominated by herbaceous (Drained Floodplain and Flow-Through Marsh Transects) and woody (Brush Transects A and B) plant species.

Transects were monitored annually, during July or August, from 1984-88. Monitoring of the Marsh, Pine Island Slough, Duck Slough, Drained Floodplain and Flow-Through Marsh Transects was conducted by documenting all plant species found within 1 m^2 quadrats, spaced at 7.6 m intervals along the transects. Along each Brush Transect, plant species were documented at ground level and in the combined subcanopy and canopy ($>$ 2 m above the ground surface), in nine 1 m^2 subplots within five permanent 9 m^2 plots.

To facilitate hydroperiod calculations, ground elevations were surveyed (nearest 3 cm) at each brush plot, and at 7.6 m intervals along all seven transects.

1 Appropriateness of this "index" location was verified by similar floodplain inundation percentages at the Basinger gaging station (Obeysekera and Loftin, 1988).

FLOODPLAIN FISH

Effects of pool stage fluctuations on fish utilization of an existing floodplain marsh were evaluated in a broadleaf marsh adjacent R1 (Figures 3 and 5). Because ground elevations within this area ranges from 12.1 - 12.2 m, most of this marsh had remained inundated since channelization. Dominant plant species in the marsh were Panicum hemitomon, Leersia hexandra, Sagittaria lancifolia, and Pontederia cordata.

Fish were sampled at monthly intervals from September - February in 1985-86 and 86-87, and from September - April in 1987-88. During each of these periods, samples were taken during rising, falling and peak points of stage hydrographs. On each date, rotenone samples were taken from three 9.29 m^2 plots[2], which were enclosed by 0.32 cm mesh block nets, at locations approximately 25, 100, and 150 m from R1. Fish were collected after all emergent vegetation within each plot was harvested. Removal of vegetation facilitated an even distribution of rotenone and allowed dead fish to float unobstructed to the surface. On the day after rotenone applications, submerged litter was harvested with dip nets and thorougly examined for fish that had sunk to the bottom.

FLOODPLAIN INVERTEBRATES

Effects of changes in floodplain hydrologic regimes on secondary production were evaluated by sampling aquatic invertebrates along all plant transects except Pine Island Slough. Because each transect had somewhat different inundation characteristics during the monitoring period, the frequency and timing of sampling intervals varied among transects. Sampling dates for the Marsh Transect were based upon the pool stage fluctuation schedule, and timed to occur during the period of peak inundation (September - October), and near the end of the receding leg of the stage hydrograph (March - April). Sampling along this transect was initiated in September 1984. A similar sampling strategy was adopted for the Brush Transects. However, because these transects were not inundated until the weirs were constructed, and only had short periods of inundation, sampling generally was limited to January and February, and did not begin until January 1986. The last invertebrate samples were taken from the Marsh and Brush Transects in April 1988. Due to even shorter periods of inundation, samples were taken along the Duck Slough Transect only in January 1986, and February and November 1987, and along the Drained Floodplain Transect in December 1987. Inundation of the Flow-Through Marsh first occurred in summer 1987, and aquatic invertebrate samples were collected in January and December 1987 and April 1988.

On each sampling date, 3-5 samples were collected at evenly spaced intervals along inundated portions of the transect. Each sample was collected from 0.196 m^2 of inundated floodplain that was enclosed within a 77 cm high, 50 cm diameter polyethylene cylinder which was pushed into the substrate. Samples were taken by vigorously swirling a 500 micron mesh dip net throughout the enclosed water column for approximately 1 minute. Though not exhaustive, samples were representative (at least) of combined relative abundances of invertebrates occurring on live and dead submerged vegetation, as well as, organisms found within the water column and unconsolidated substrate (plant litter). Smock et al. (1981) used a similar technique to sample littoral invertebrate communities of a Georgia lake. Samples were washed

[2] Only one plot, located approximately 20 m from R1, was sampled during September 1985.

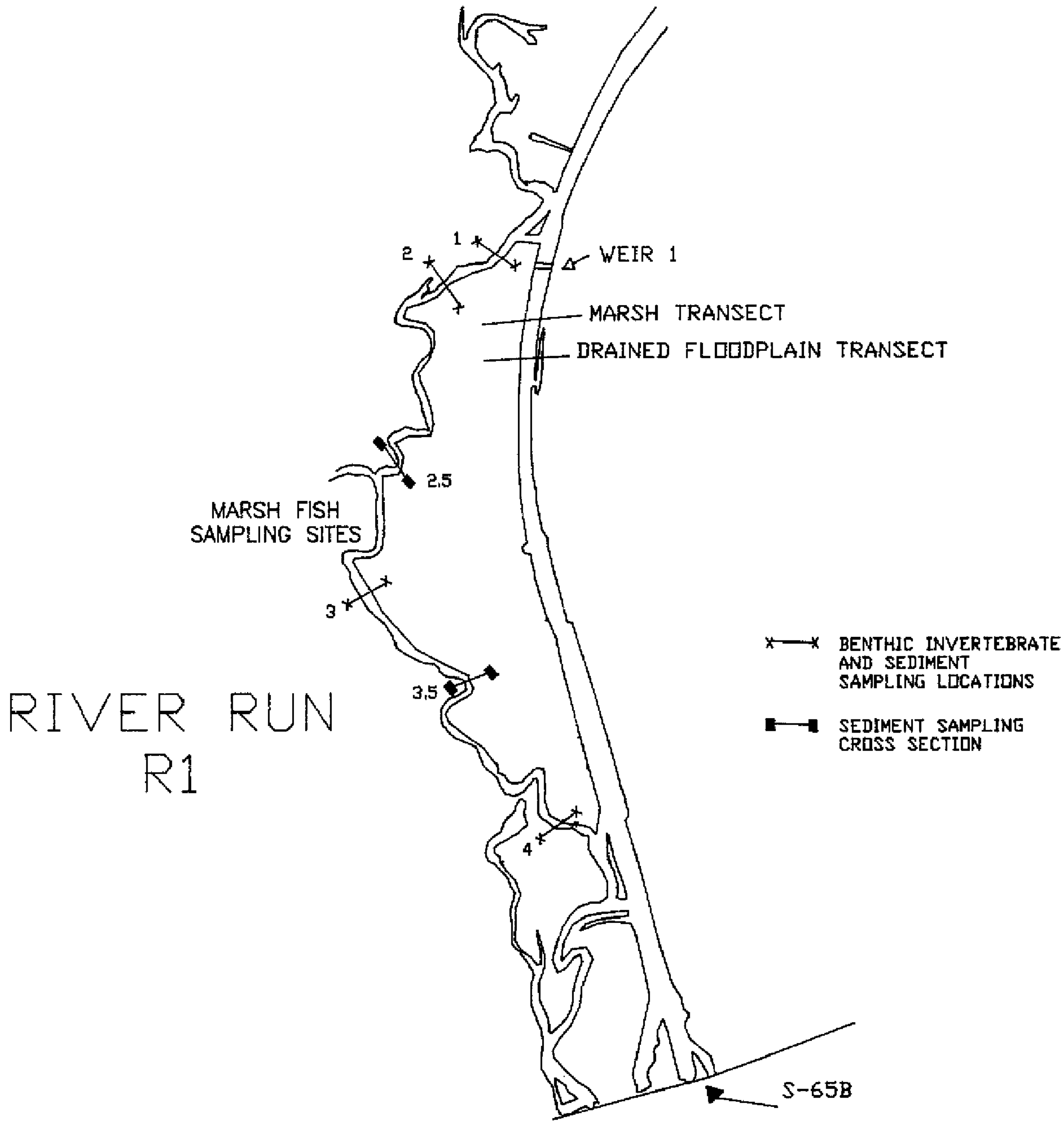

FIGURE 5. Map of river run adjacent Weir #1, showing fish, sediment and benthic invertebrate sampling sites and locations of Marsh and Drained Floodplain transescts.

in a 516 micron mesh sieve bucket and preserved in 95 percent ethyl alcohol. In the laboratory, samples were divided (*f.* Waters, 1969) into either 8 or 32 subsamples, depending upon the amount of detritus in the sample. Reported estimates of sample densities were derived by extrapolating means of four 1/8 or 1/32 subsamples.

The ground elevation of each sample was determined by measured water depths (nearest cm) within the enclosed sample area and stage data from that sampling date. These elevations were compared with previous stage records to estimate the number of days that the sampled area was inundated prior to sampling.

RIVER CHANNEL HABITAT CHARACTERISTICS

To assess effects of reintroduced flow on river channel habitat, bottom profiles and sediment characteristics were monitored along 31 cross-sections in R1-4 (Figures 5-8). Point sediment samples were taken from a stationary boat, at 1.5 m intervals along each cross-section, using a 144 cm long, 6.2 cm diameter, Plexiglas sediment corer equipped with threaded extension pipe. At each point, the sediment corer was slowly lowered down the water column and, if present, through the soft, organic matter layer, until firm substrate was encountered. After this depth (nearest cm) was recorded, the corer was pushed further into the substrate to seal the core, and then lifted back onto the boat where measurements of the length (depth) of surface organic deposits were taken.

Initial (baseline) samples were taken from November 1984 - July 1985, prior to reintroduction of flow, and at annual intervals during October - November from 1985-88.

RIVER CHANNEL INVERTEBRATES

Benthic invertebrate densities and community structure were evaluated at four sampling sites in R1,2,4 and 5, and five sites in R3 (Figures 5-8). Initial samples were taken during July - August 1984 from mid-channel locations at all five sites in R3 (six replicate samples/site), and sites 1 and 4 in R5 (four replicate samples/site). All subsequent sampling was conducted in December, April and August, through 1987, and consisted of two samples from both mid-channel and near-bank locations at all sites. Near-bank samples typically were taken within littoral vegetation, 0.5 - 2 m from each (opposite) shoreline. Mid-channel samples were taken outside the littoral zone. Samples were collected with an Ekman dredge (0.023 m^2) washed in a 516 micron mesh sieve bucket, and preserved in 95 percent ethyl alcohol. In the laboratory, samples with a large amount of detrital material were divided into 8 subsamples (*f.* Waters, 1969), and total sample densities were derived by extrapolating means of four subsamples.

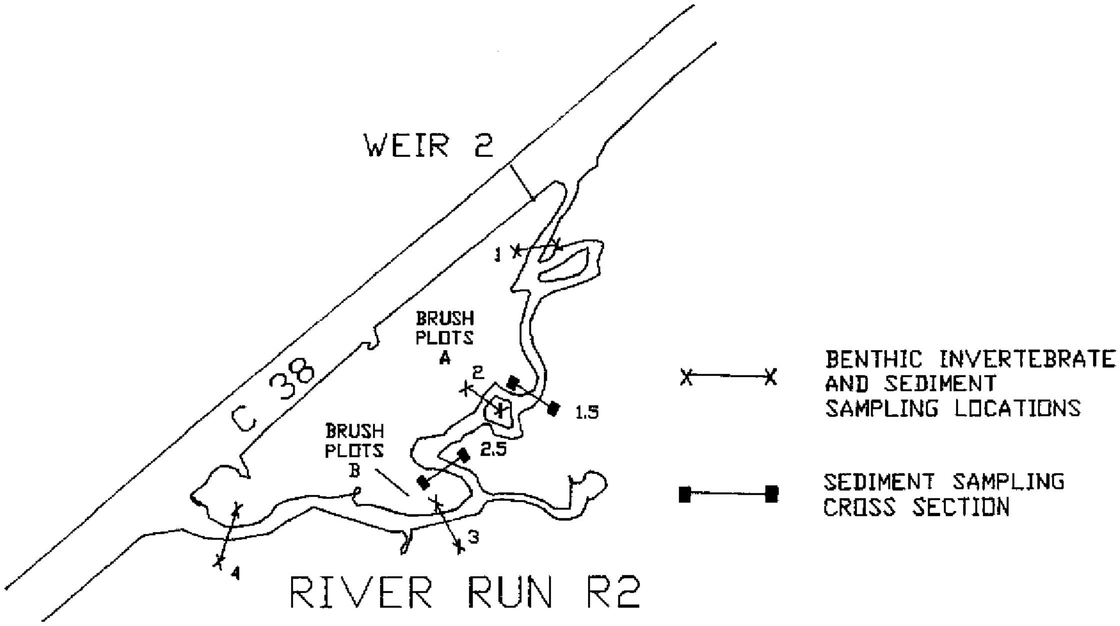

FIGURE 6. Map of river run adjacent Weir #2, showing sediment and benthic invertebrate sampling sites and locations of Brush Plots A and B.

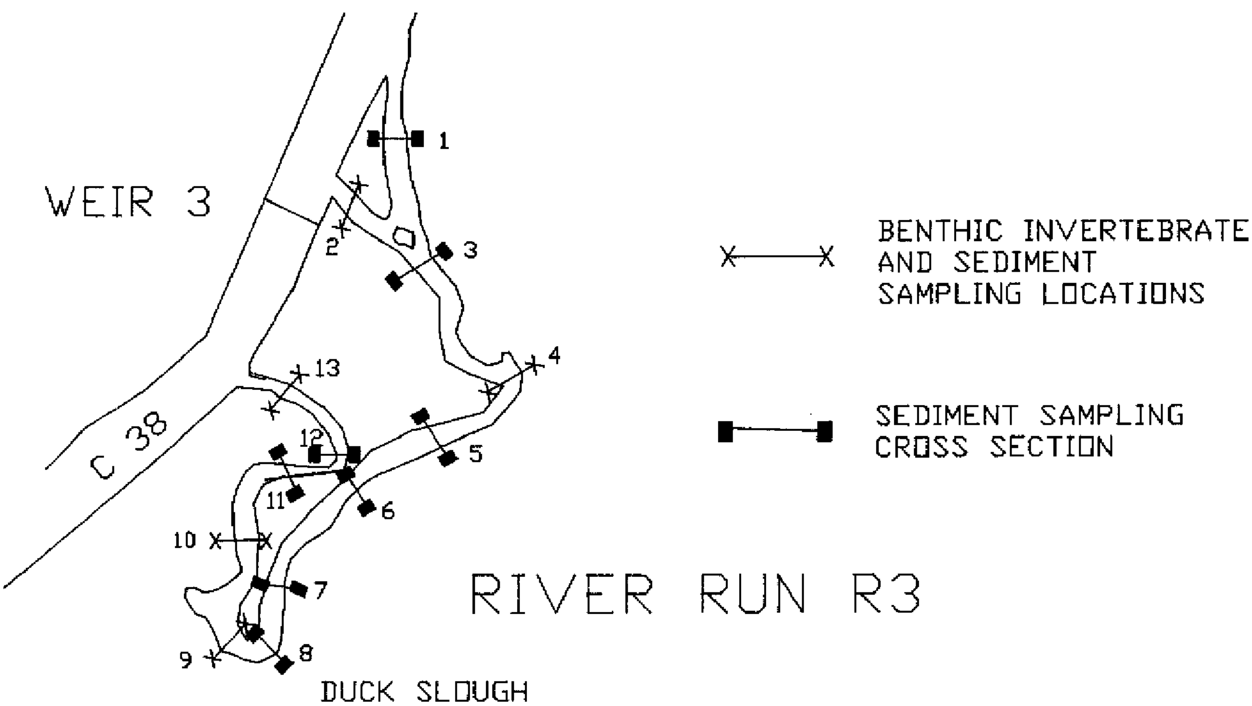

FIGURE 7. Map of river run adjacent Weir #3, showing sediment and benthic invertebrate sampling sites and location of Duck Slough transect.

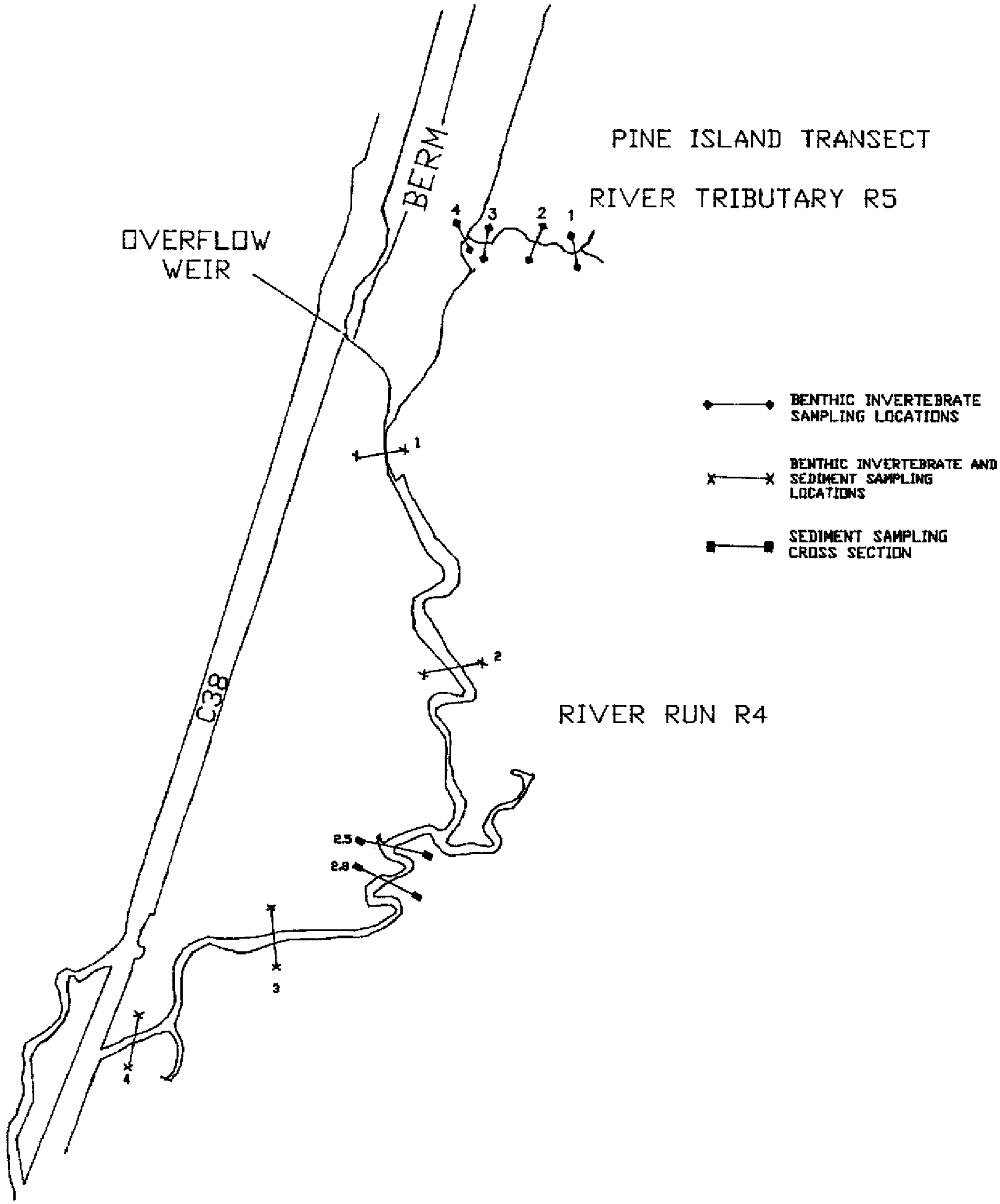

FIGURE 8. Map of Pine Island Slough tributary and adjoining river run showing sediment and benthic invertebrate sampling locations.

RESULTS

HYDROLOGY

Pre-channelization

Pre-channelization stage characteristics indicate that the floodplain was typically subjected to prolonged and extensive inundation. A comparison of historic (1942-67) stage duration data from the Fort Kissimmee gaging station with adjacent floodplain elevations indicates that most (94 percent) of the floodplain was inundated during 52 percent of this period of record (Table 1). These data also indicate that most of the floodplain was generally exposed to water depths of 0.3 - 0.6 m, although 42 percent of the floodplain had depths $\geq$ 0.9 m during 31 percent of the period of record, including nine spans of 4-10 months. In fact, historical aerial photographs show many small backwater lakes adjoining the river channel throughout the valley.

TABLE 1. Inundation frequencies (1942-67) for 171 ha of of floodplain adjacent Fort Kissimmee. Frequencies show proportions of the period of record that stages equalled or exceeded given elevations. Floodplain percentages represent measured areas of floodplain that were less than or equal to given elevations.

Elevation	% of Floodplain	Inundation Frequency
12.2 m	0.99	93.65 %
12.5	15.85	88.73
12.8	41.89	78.98
13.1	78.73	68.04
13.4	94.03	52.52
13.7	100.00	30.46

Although mean monthly stages suggest a typical subtropical wet-dry cycle (Figure 9), only peripheral areas of the floodplain underwent consistent seasonal drying on an annual basis. Seventy-nine percent of the floodplain, for example, would have remained inundated during average annual low stages in May and June. This portion (i.e., 79 percent) of the floodplain did not undergo annual drying during 11 of the 25-year period of record, and was constantly inundated for a period of two consecutive years during three intervals, and for a span of 1006 days from 1946-49. As indicated by inundation frequencies (Table 1), extensive droughts were not common; however, 84 percent of the floodplain was dry for at least five months during three separate years.

Prolonged floodplain inundation was facilitated by slow stage recession rates. Between 1941-61[3], average recession rates at Fort Kissimmee ranged from 0.12 - 0.27 m/month during months in which water levels declined. Although stages occasionally declined more rapidly during shorter time intervals (e.g., the 3-or 4-day period following a major storm event), recession rates exceeding 0.03 m/day were

3 Because stage records suggest that the flood control project began to affect recession rates at this location in 1962, only stage data prior to this date were used in this analysis.

rare. At Fort Kissimmee, the most rapid short-term recession rate prior to channelization occurred from 17-31 October 1956, when water levels declined from 14.4 to 13.5 m.

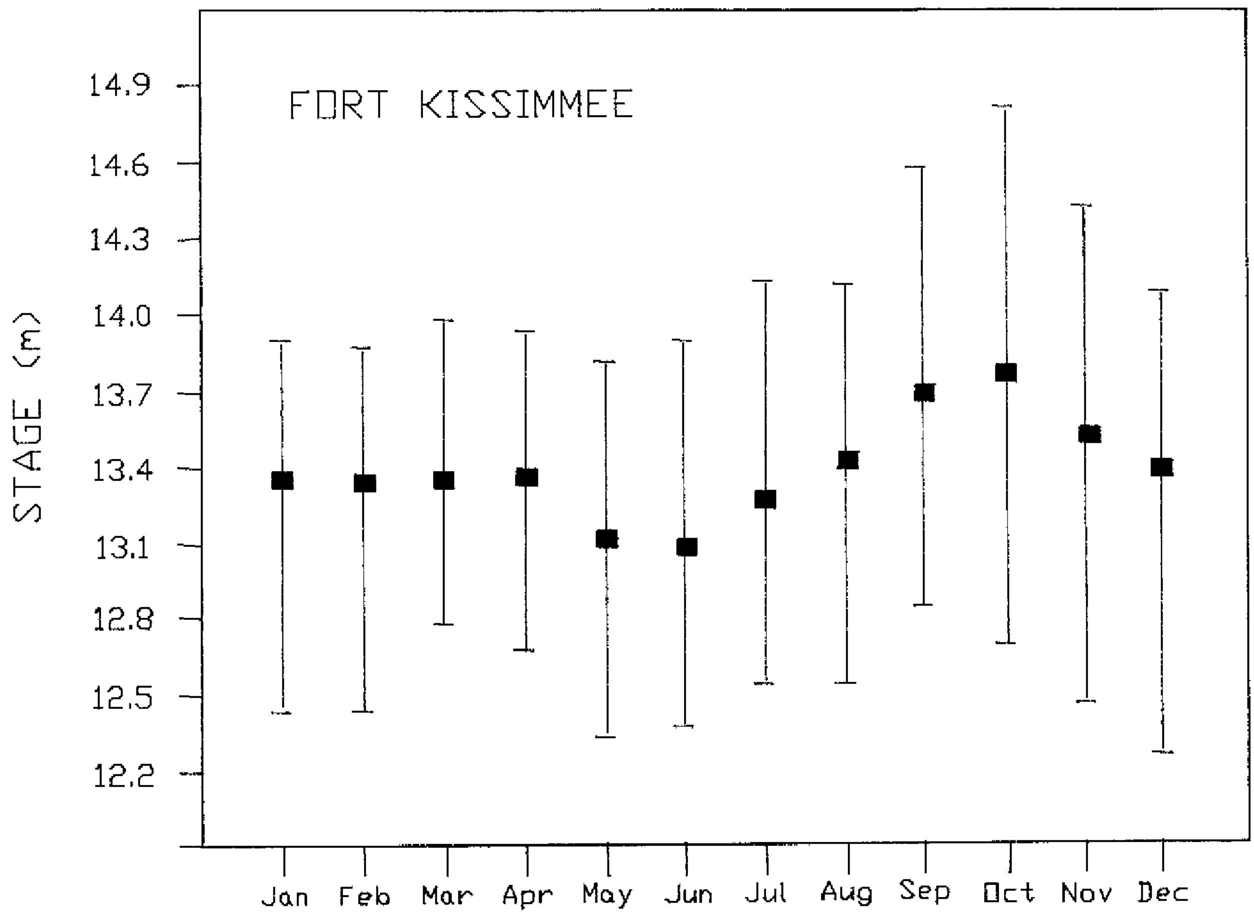

FIGURE 9. Mean monthly stages (N.G.V.D.) at Fort Kissimmee during pre-channelization period of record (1942-67). Ranges show stage variation that occurred during 80% of the period.

Pre-channelization discharge data from the outlet of Lake Kissimmee and present location of S-65E (Figure 10) indicate that the river had continuous flow throughout the historical period of record (1934-60). Discharge through channels presently found in Pool B exceeded 11 m^3/s during at least[4] 90 percent of this time interval. Highest discharges typically occurred at the end of the wet season (September - November), but flow was highly variable among years (Figure 11). Sixty-four percent of October discharges, for example, ranged between 18 m^3/s and a mean of 65 m^3/s. Due to a discharge/stage relationship that resulted in overbank flow when discharge ranged between 40-57 m^3/s, average velocities in the unmodified Kissimmee River did not exceed 0.6 m/s (Figure 12).

[4] Lake Kissimmee discharges were supplemented by inflows from tributary watersheds south of the lake.

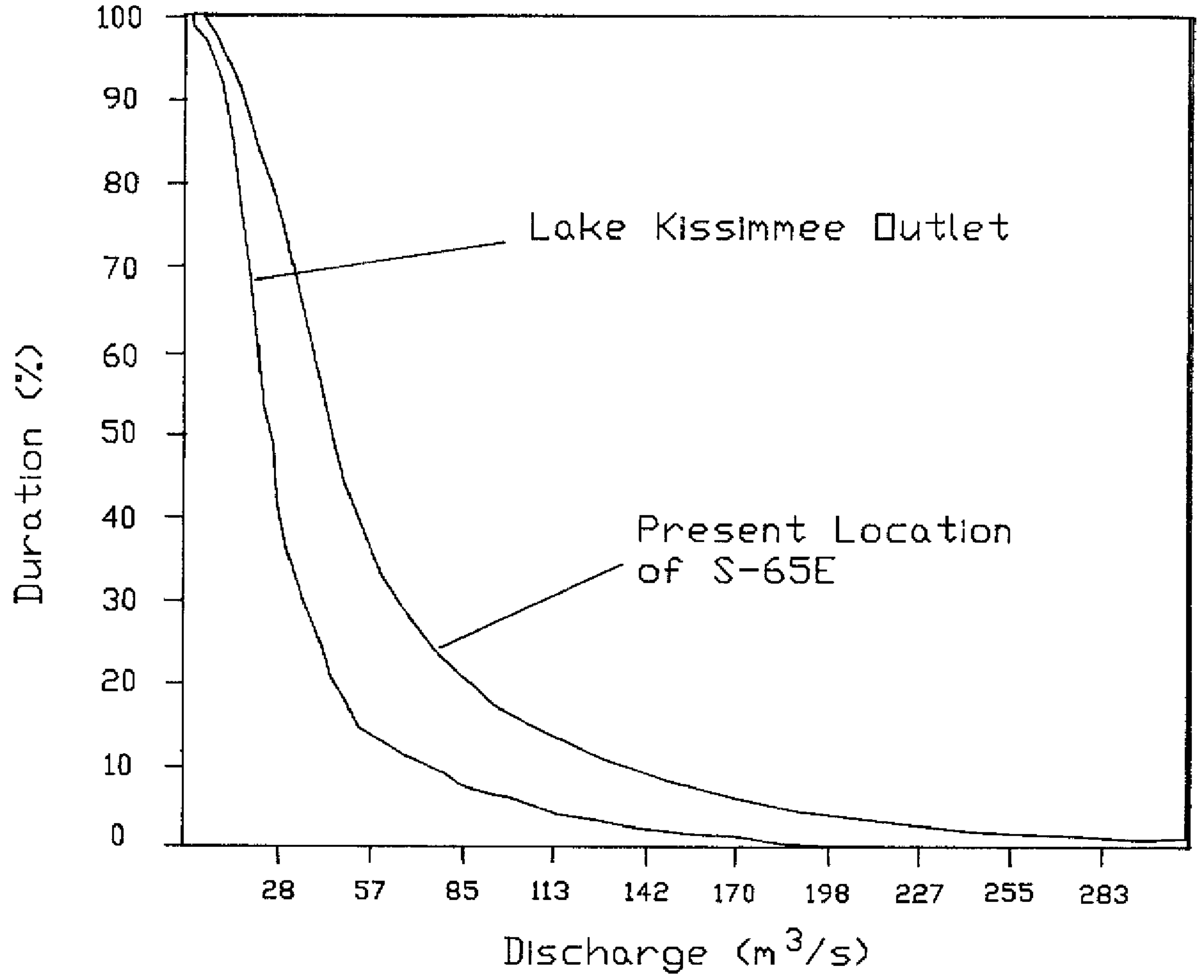

FIGURE 10. Pre-channelization discharge duration curves for the outlet of Lake Kissimmee (1934-60) and present location of S-65E (1930-60). Duration percentages represent the proportion of the period of record that given discharges were equalled or exceeded.

1969-1983

Due to the 12.2 m regulation stage, Pool B stages were less variable seasonally and annually during 1969-83 than during the pre-channelization period of record (Figure 13). Following channelization, the range of variability during each month was typically $\leq$ 0.6 m, which is less than half of the range of monthly stage variability that occurred during the pre-channelization period. Climatic conditions (i.e., lack of rainfall and evapotranspiration losses) and/or experimental drawdowns frequently lowered post-channelization, dry season stages to 11.6 - 11.9 m, but unlike the pre-channelization period, during which stages typically exhibited a wet-season peak, monthly stages were relatively constant from August - April during 1969-83.

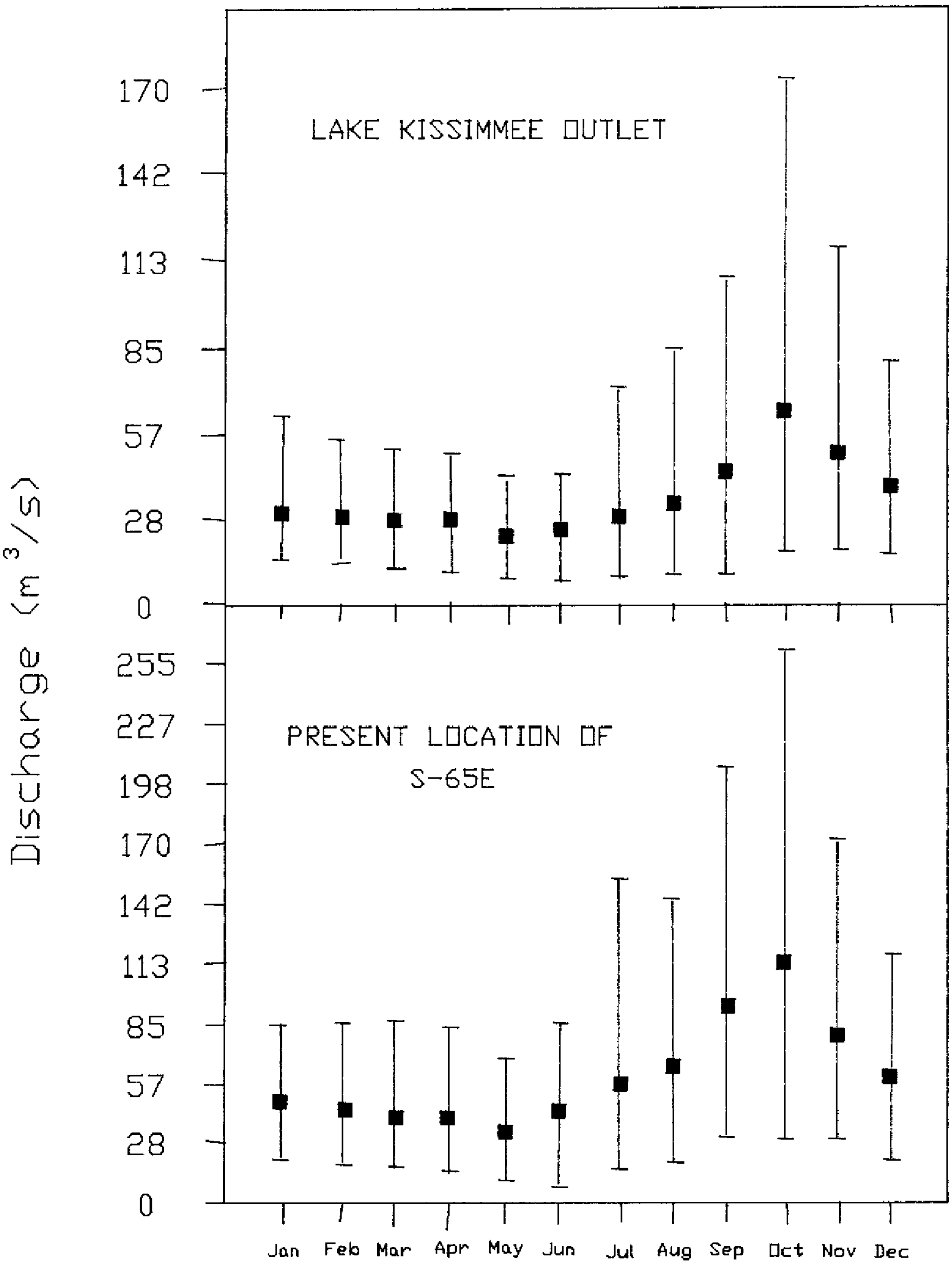

FIGURE 11. Mean monthly discharges at the outlet of Lake Kissimmee (1934-60) and present location of S-65E (1930-60). Ranges show discharge variation that occurred during 80% of the period of record.

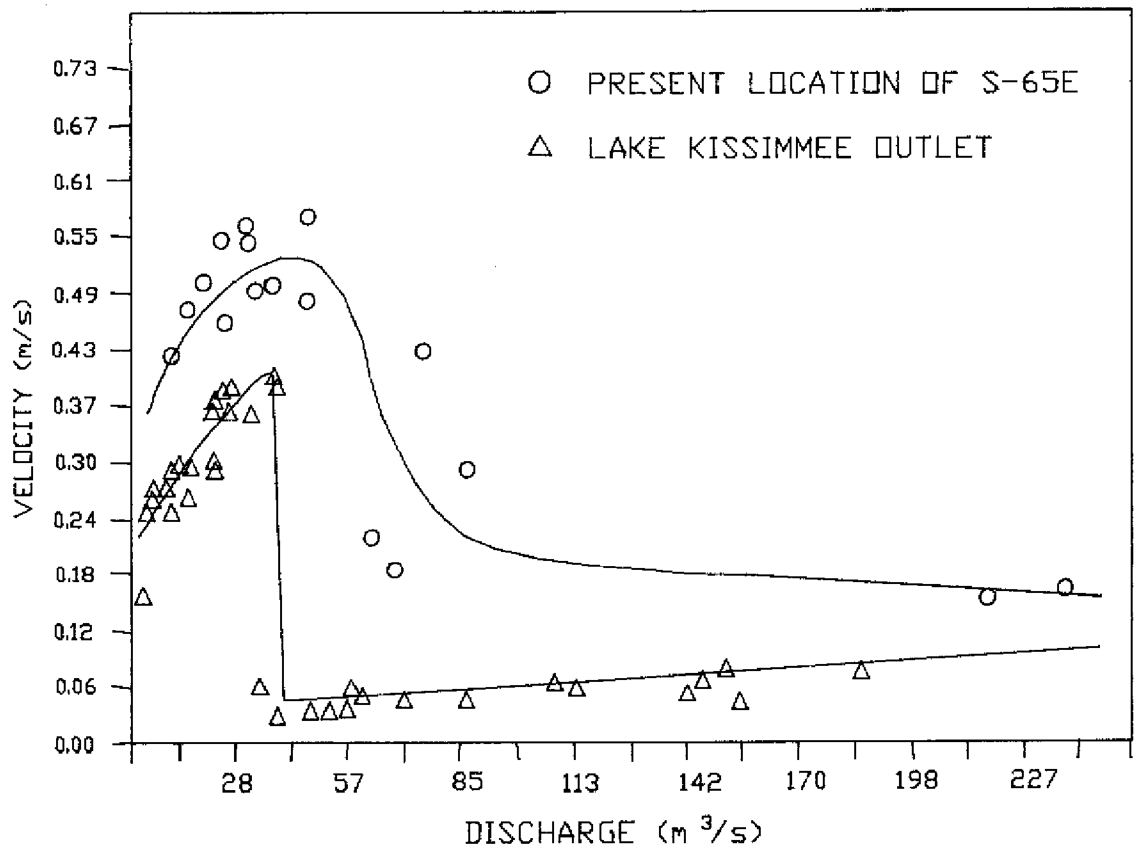

FIGURE 12. Pre-channelization Kissimmee River discharge/velocity relationships from stations at the outlet of Lake Kissimmee and present location of S-65E (based on U.S.G.S. data from 1950-51, compiled by Enge, 1975 and reported in Huber et al., 1976).

In addition to altering hydroperiods within the impounded portion of the pool, channelization and the 12.2 m regulation stage caused complete drainage of at least 2023 ha of wetlands (Milleson et al., 1980) which formerly occurred at higher elevations (i.e., > 12.5 m) within the northern half of the Pool B floodplain.

Since channelization, flow has been eliminated through all remaining river channels except runs draining tributary watersheds. Of the five runs monitored during the Demonstration Project, only the tributary draining Pine Island Slough (R5) received substantial flow during the pre-demonstration project period. The river run that occurs downstream of this tributary (R4) probably also received limited drainage from the lower end of the slough; however, any inflows to this run from R5 would have short-circuited to C-38 through the upper portion of R4 (Figure 8). Quantitative data on flow through R4 or R5 is not available, but observations during the Demonstration Project suggest that baseflow through R5 can be continuous during the wet season and intermittent throughout the remainder of the year.

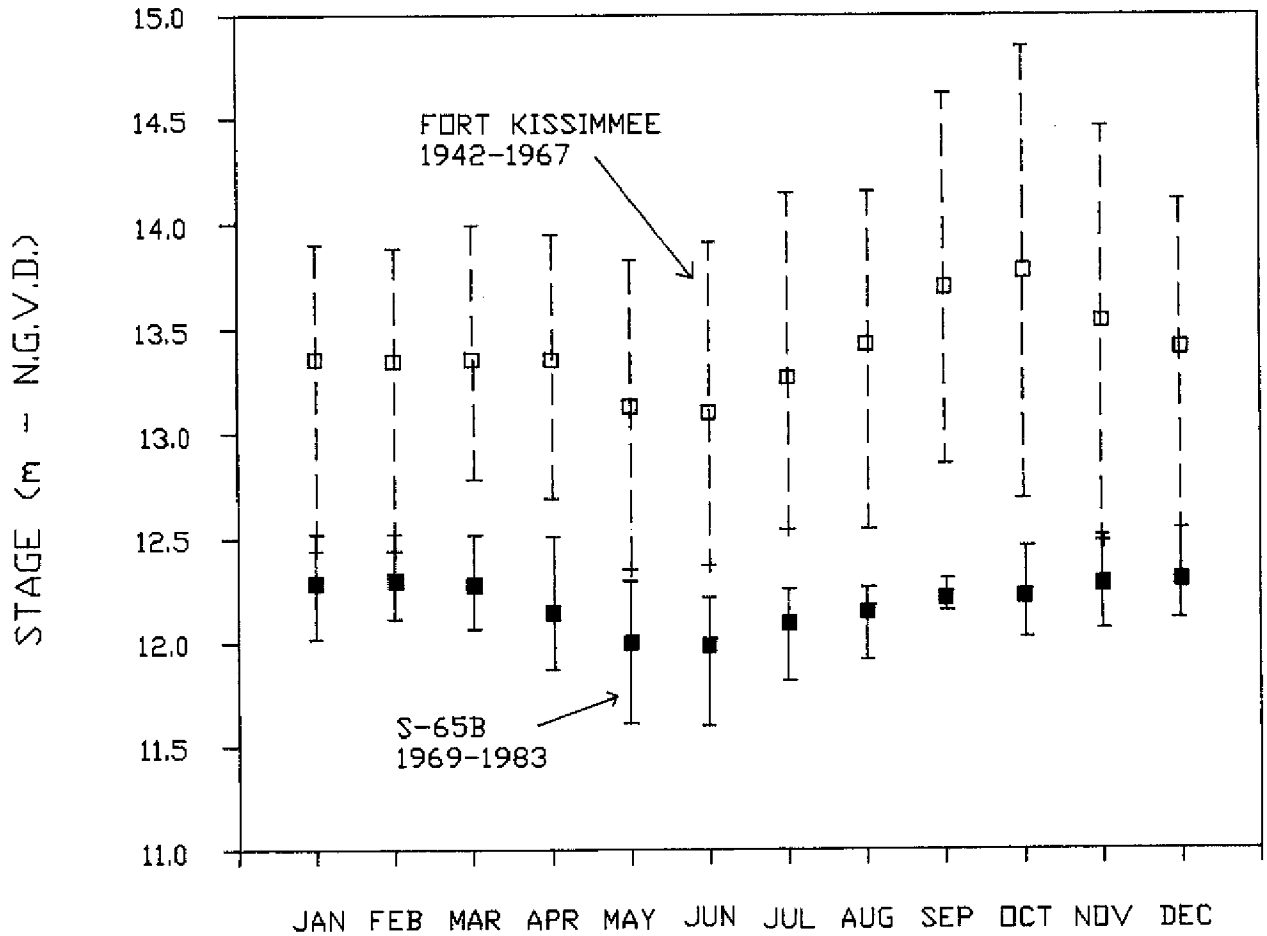

FIGURE 13. Mean monthly stages (N.G.V.D.) at S-65B from 1969-83. For comparison, historical (1942-67) mean monthly stages at Fort Kissimmee (Figure 9) also are shown. Ranges show monthly stage variation that occurred during 80% of this period.

1984-1988

Stage

A fairly constant 12.2 m stage was maintained in Pool B during the first nine months of 1984 (Figure 14a); however, from 1 October - 6 November, the stage was lowered to 10.4 m to accomodate construction of weir #3. This stage was held for 10 days, after which the pool stage gradually was returned to 12.2 m, where it remained from 21 November through December.

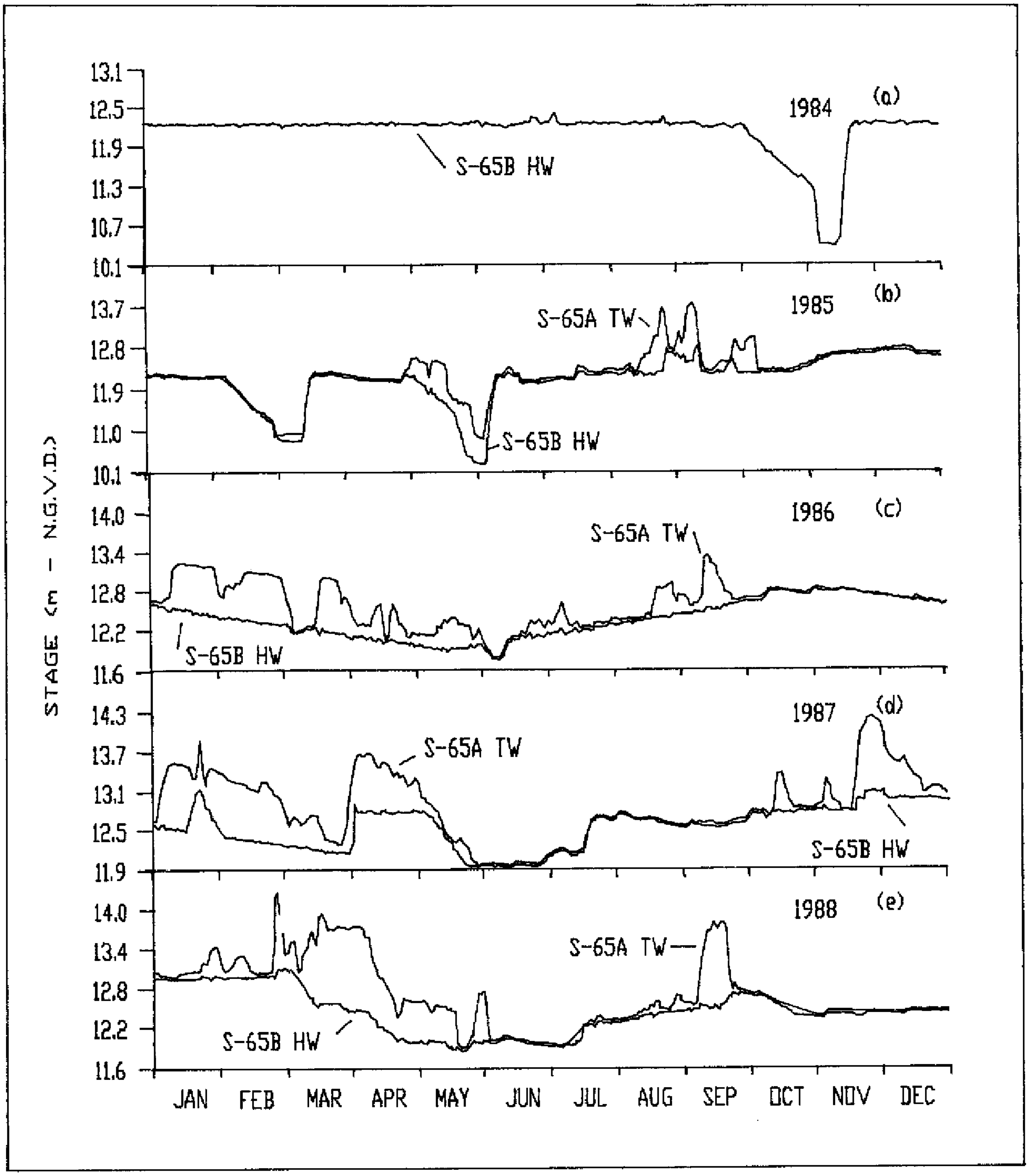

FIGURE 14. Annual stage hydrographs for Pool B during 1984-88. Data represent mean daily stages above S-65B (HW) and below S-65A (TW).

Pool B stages during 1985 (Figure 14b) were influenced by several factors, including weir construction, discharge events, and initiation of pool stage fluctuation. From 5 February - 16 March and 2 May - 9 June the pool stage was lowered again to 10.4 m to facilitate construction of weirs #1 and 2. Additionally, as during previous post-channelization years, the pool briefly (16 days) fell to 12.0 - 12.1 m during dry periods in April, and again in late June - early July. Otherwise, the stage at the lower end of Pool B was held close to 12.2 m from January - late August. However, from 27 April - 15 May, stages at the upper end of the pool were as much as 0.4 m higher, due to backwater effects of the weirs when water was discharged through the pool. Similarly, during high discharges in late August - early September, the stage below S-65A reached as high as 13.8 m, while water levels at the lower end of the pool increased to 12.4 - 12.8 m. After discharges decreased, a 12.2 m stage was restored at both the upper and lower ends of the pool for a few days in mid-September. Although this stage was maintained through late October at the lower end of the pool, increased discharges raised upper Pool B stages 0.3 - 0.8 m higher from 20 September - 7 October. On 28 October 1985, pool stage fluctuation was initiated. During the next two months, the entire pool was raised slowly to a mid-December peak of 12.8 m, and then lowered to 12.6 m by the end of the year.

Except for nine days at the beginning of June, when the pool stage fell below 11.9 m, stages at the lower end of Pool B (i.e., from S-65B to weir #1) followed the pool stage fluctuation schedule closely during 1986 (Figure 14c). However, upstream of each weir, stages rose whenever discharges through the pool increased, and fell rapidly when flows declined or ceased. As a result, stage hydrographs at any location upstream of weir #1 exhibited numerous "spikes" (see S-65A tailwater hydrograph in Figure 14c), which corresponded with the erratic discharge events that occurred throughout this year. During the highest discharge periods, stages immediately downstream of S-65A were 0.6 - 0.9 m higher than water levels at the lower end of the pool. Stages at the upper end of the pool were typically between 12.8 and 13.3 m during most of January - March, and periodically exceeded 12.8 m as the pool stage was raised during August - November.

During 1987, Pool B underwent a second complete year of pool stage fluctuation; however, deviations from the schedule occurred during a January high discharge test, and from April - May and July - mid-September (Figure 14d). During the discharge test, the upstream stage at S-65B was raised from 12.5 m on 16 January to a peak of 13.2 m on 22 January. The stage at the lower end of the pool was lowered gradually during the next 10 days, and followed the downward leg of the fluctuation schedule from 2 February - 31 March. Stages below S-65A were 0.6 - 0.9 m higher than the lower end of the pool during most of January and February, but declined to 12.3 m on 26 March. In early April, the pool stage was raised above schedule to accommodate high discharges and prevent structural damage to weir #3, which sustained a washout along its eastern end. During April, the upstream stage at S-65B was held at 12.8 m, while the stage at the upper end of the pool ranged from 13.2 - 13.7 m. The entire pool was returned to a low stage of 12.0 m by the end of May, and remained on schedule until 18 July when the pool stage was raised above 12.5 m to facilitate repair of the washout adjacent weir #3. The pool remained above schedule until mid-September. The scheduled high pool stage of 12.8 m was reached at the lower end of the pool at the end of October; however, from 20 November - 31 December, stages at S-65B were increased to 13.0 - 13.1 m to accommodate high discharges. Stages below S-65A exceeded 14.0 m at the height of these highest discharges (22 November - 1 December), but declined to 13.0 m by the end of the year.

Stages were maintained above 13.0 m at the lower end of the pool through 4 March 1988, and then slowly lowered until the downward leg of the fluctuation schedule was intercepted at the end of April (Figure 14e). Due to high flows through the pool, including a second high discharge test in late February, stages below S-65A were 0.3 - 1.0 m higher than stages at the lower end of the pool from 25 January through most of May. However, as during previous years, stage hydrographs for this period show several erratic and unnatural fluctuations, including periods during which stages at the upper end of the pool decreased by 0.6 m within 48 hours. Except for about 15 days in mid-September, when high discharges caused stages at the upper end of the pool to rise above 13.7 m, the entire pool followed the fluctuation schedule from June - September. Although a wet season high of 12.7 m was reached at the lower end of the pool on 26 September, climatic conditions did not allow this high to be maintained and the pool fell 0.2 - 0.5 m below schedule during October -December.

Compared to 1969-83 (Figure 13), mean monthly stages at the lower end of Pool B (Figure 15a) increased by approximately 0.3 m during the months of August - February of 1985-88. Stage variability (e.g., range of stages that occurred during 80 percent of the period) at S-65B also increased slightly (by about 0.2 m) during most months; however, June stages were much more stable during the Demonstration Project monitoring period, typically ranging only from 11.9 - 12.0 m. During several months, stage variability was greater at Fort Kissimmee (Figure 15b) than at the lower end of the pool, and was similar to stage variability in pre-channelization records (Figure 9) during March and April. Mean monthly stages at Fort Kissimmee also increased during the Demonstration Project, but remained slightly (0.1 - 0.2 m) below average pre-channelization stages at this location during January and February, and 0.4 - 1.1 m lower in March - December.

Stage recession rates during the Demonstration Project frequently were much more rapid than rates at which water levels typically declined prior to channelization. For example, during 24 periods from June 1985 - December 1988, water levels on the floodplain declined at a rate $\geq$ 0.08 m/day. During most of these periods stages fell 0.2 - 1.3 m in 2 - 7 days, and during nine of these intervals, recession rates exceeded 0.2 m/day.

Discharge

Construction of weir #3 was completed in December 1984, but no substantial flow was diverted through R3 during this year.

During 1985, flow was generated through Pool B river runs for the first time since channelization. Significant flows were initially diverted from 29 April - 2 June, when discharges exceeded 11 m^3/s for 18 days in R3 and 10 days in R2 (Table 2). During high discharge periods in August - October, flows exceeded 11 m^3/s for 40 days in R1 and 28 days in R2 and R3. On 8-9 September, discharges through these runs reached 71-79 m^3/s.

Significant discharge was diverted through runs adjacent weirs during seven months in 1986 (Table 3). Highest discharges occurred primarily during January - May, when diverted flows exceeded 11 m^3/s about 60 percent of the time in R2 and R3, and for 73 percent of this period in R1. Only minimal (i.e., $<$ 11 m^3/s) or no flow occurred through R1-3 during most of June, July, and October - December, but discharges through these runs exceeded 11 m^3/s for 24-30 days during August and September. By early summer 1986, the berm for the Flow-Through Marsh was completed and the upstream connection of R4 with C-38 was blocked. Thereafter, all water draining from the Pine Island Slough watershed passed through the entire length of R4, and reestablished periodic or seasonal flow through this section of river.

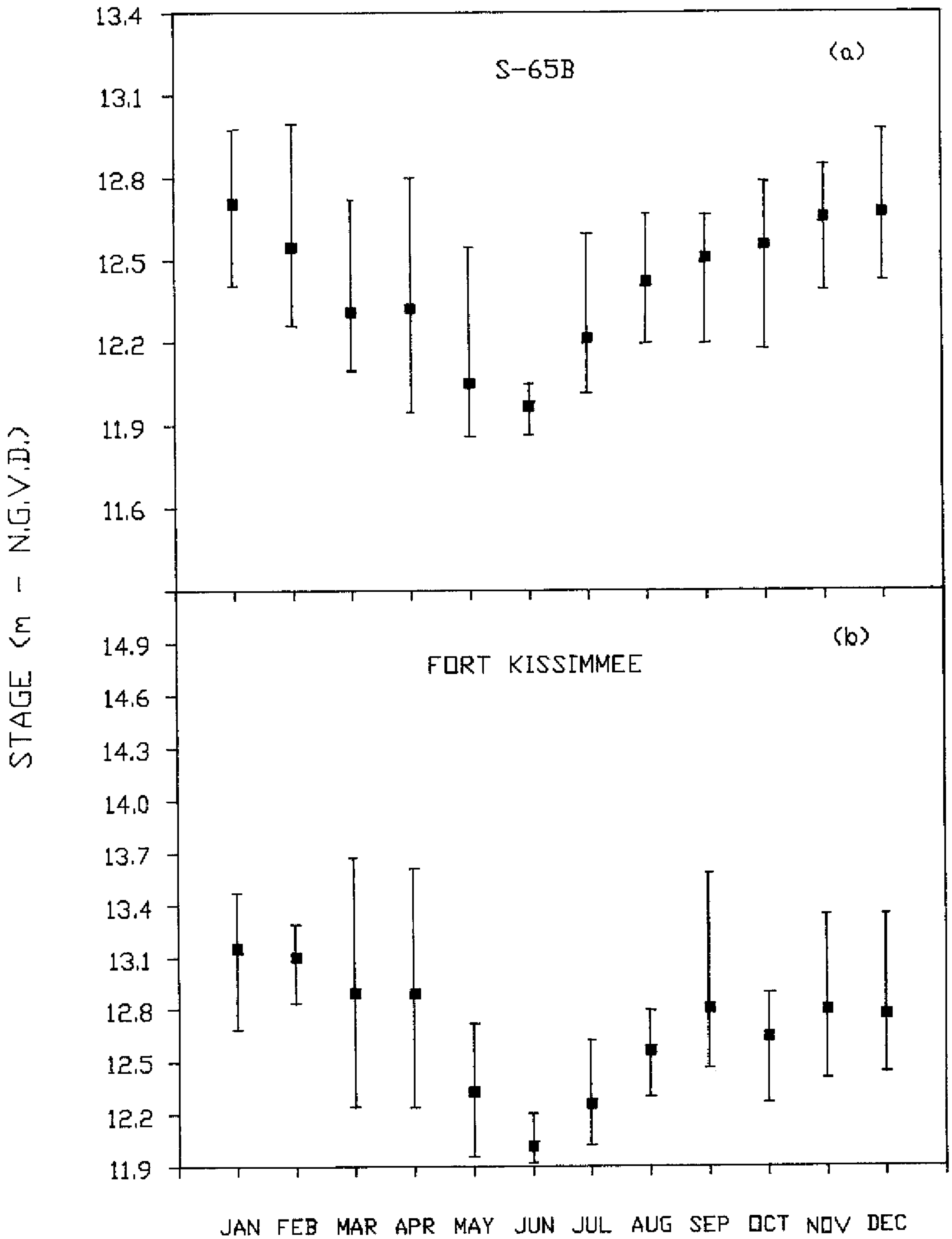

FIGURE 15. Mean monthly stages at S-65B and Fort Kissimmee during July 1985-December 1988. Ranges show monthly stage variation that occurred during 80% of this period.

TABLE 2. Average daily discharges (m^3/s) through R1-3 during 1985. Data show number of days during each month that diverted discharges ranged within given categories.

Month	Discharge				
	0-11	11-26	26-41	41-68	>68
			R1		
January					
February			NO DATA		
March					
April	30	-	-	-	-
May	31	-	-	-	-
June	30	-	-	-	-
July	28	-	-	-	-
August	16	12	3	-	-
September	13	6	7	2	2
October	23	4	4	-	-
November	30	-	-	-	-
December	31	-	-	-	-
			R2		
January					
February			NO DATA		
March					
April	30	-	-	-	-
May	23	8	-	-	-
June	28	2	-	-	-
July	31	-	-	-	-
August	24	5	-	2	-
September	17	4	5	2	2
October	23	8	-	-	-
November	30	-	-	-	-
December	31	-	-	-	-
			R3		
January					
February			NO DATA		
March					
April	28	2	-	-	-
May	17	13	1	-	-
June	28	2	-	-	-
July	31	-	-	-	-
August	24	5	1	1	-
September	17	6	3	2	2
October	23	8	-	-	-
November	30	-	-	-	-
December	31	-	-	-	-

TABLE 3. Average daily discharges (m^3/s) through R1-3 during 1986. Data show number of days during each month that diverted discharges ranged within given categories.

Month	Discharge				
	0-11	11-26	26-41	41-68	>68
			R1		
January	8	3	20	-	-
February	11	12	18	-	-
March	12	5	14	-	-
April	9	19	2	-	-
May	14	17	-	-	-
June	25	5	-	-	-
July	25	6	-	-	-
August	18	12	1	-	-
September	13	9	7	1	-
October	31	-	-	-	-
November	25	5	-	-	-
December	30	1	-	-	-
			R2		
January	8	4	19	-	-
February	1	11	16	-	-
March	14	8	9	-	-
April	17	13	-	-	-
May	16	15	-	-	-
June	29	1	-	-	-
July	29	2	-	-	-
August	19	12	-	-	-
September	16	8	6	-	-
October	31	-	-	-	-
November	30	-	-	-	-
December	31	-	-	-	-
			R3		
January	9	8	14	-	-
February	1	25	2	-	-
March	13	10	8	-	-
April	19	11	-	-	-
May	19	12	-	-	-
June	30	-	-	-	-
July	29	2	-	-	-
August	21	10	-	-	-
September	16	11	3	-	-
October	31	-	-	-	-
November	26	4	-	-	-
December	31	-	-	-	-

As in 1986, substantial flows were diverted through river runs adjacent weirs during the first five months of 1987 (Table 4). Discharges through R1-3 exceeded 11 m^3/s during 83-90 percent of the days during this period, and were greater than 26 m^3/s for most of January, February and April. From June - September, minimal or no flows occurred in these runs, but sustained discharges > 11 m^3/s were diverted through R2 and R3 for at least 50 days, and in R1 for 80 days, during October - December. This was also the first complete year during which R4 carried inflows from Pine Island Slough.

In 1988, January - May discharges through R1-3 were virtually identical to the previous year (Table 5). Flows through these runs exceeded 11 m^3/s for 83-91 percent of this period, and were > 26 m^3/s during most of January, February and April. However, the only other period during this year that significant flows were diverted was September, when discharge through R1-3 exceeded 11 m^3/s for 14 - 18 days.

FLOODPLAIN VEGETATION

Marsh Transect

Ground elevations along this transect ranged from 11.8 - 12.6 m, but 89 percent of transect points had elevations between 11.9 and 12.2 m. This elevation range (11.9 - 12.2 m) represents approximately 27 percent of the floodplain at this location, and 71 percent of the floodplain in this portion of the pool lies below 12.2 m.

Compared to the 1969-84 pre-demonstration project period (Figure 16a), most of the Marsh Transect was inundated 20-35 percent fewer days in 1984-85 (Figure 16b) when three extreme stage drawdowns were conducted to accomodate weir construction. During the three subsequent years (Figure 16c-e), annual inundation frequencies of transect elevations ranged from about 60-90 percent, which was not substantially different from the pre-demonstration project period. However, higher than scheduled stages during 1987 and 1988 resulted in extended periods of greater water depths. The average transect elevation (12.1 m), for example, had water depths of 50 cm for 127 days (35 percent) in 1986-87 and 196 days (48 percent) in 1987-88 compared to a mean of only 33 days during annual periods from 1969-1984, 0 days in 1984-85 and 39 days in 1985-86.

Plant species composition of this marsh was representative of existing broadleaf marshes that occur on the floodplain at the lower end of C-38 pools (Milleson et al., 1980). Two of the most common species present in 1984, Alternanthera philoxeroides and Sagittaria lancifolia, remained dominant during the course of the Demonstration Project monitoring period, but seven species showed substantial changes in frequency along the transect (Table 6).

Hydrocotyle sp., Leersia hexandra, Pontederia cordata and Bacopa caroliniana showed decreased frequencies, while occurrences of Polygonum punctatum, Panicum hemitomon and Ludwigia peruviana increased. Panicum expanded its distribution primarily during 1984-85, when inundation frequencies declined (i.e., relative to previous years). The frequency of occurrence of Polygonum also increased during this relatively "dry" year, but distributions of Polygonum, Bacopa, Leersia, and Ludwigia appeared to be affected mostly by increased water depths during the last three years of monitoring. Frequencies of Leersia declined, and Polygonum and Ludwigia increased, primarily during 1986-88, when the frequency of higher water depths increased on most elevations along the transect. Bacopa was eliminated from the

TABLE 4. Average daily discharges (m^3/s) through R1-3 during 1987. Data show number of days during each month that diverted discharges ranged within given categories.

Month	Discharge				
	0-11	11-26	26-41	41-68	>68
			R1		
January	3	3	8	17	-
February	-	-	17	11	-
March	1	24	6	-	-
April	-	1	2	22	5
May	10	18	2	-	-
June	30	-	-	-	-
July	31	-	-	-	-
August	31	-	-	-	-
September	18	12	-	-	-
October	8	16	4	3	-
November	2	12	6	1	9
December	2	14	9	4	2
			R2		
January	4	2	7	18	-
February	-	3	22	3	-
March	8	22	1	-	-
April	-	1	5	14	10
May	14	15	2	-	-
June	30	-	-	-	-
July	31	-	-	-	-
August	31	-	-	-	-
September	30	-	-	-	-
October	22	5	4	-	-
November	11	6	3	1	9
December	2	15	8	4	2
			R2		
January	4	5	8	14	-
February	-	11	17	-	-
March	8	22	1	-	-
April	-	1	2	20	7
May	11	13	6	1	-
June	30	-	-	-	-
July	31	-	-	-	-
August	31	-	-	-	-
September	30	-	-	-	-
October	23	7	1	-	-
November	13	6	1	1	9
December	6	15	8	1	1

TABLE 5. Average daily discharges (m^3/s) through R1-3 during 1988. Data show number of days during each month that diverted discharges ranged within given categories.

Month	Discharge				
	0-11	11-26	26-41	41-68	>68
			R-1		
January	8	3	19	1	-
February	-	10	3	7	9
March	-	2	5	19	5
April	-	6	14	5	5
May	7	16	8	-	-
June	28	1	1	-	-
July	31	-	-	-	-
August	27	4	-	-	-
September	12	5	1	4	8
October	31	-	-	-	-
November	30	-	-	-	-
December	31	-	-	-	-
			R-2		
January	9	6	16	-	-
February	3	8	5	5	8
March	-	2	5	19	5
April	-	15	6	4	5
May	9	19	3	-	-
June	28	1	1	-	-
July	31	-	-	-	-
August	29	2	-	-	-
September	13	4	1	3	9
October	31	-	-	-	-
November	30	-	-	-	-
December	31	-	-	-	-
			R-3		
January	9	6	16	-	-
January	11	15	5	-	-
February	7	5	5	9	3
March	-	2	8	12	9
April	-	15	6	4	5
May	8	20	3	-	-
June	28	1	1	-	-
July	31	-	-	-	-
August	31	-	-	-	-
September	16	2	1	4	7
October	31	-	-	-	-
November	30	-	-	-	-
December	31	-	-	-	-

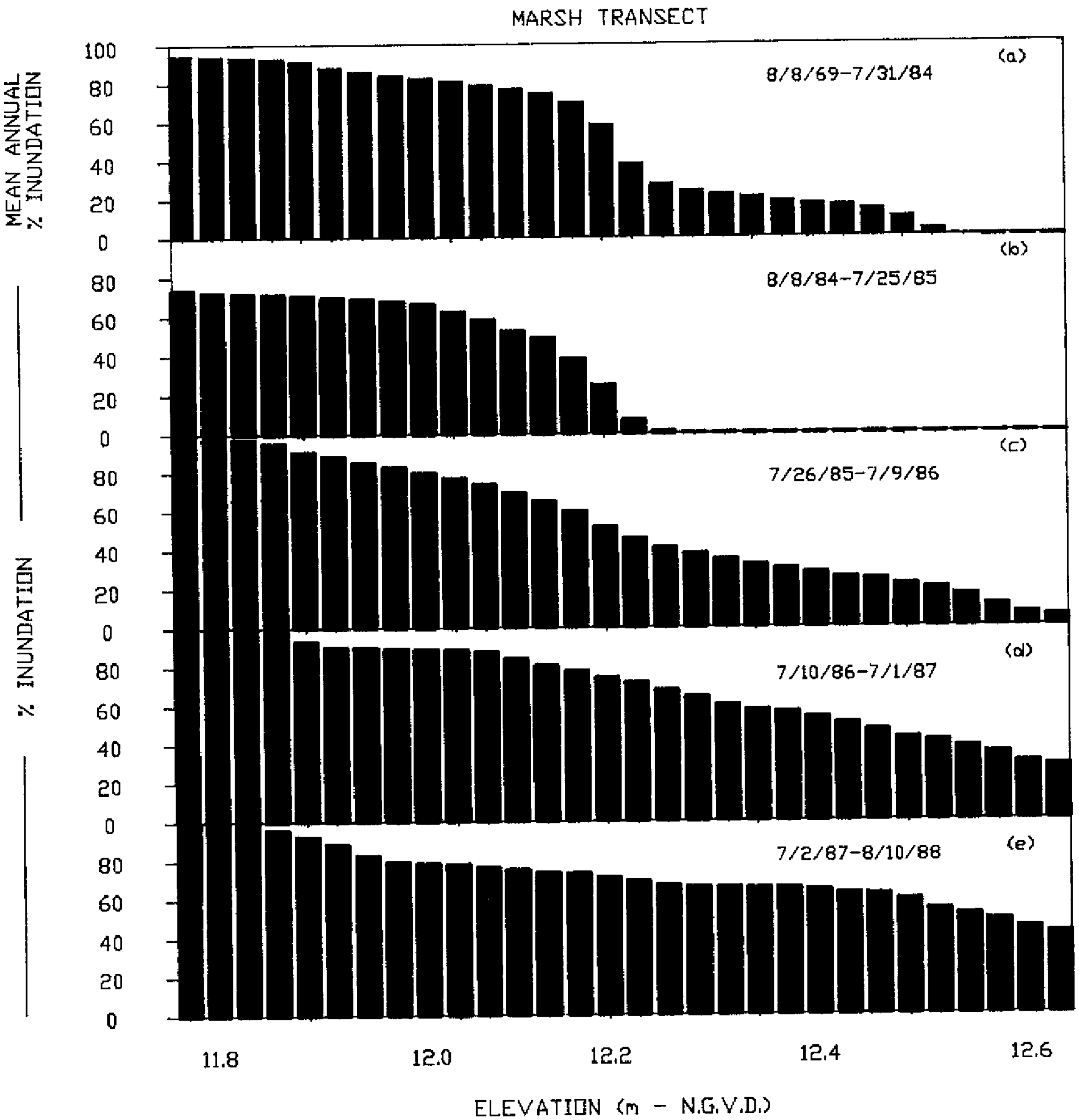

FIGURE 16. Annual hydroperiods for floodplain elevations along the Marsh Transect (1969-88). Hydroperiods are expressed as the proportion of each period that given elevations were inundated.

lowest elevations along the transect in 1985-86, and from progressively higher elevations during each of the next two years. During 1988, Bacopa was found only at elevations $\geq$ 12.1 m. Annual changes in frequency of occurrence of Hydrocotyle and Pontederia were not definitively correlated with project-related changes in hydrology, but Patton and Judd (1988) found that increased water depths eliminated Hydrocotyle umbellata in another Florida marsh.

TABLE 6. Dominant plant species along the Marsh Transect. Data show the number of quadrats in which each plant species occurred during each sampling year. A total of 53 quadrats were sampled along this 404 m transect.

Species	Sampling Year				
	1984	1985	1986	1987	1988
Alternanthera philoxeroides	39	38	45	47	41
Bacopa carolininana	22	19	15	10	4
Cephalanthus occidentalis	12	10	14	15	14
Hydrochloa caroliniensis	15	14	12	11	7
Hydrocotyle sp.	43	31	13	21	2
Leersia hexandra	45	43	49	39	14
Ludwigia peruviana	4	4	2	4	13
Nuphar luteum	12	6	6	9	11
Panicum hemitomon	24	34	29	30	38
Polygonum punctatum	5	14	16	18	40
Pontederia cordata	31	32	23	26	18
Sagittaria lancifolia	45	45	46	44	40

Drained Floodplain Transect

Ground elevations along most of this transect ranged between 12.2 - 12.7 m and represent approximately 20 percent of the floodplain at this location. Mean annual pre-demonstration project inundation frequencies for this elevation range were skewed (Figure 17a): elevations > 12.5 m were rarely inundated, elevations between 12.3 - 12.5 m were inundated 50-100 days, and elevations < 12.2 m had surface water 200-250 days. During 1984-85 most of the transect was dry throughout the year (Figure 17b). Inundation frequencies increased each subsequent year (Figure 17c-e) and exceeded annual means for 1969-84 by about 10 percent in 1985-86, 30 percent in 1986-87, and 50 percent in 1987-88. During the last year of the monitoring period transect elevations were inundated 150-275 days.

At the beginning of the monitoring period plant species composition along this transect consisted of a diverse mixture of xerophytic and mesophytic (i.e., facultative upland or facultative wetland) taxa, with scattered obligate wetland species. The most dominant species in 1984 were Centella asiatica, Axonopus spp. (mostly A. affinis or A. compressus), Sacciolepis indica, Cyperus spp., Hydrocotyle sp., Eupatorium capillifolium, Fuirena pumila, and Phyla nodiflora (Table 7). Frequencies of Phyla and Eupatorium, and three other abundant taxa, Ambrosia artemisiifolia, Bacopa caroliniana, and Myrica cerifera, showed no major net change

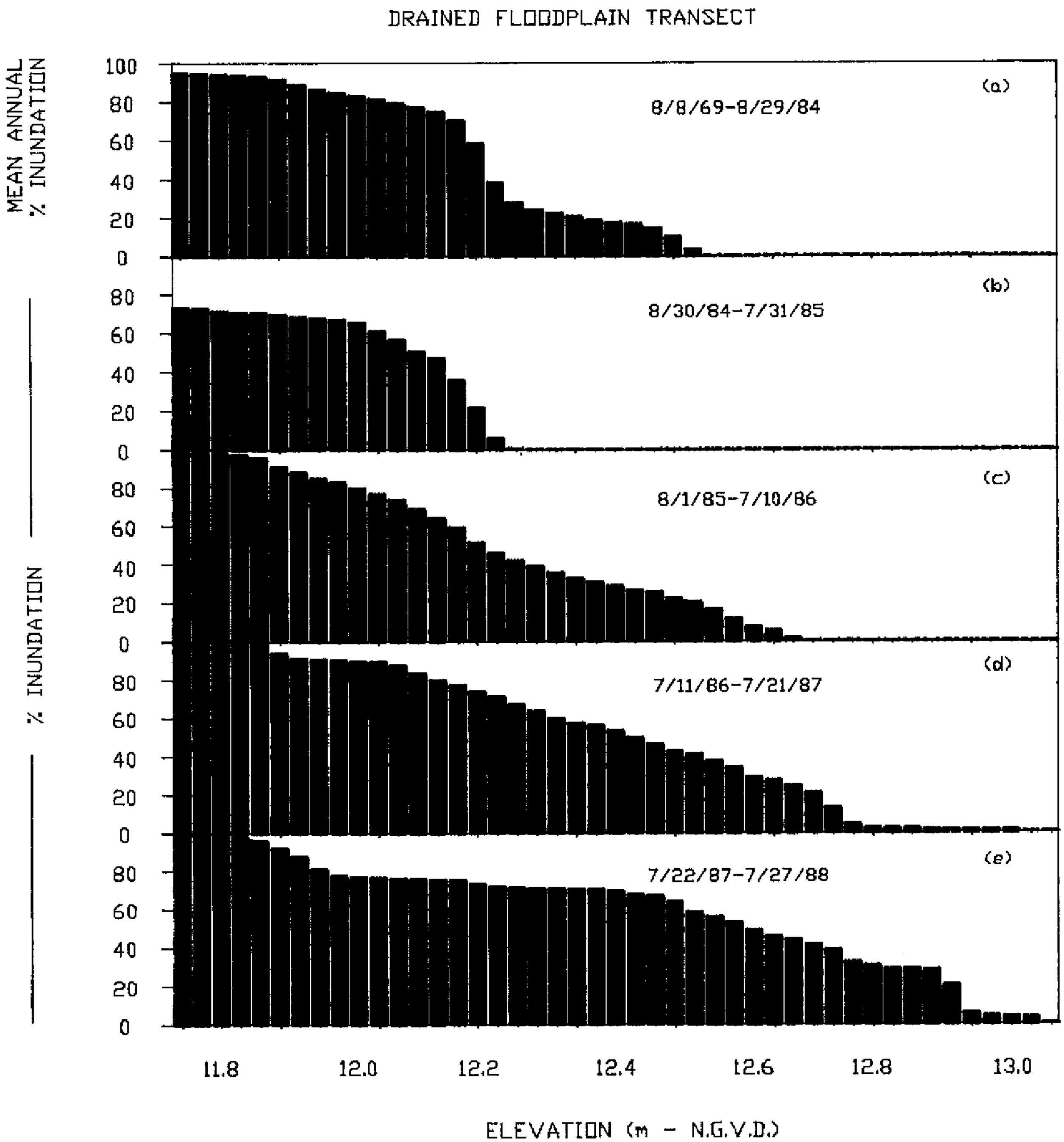

FIGURE 17. Annual hydroperiods for floodplain elevations along the Drained Floodplain Transect (1969-88). Hydroperiods are expressed as the proportion of each period that given elevations were inundated.

TABLE 7. Dominant plant species along the Drained Floodplain Transect. Data show the number of quadrats in which each plant species occurred during each sampling year. A total of 49 quadrats were sampled along this 366 m transect.

Species	Sampling Year				
	1984	1985	1986	1987	1988
Alternanthera philoxeroides	11	9	11	37	37
Ambrosia artemisiifolia	24	26	46	20	29
Axonopus spp.	43	45	38	21	23
Bacopa caroliniana	12	14	11	15	13
Bacopa monnieri	3	2	7	8	18
Boltonia diffusa	22	18	1	16	5
Centella asiatica	47	49	48	35	36
Commelina diffusa	1	1	1	7	15
Cuphea carthagenensis	12	9	22	25	29
Cyperus spp.	36	19	24	11	15
Desmodium paniculatum	10	15	16	3	1
Diodea virginiana	13	15	16	18	40
Eleocharis vivipara	22	14	29	20	33
Eupatorium capillifolium	30	35	15	9	24
Fuirena pumila	29	6	12	3	1
Hydrochloa caroliniensis	13	17	13	21	37
Hydrocotyle sp.	32	10	16	23	13
Ludwigia repens	7	2	8	18	22
Myrica cerifera	16	22	21	12	16
Panicum hemitomon	14	14	13	15	23
Panicum repens	19	21	28	31	31
Paspalum conjugatum	17	16	8	10	5
Phyla nodiflora	28	28	37	29	35
Polygonum punctatum	2	0	4	13	24
Rhexia mariana	10	15	1	2	1
Rhynchospora microcarpa	21	23	20	14	1
Sacciolepis indica	34	10	6	2	0
Sesbania punicea	18	36	37	23	38

at the end of the monitoring period, although the distribution of Eupatorium decreased markedly during 1985-86. Frequencies of four of the other dominant taxa during 1984, Cyperus spp., Fuirena, Hydrocotyle, and Sacciolepis indica, declined markedly during the dry period from 1984-85 and remained low or decreased during the remainder of the monitoring period. The distribution of only one species, Sesbania punicea, increased substantially during this dry year. As inundation frequencies increased during the next three years, frequency of occurrence of at least eight wetland indicator species increased. By 1988, these species were either dominant (e.g., Alternanthera philoxeroides, Hydrochloa caroliniensis, Eleocharis vivipara, and Panicum repens) or among the most common species (e.g., Polygonum punctatum, Panicum hemitomon, Ludwigia repens, and Bacopa monneri) along the transect. Higher inundation frequencies also appeared to lead to increased frequencies of three facultative wetland species, Diodea virginiana, Cuphea carthagenensis and Commelina diffusa. The only species that showed a net increase in frequency of occurrence that did not appear to be correlated with project-related changes in inundation frequencies was an annual herb, Sesbania punicea. Meanwhile, higher inundation frequencies seemed to result in decreased frequency of occurrence of mesic carpetgrass species (i.e., A. affinis and A. compressus) and five transitional species: Centella, Rhynchospora microcarpa, Paspalum conjugatum, Desmodium paniculatum, and Rhexia mariana.

Brush Transect A

Plot elevations ranged from 12.2 - 12.8 m, which represents the lower 43 percent of floodplain elevations at this location. Prior to the Demonstration Project, plots 1 and 2 were rarely inundated, while plots 3-5 were covered with surface water for an average of 52 - 100 days each year (Figure 18). All plots were dry during the entire year in 1984-85, and plots 1 and 2 remained dry for most of 1985-86. Compared to annual intervals from 1969-84, inundation frequencies of plots 3-5 were 30 percent higher in 1985-86 and about 50 percent higher from 1986-88. In 1986-87, plots 1 and 2 were inundated for substantial periods for the first time since channelization. In 1987-88, plots 3-5 were covered with surface water for 270 days, inundation frequencies of plot 2 increased to 193 days, and plot 1 was flooded for 167 days. The backwater effect of Weir #1 resulted in as many as 71 additional days of inundation during each year from 1985-88.

The floodplain plant community represented by these brush plots consisted of shrubs, vines, and herbaceous species. At the beginning of the monitoring period, three species, Panicum hemitomon, Sambucus canadensis, and Urena lobata were dominant along the entire transect (Table 8). Sambucus also dominated the canopy. However, pre-demonstration project inundation frequencies seemed to influence the distribution of most other common species. Alternanthera philoxeroides, Andropogon virginicus, Boehmeria cylindrica, Hydrocotyle sp., Juncus effusus, and Ludwigia peruviana, for example, were found either exclusively, or primarily, in plots exposed to some annual inundation (i.e., plots 3-5), while pepper vine, Ampelopsis arborea, occurred only in higher elevation plots (1 and 2) which had not been inundated since channelization. The distributions of two other common understory species, Ambrosia artemisiifolia and Paspalum conjugatum, and Baccharis halimifolia in the canopy, were not correlated with pre-demonstration project inundation frequencies.

Increased inundation frequencies during the Demonstration Project led to a major change in the brush community. The most significant change was the virtual elimination of the dominant shrub species, Sambucus canadensis (Table 8). Loss of this species began in plots 3-5 in 1985-86, when inundation frequencies of these elevations increased to 130-185 days. Sambucus was eliminated from plots 1 and 2 in 1987-88, when inundation of these elevations increased to similar frequencies. Decreased frequencies of occurrence of Urena lobata and Boehmeria cylindrica also appeared to be caused by increased inundation, as both species declined during years of prolonged flooding (1986-88). In contrast, patterns by which distributions of Paspalum conjugatum, Andropogon virginicus, Juncus effusus, and Hydrocotyle sp. declined were not suggestive of a strong correlation with inundation frequencies. However, loss of Hydrocotyle from plots 4 and 5 in 1985-86 may have been caused by increased water depths. For at least 60 days during this year, elevations represented by these plots were exposed to stages that were 30 cm higher than any annual intervals from 1969-84 (i.e., after channelization). Distributions of three species,

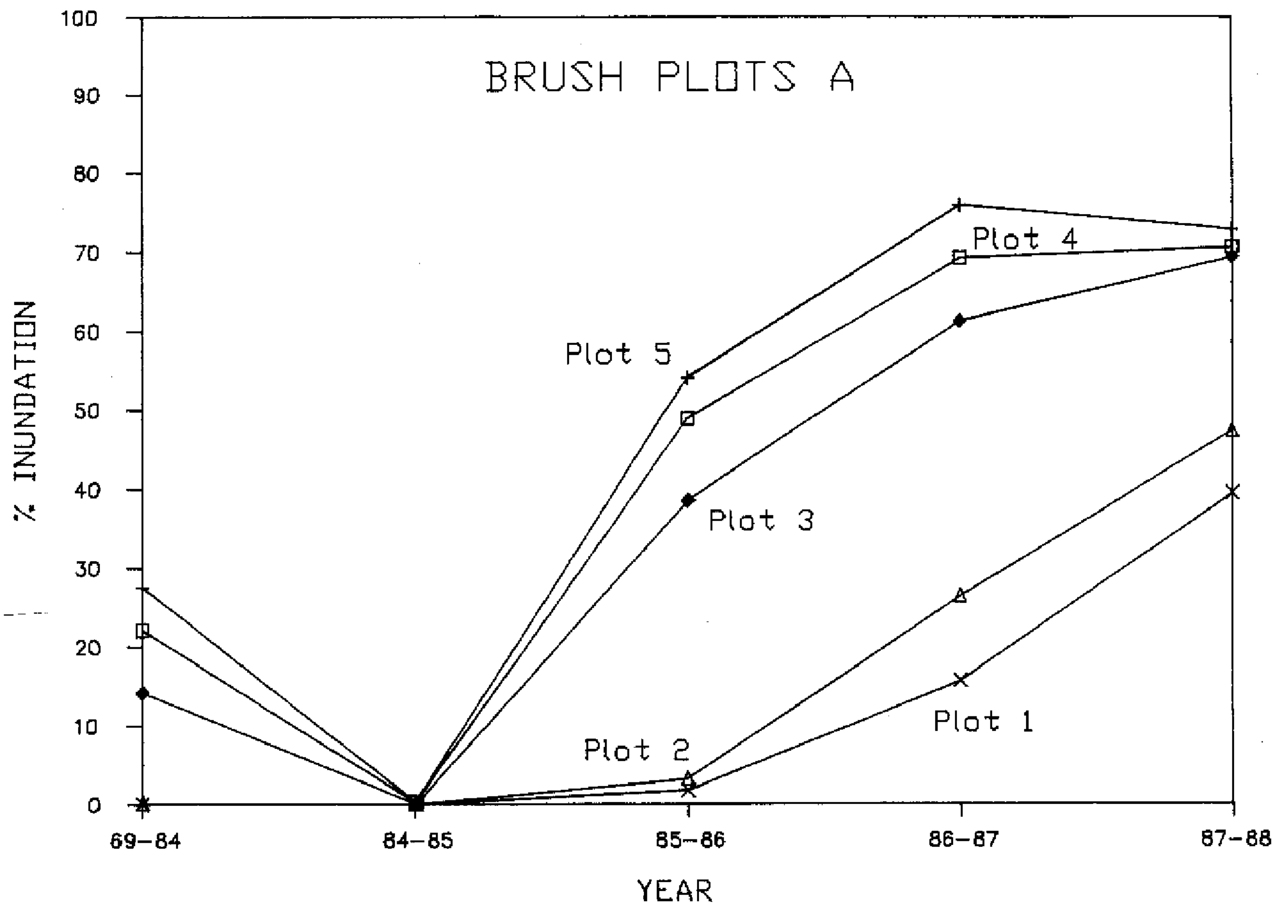

FIGURE 18. Annual hydroperiods for Brush Plots A (1969-88). Hydroperiods are expressed as the proportion of each period that each plot was inundated.

TABLE 8. Dominant plant species in Brush Transect A. Data show the number of subplots in which each plant species occurred. A total of nine subplots were sampled within each plot.

	Sampling Year																								
Species	1984					1985					1986					1987					1988				
Plots	1	2	3	4	5	1	2	3	4	5	1	2	3	4	5	1	2	3	4	5	1	2	3	4	5
Understory																									
Alternanthera philoxeroides	1	0	5	9	9	0	0	5	5	9	1	0	6	9	9	2	0	8	9	9	2	6	9	8	6
Ambrosia artemisiifolia	8	0	7	0	3	8	0	8	1	0	9	0	4	0	0	0	0	0	0	0	6	0	0	0	4
Ampelopsis arborea	7	9	0	0	0	7	9	0	0	2	8	9	0	1	2	7	9	0	0	2	9	9	0	6	0
Andropogon virginicus	1	0	4	3	4	2	0	4	1	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Boehmeria cylindrica	0	0	3	6	6	0	0	3	9	7	2	9	2	9	7	1	1	0	3	8	0	1	1	0	3
Commelina diffusa	0	0	0	5	0	0	0	0	3	5	1	5	2	3	2	1	9	0	2	0	0	0	0	0	0
Hydrocotyle sp	0	0	0	6	8	0	0	0	8	8	0	0	0	1	1	0	0	0	0	1	0	0	0	0	0
Juncus effusus	2	0	2	7	2	5	0	1	2	2	2	0	1	2	0	3	0	0	0	0	3	0	0	0	0
Ludwigia peruviana	0	0	8	3	0	0	0	5	6	0	0	0	2	1	1	1	0	2	3	0	0	0	5	3	3
Panicum hemitomon	9	6	9	6	8	5	3	9	7	8	5	5	9	9	9	9	8	9	9	8	9	9	9	9	9
Paspalum conjugatum	7	2	6	6	1	7	5	0	1	0	7	8	0	0	0	9	6	0	0	1	8	0	0	0	0
Sambucus canadensis	9	9	3	2	8	9	9	6	3	7	8	9	1	0	0	9	7	0	0	0	1	0	0	0	0
Urena lobata	9	9	9	6	3	9	9	9	7	1	9	9	9	6	1	9	9	1	0	1	8	2	1	0	0
Subcanopy/Canopy																									
Ampelopsis arborea	9	9	0	0	0	9	9	0	0	0	9	9	0	1	0	8	7	0	1	0	6	1	0	5	0
Baccharis halimifolia	9	0	2	5	0	5	0	2	7	0	4	0	3	0	0	5	0	2	0	0	6	0	0	0	0
Ludwigia peruviana	0	0	7	8	0	0	0	0	5	0	0	0	0	0	0	0	0	3	1	0	1	2	7	3	0
Myrica cerifera	0	0	4	0	5	0	0	5	0	4	0	0	6	0	5	0	0	7	0	6	0	0	8	0	4
Panicum hemitomon	0	0	5	2	0	0	1	0	1	0	1	3	2	8	1	5	6	9	9	3	6	3	8	7	0
Salix caroliniana	0	0	0	0	3	0	0	0	0	6	0	0	0	4	6	0	0	0	8	7	0	0	0	9	9
Sambucus canadensis	9	9	3	2	9	4	6	5	4	9	6	6	0	0	0	2	1	0	0	0	1	0	0	0	0

Alternanthera philoxeroides, Ampelopsis arborea, and Panicum hemitomon, expanded during the study. These species dominated the understory in 1988. Enhanced growth of P. hemitomon also was reflected by increased frequency of occurrence of this species in the subcanopy. Other changes in the canopy layer included elimination of Sambucus from all plots, loss of Baccharis from plots 3 and 4, and expansion of willow, Salix caroliniana, in plots 4 and 5.

Brush Transect B

Plot elevations ranged from 12.2 - 12.4 m, which was similar to plots 3-5 along Brush Transect A and represents the lowest 13 percent of the floodplain in this portion of Pool B. From 1969-84, plot 7 was inundated an average of 138 days each year, while annual inundation frequencies for plots 6,8,9 and 10 averaged 60-80 days (Figure 19). In 1984-85, plot 7 was inundated for 45 days, and all other plots were dry during most of the year. During the following year, inundation frequencies were 30% higher than the pre-demonstration project period and ranged from 154-235 days. From 1986-88 annual inundation frequencies of these elevations increased to 236-303 days. The backwater effect of Weir #1 resulted in additional inundation of 60-82 days in 1985-86, up to 37 days in 1986-87, and no more than 12 days in 1987-88.

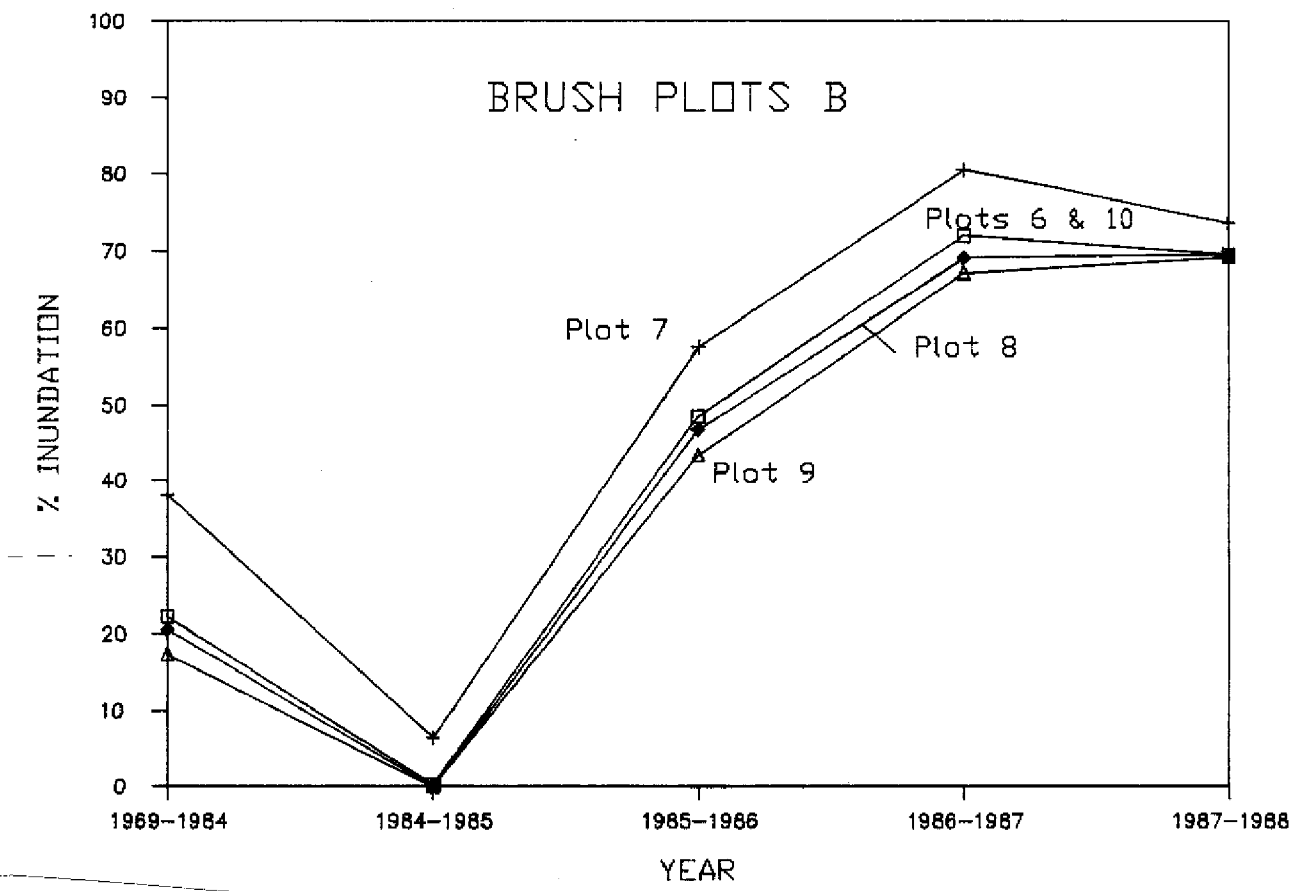

FIGURE 19. Annual hydroperiods for Brush Plots B (1969-88). Hydroperiods are expressed as the proportion of each period that each plot was inundated.

In 1984, nine plant species were common throughout this transect (Table 9). These included a vine, Melothria pendula, three typical wetland species, Alternanthera philoxeroides, Hydrocotyle sp., and Ludwigia peruviana, four transitional or seasonal wetland species, Boehmeria cylindrica, Commelina diffusa, Juncus effusus, and Sambucus canadensis, and one upland species, Urena lobata. The canopy was dominated by Sambucus, Ludwigia peruviana and Salix caroliniana.

Frequencies of occurrence of all dominant understory species except Alternanthera and Urena declined during the monitoring period (Table 9). The distribution of Sambucus initially expanded during dry conditions that occurred between 1984 and 1985 sampling dates, but frequency of occurrence of this species declined significantly when inundation frequencies increased in 1985-86. Distributions of Hydrocotyle, Ludwigia peruviana, Melothria, and Solidago fistulosa also declined markedly during 1985-86. The decline of Hydrocotyle during 1985-86 continued during the next two years, and likely was caused by increased water depths (as along Brush Transect A). Losses of Melothria and Solidago, and a concurrent increase in frequency of occurrence of Boehmeria appeared to be correlated with increased inundation frequencies. However, higher inundation frequencies during 1986-87 appear to have led to a subsequent decline in the distribution of Boehmeria. Other major changes in species composition during 1986-88 included decreased frequency of occurrence of Juncus effusus, and increased frequencies of Alternanthera, Ipomoea alba, and Sarcostemma clausa. These three species and Urena lobata were the dominant plants in the understory of this transect at the end of the monitoring period. Though not common throughout the transect, a threatened species (Ward, 1978), the climbing dayflower, Commelina gigas, colonized plot 10 in 1985-86 and spread during ensuing years.

Changes in plant species composition of the canopy were similar to those that occurred in the understory. The most significant changes were the loss of Sambucus in 1985-86 and subsequent increases in frequencies of Ipomoea alba and Sarcostemma clausa. These vines and Salix caroliniana dominated the canopy of this transect in 1988.

Duck Slough Transect

Ground elevations along this transect ranged from 12.5 - 13.0 m, but 73 percent of the transect points were at elevations between 12.7 - 12.9 m, 17 percent were < 12.7 m, and only 10 percent were > 12.9 m. At this location in the pool, 42 percent of the floodplain has elevations < 12.8 m and about 25 percent of this area lies between 12.5 and 12.8 m.

Available hydroperiod data from 1969-84 (i.e., S-65B headwater stages) suggests that these transect elevations had little or no surface water during the pre-demonstration project period (Figure 20a). However, because this transect was located at the edge of a slough which receives drainage from a tributary watershed, C-38 stages may not have consistently reflected stages that occurred along the transect. When this watershed contributes runoff, for example, a water level gradient is formed between inflows to the slough and C-38 stages. Whenever this gradient exists, stages in the slough will be higher than in C-38. In addition, field observations indicated that surface water frequently "ponds" in isolated depressions in the slough. Although accurate stage data for this slough clearly are not available for 1969-84, it is likely that hydroperiods were shorter in 1984-85 (Figure 20b), and substantially longer

TABLE 9. Dominant plant species in Brush Transect B. Data show the number of subplots in which each plant species occurred. A total of nine subplots were sampled within each plot.

	Sampling Year																								
	1984					1985					1986					1987					1988				
Species / Plots	6	7	8	9	10	6	7	8	9	10	6	7	8	9	10	6	7	8	9	10	6	7	8	9	10
Understory																									
Alternanthera philoxeroides	9	2	6	1	6	9	0	6	1	5	9	3	3	0	6	9	9	9	7	9	8	9	8	7	9
Boehmeria cylindrica	7	3	7	4	8	7	4	6	4	9	9	9	8	9	8	4	8	0	1	3	6	3	0	0	2
Commelina diffusa	9	5	8	1	9	9	5	7	1	4	9	6	5	1	5	5	8	2	2	0	6	6	1	1	0
Commelina gigas	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	9	2	0	0	0	9
Hydrocotyle umbellata	6	7	5	9	8	4	7	7	6	8	0	4	3	4	0	1	1	0	0	0	1	0	0	0	0
Ipomoea alba	1	0	4	1	0	0	0	3	1	0	1	1	1	0	1	0	0	2	0	1	6	3	9	6	1
Juncus effusus	4	6	0	3	5	4	4	0	4	4	4	4	0	2	4	2	4	0	3	2	0	0	0	0	0
Ludwigia peruviana	6	6	5	8	5	4	6	3	5	5	0	0	3	5	1	0	0	5	7	0	0	1	8	8	0
Melothria pendula	2	6	0	7	6	6	5	2	4	5	0	0	0	0	3	0	0	0	0	0	2	0	1	0	1
Panicum hemitomon	0	0	8	0	1	0	0	8	0	0	3	0	9	0	0	9	1	9	0	0	9	1	2	1	0
Paspalum conjugatum	0	3	2	4	0	0	6	3	9	0	0	8	3	9	1	0	6	0	7	1	0	2	0	4	0
Polygonum punctatum	9	0	2	0	2	4	0	0	0	0	8	2	2	1	2	0	1	0	0	0	3	2	1	0	1
Sambucus canadensis	3	7	4	5	6	8	8	7	5	8	0	0	1	2	3	0	0	1	2	2	0	0	1	0	2
Sarcostemma clausa	0	0	0	2	0	0	0	1	2	0	0	0	1	1	1	0	0	3	8	3	5	7	5	9	2
Solidago fistulosa	1	5	0	2	0	3	7	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Urena lobata	9	8	9	9	9	9	0	9	9	8	6	9	8	8	8	6	9	4	9	9	9	9	3	8	8
Subcanopy/Canopy																									
Ipomoea alba	2	0	6	0	0	0	0	6	0	0	0	0	0	0	1	0	0	2	0	2	9	2	9	5	0
Ludwigia peruviana	7	4	6	3	2	6	5	6	2	1	0	0	2	3	0	0	0	7	4	0	1	0	9	7	0
Salix caroliniana	9	9	0	9	8	8	9	0	9	9	9	9	0	9	7	9	9	0	9	9	9	9	4	9	9
Sambucus canadensis	6	8	8	7	7	8	8	9	8	9	0	0	1	0	3	0	0	0	0	3	0	0	2	0	1
Sarcostemma clausa	0	0	2	5	1	0	0	2	4	0	0	0	4	5	3	5	0	7	4	5	9	1	0	6	7
Urena lobata	2	0	4	1	0	4	3	7	2	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

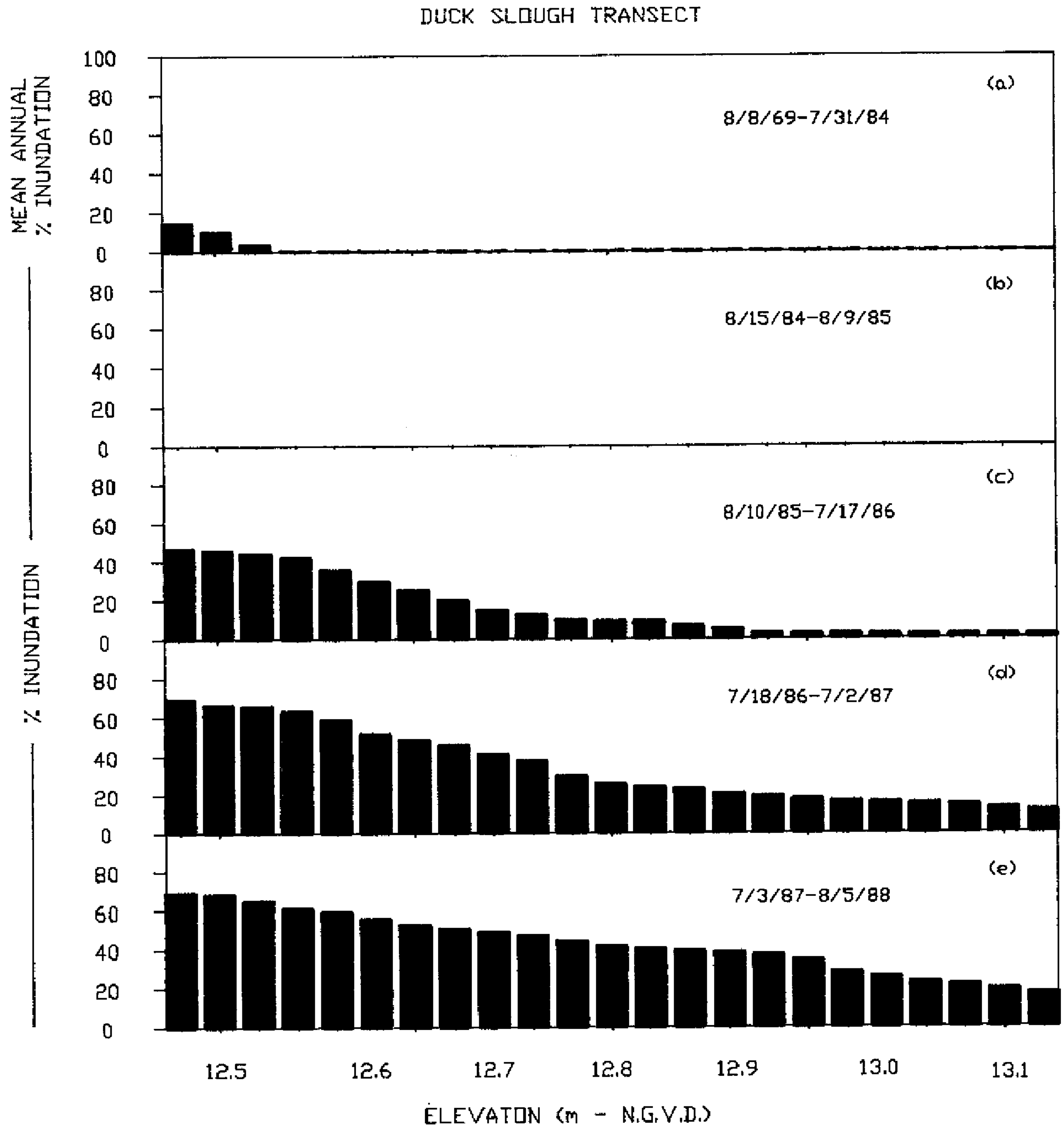

FIGURE 20. Annual hydroperiods for floodplain elevations along the Duck Slough Transect (1969-88). Hydroperiods are expressed as the proportion of each period that given elevations were inundated.

from 1985-88 (Figure 20c-e) than annual inundation frequencies during the pre-demonstration project period. Lower pool stages in 1984-85 would have increased the slope of the water level gradient between the slough and canal, thereby causing more rapid drainage of transect elevations. In 1985-86, higher pool stages produced inundation frequencies that ranged from 23-86 days along 73 percent of the transect (i.e., elevations between 12.7 - 12.9 m). Based upon pool stages, inundation frequencies of these elevations increased to 80-168 days during 1986-87, and ranged from 155-209 days in 1987-88. Backwater effects of weirs caused as many as 88 days of inundation along this transect in 1985-86, 72 days in 1986-87, and 48 days in 1987-88.

Plant species composition in 1984 (Table 10) confirmed that slough inundation frequencies during 1969-84 were greater than hydroperiods indicated by pool(canal) stages. Dominant plant taxa along this transect in 1984 were either typical marsh or seasonal wetland indicators, including Alternanthera philoxeroides, Centella asiatica, Diodea virginiana, Hydrocotyle sp., Ludwigia repens, Panicum hemitomon and Polygonum spp.(punctatum and hydropiperiodes). Most of the other common species, such as Bacopa caroliniana, Eleocharis vivipara, Hydrochloa caroliniensis, Leersia hexandra, Pontederia cordata, Proserpinaca palustris, Rhynchospora inundata, and Sagittaria lancifolia also were wetland taxa. The only species which were common in 1984 and tend to be found in more upland or drained habitats were Eupatorium capillifolium and Euthamia minor. Shifts in plant community composition in 1984-85 were indicative of lower pool stages which occurred during this year. Frequencies of xerophytic and mesophytic taxa such as Eupatorium, Ambrosia artemisiifolia, Andropogon virginicus, Baccharis halimifolia, and Oxalis florida increased, while occurrences of Bacopa caroliniana, Eleocharis vivipara, Hydrochloa, Hydrocotyle, Ludwigia repens, Polygonum, and Rhynchospora inundata declined. Most of these trends were reversed during the following two years when pool stages led to greater inundation frequencies in the slough. As a result, most of the dominant and/or common species in 1988 were similar to those found in 1984; however, frequencies of Eupatorium, Euthamia, Hydrochloa, Hydrocotyle, and Ludwigia repens exhibited a net decline, while occurrences of Cuphea carthagenensis, Leersia and Phyla nodiflora increased slightly (Table 10).

Pine Island Slough Transect

Ground elevations slope from about 13.5 m at the eastern terminus of the transect to 13.1 m at the beginning of the transect. However, due to a low, man-made berm which occurs about midway along the transect, hydroperiods along eastern and western halves of the transect were largely independent. The eastern half is within the main stream of a slough in which some surface water is maintained through most of the year (personal observations). Thus, even though ground elevations along this half of the transect are 1.1 - 1.4 m above typical pre-demonstration project pool stages, it is likely that this portion of the transect has been continuously (i.e., prior to and after channelization) exposed to long annual hydroperiods. Hydroperiods along this portion of the transect were not affected appreciably by pool stage fluctuations during the Demonstration Project, except perhaps in 1984-85, when canal stages were drawn down to extremely low levels for several months.

TABLE 10. Dominant plant species along the Duck Slough Transect. Data show the number of quadrats in which each plant species occurred during each sampling year. A total of 50 quadrats were sampled along this 366 m transect.

Species	Sampling Year				
	1984	1985	1986	1987	1988
Alternanthera philoxeroides	28	25	34	37	32
Ambrosia artemisiifolia	5	20	13	5	10
Andropogon virginicus	0	11	12	5	0
Axonopus furcatus	3	6	13	10	4
Baccharis halimifolia	1	13	2	1	1
Bacopa caroliniana	18	4	10	17	17
Boltonia diffusa.	14	21	18	16	12
Centella asiatica	37	44	44	38	31
Cuphea carthagenensis	2	2	9	12	17
Cyperus haspan	7	3	15	26	3
Cyperus polystachyos	8	2	13	6	4
Diodea virginiana	38	38	40	38	38
Echinochloa walteri	5	12	7	0	4
Eleocharis vivipara	22	5	25	26	20
Eupatorium capillifolium	17	46	14	0	8
Euthamia minor	14	7	0	0	0
Hydrochloa caroliniensis	22	11	3	9	4
Hydrocotyle sp.	39	1	13	35	8
Juncus effusus	5	7	11	13	12
Leersia hexandra	23	28	21	35	34
Ludwigia repens	35	12	30	38	24
Oxalis florida	1	17	13	4	8
Panicum hemitomon	42	46	41	48	50
Phyla nodiflora	16	21	25	21	27
Polygonum spp.	32	6	27	27	37
Pontederia cordata	17	12	9	15	10
Proserpinaca palustris	19	14	21	20	10
Rhynchospora inundata	14	4	21	25	2
Rhynchospora microcarpa	14	25	23	20	20
Sagittaria lancifolia	15	11	15	13	16

In contrast, water levels along the western half of the transect, where ground elevations range from 13.1 - 13.3 m, were influenced by C-38 drainage. Elevations along this portion of the transect represent approximately 10 percent of the floodplain in this section of the pool, and 30 percent of the floodplain in this area lies below 13.4 m. Canal stages suggest that water levels did not rise above transect elevations in 1984-85 (Figure 21b), or at any time during the 15-year pre-demonstration project period (Figure 21a); however, as in Duck Slough, the water level gradient between the main stream of Pine Island Slough and C-38 probably resulted in at least periodic inundation of the western half of the transect during this interval. Subsequently, backwater effects of the three Demonstration Project weirs caused this portion of the transect to be inundated for at least 38-67 days in 1985-86 (Figure 21c), 59-141 days in 1986-87 (Figure 21d) and 92-179 days in 1987-88 (Figure 21e).

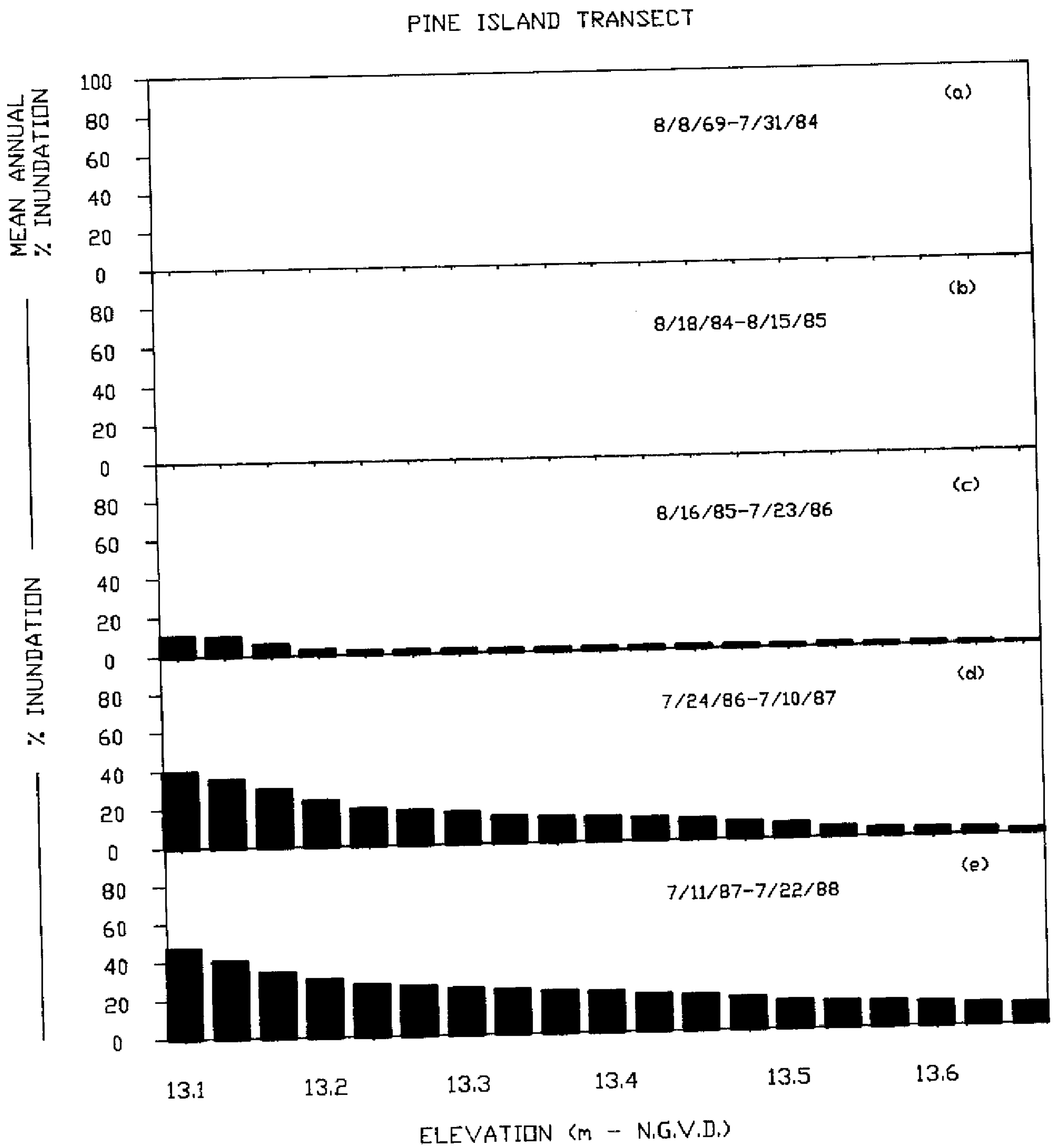

FIGURE 21. Annual hydroperiods for floodplain elevations along the Pine Island Slough Transect (1969-88). Hydroperiods are expressed as the proportion of each period that given elevations were inundated.

At the beginning of the monitoring period the plant community along the eastern half of the transect (i.e., in the main stream of Pine Island Slough) consisted of typical broadleaf marsh species, dominated by Leersia hexandra, Panicum hemitomon, Sagittaria lancifolia, Mikania scandens, Pontederia cordata, and Polygonum sp. This community showed little change from 1984-88; however, frequencies of Pontederia, Mikania, Hydrocotyle and Polygonum declined for reasons which probably were not related to the Demonstration Project (Table 11).

Plant species composition along the western half of the transect was indicative of a Rhynchospora wet prairie. In 1984, this community was dominated by Rhynchospora inundata, Leersia hexandra, Centella asiatica, Bacopa caroliniana, Diodea virginiana, Eleocharis vivipara, and Ludwigia repens. Other common species included Hydrochloa caroliniensis, Hydrocotyle sp., Panicum hemitomon, Pontederia cordata, and Proserpinaca palustris. Frequencies of Leersia and Ludwigia declined, while the distribution of an upland species, Eupatorium capillifolium, increased during dry conditions that occurred the following year. These changes were reversed as inundation frequencies increased in 1985-86. Frequencies of Centella, Hydrochloa, Hydrocotyle, and Ludwigia repens decreased, and the distribution of Panicum hemitomon increased during the last two years of the monitoring period, but the plant community represented by this half of the transect did not exhibit significant changes in overall species composition between 1984 and 1988 (Table 11).

Flow-Through Marsh Transect

Ground elevations along this transect ranged from 12.9 - 13.7 m and represented approximately 50 percent of the floodplain in this portion of the pool. Channelization completely drained this section of the Pool B floodplain and no surface water was present along transect elevations from 1969 through spring 1986 (Figure 22a-c). Following completion of the berm along the northeastern bank of C-38, drainage from the eastern edge of the floodplain was retained in the Flow-Through Marsh, and water levels in this area became independent of C-38 stages. Thus, transect inundation frequencies indicated by canal stages from August 1985 - July 1986 (Figure 22c) are underestimates, because at least some transect elevations were inundated by a 5 cm rainfall that occurred in May 1986, and most of transect was flooded after this area received 30.5 cm of rain in June. Although continuous stage data for the Flow-Through Marsh were not available until a recorder was installed in April 1987, field observations and rainfall records suggest that at least the western portion of the transect was continuously inundated from June 1986 - March 1987.

The transect remained inundated through April 1987, but was completely dry during the last half of June. It is likely that these inundation frequencies were comparable to hydroperiods in the Flow-Through Marsh in 1987-88 (Figure 22e), when stage records indicate that 85 percent of transect elevations were inundated for at least 248 days. Because drainage of water from the Flow-Through Marsh was impeded by a transverse central ridge, and spoil pile at its southern end, this area had hydrologic characteristics of an impoundment rather than a flowing system.

TABLE 11. Dominant plant species along the Pine Island Slough Transect. Data show the number of quadrats in which each plant species occurred during each sampling year. A total of 27 quadrats were sampled along the western 198 m of the transect and 33 quadrats were sampled along the eastern 221 m.

Species	Sampling Year				
	1984	1985	1986	1987	1988
Western 198 m					
Andropogon virginicus	4	10	11	5	0
Bacopa caroliniana	19	14	15	21	23
Boltonia diffusa	14	17	1	4	0
Centella asiatica	26	25	25	13	10
Cuphea carthagenensis	14	14	11	6	10
Diodea virginiana	19	17	21	19	16
Eleocharis vivipara/flavescens	18	15	8	8	6
Eupatorium capillifolium	6	21	0	0	7
Hydrochloa caroliniensis	14	7	9	1	5
Hydrocotyle sp.	14	15	10	19	5
Leersia hexandra	26	15	23	25	25
Ludwigia repens	19	7	16	21	12
Mikania scandens	11	13	15	16	16
Panicum hemitomon	16	21	16	25	26
Polygonum hydropiperoides	8	3	9	7	7
Pontederia cordata	15	17	17	9	9
Proserpinaca palustris	16	24	19	13	9
Psilocarya nitens	11	0	0	0	0
Rhynchospora inundata	26	26	26	23	21
Rhynchospora microcarpa	4	6	1	19	6
Eastern 221 m					
Bacopa caroliniana	14	9	13	21	7
Centella asiatica	9	12	11	7	8
Cephalanthus occidentalis	7	8	7	8	9
Diodea virginiana	12	9	9	9	10
Hydrocotyle sp.	9	11	4	9	0
Leersia hexandra	29	24	27	28	27
Ludwigia repens	8	11	10	11	5
Mikania scandens	27	22	26	23	9
Panicum hemitomon	29	26	27	27	28
Polygonum sp	18	4	7	6	5
Pontederia cordata	18	13	11	14	8
Rhynchospora inundata	11	11	14	8	11
Sacciolepis striata	1	8	10	7	4
Sagittaria lancifolia	22	22	20	21	21

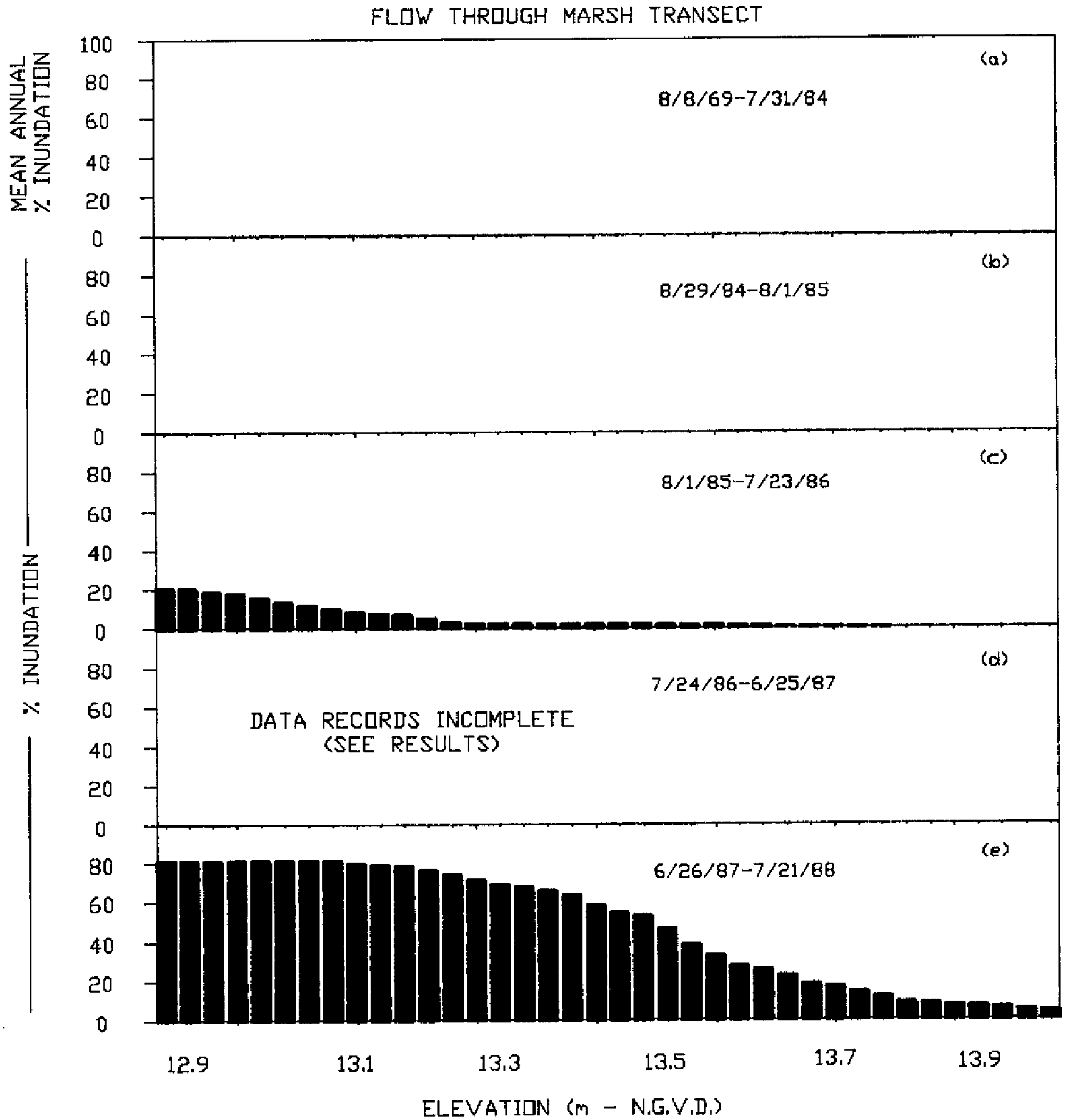

FIGURE 22. Annual hydroperiods for floodplain elevations along the Flow-Through Marsh Transect (1969-88). Hydroperiods are expressed as the proportion of each period that given elevations were inundated.

In 1984, the Flow-Through Marsh plant community was representative of a typical unimproved pasture, dominated by Eupatorium capillifolium, Centella asiatica, Axonopus spp., Ambrosia artemisiifolia, Cuphea carthagenensis, Digitaria sp., Diodea virginiana, and Polygonum sp. (probably hydropiperoides) (Table 12). Frequencies of Cuphea and Polygonum declined during 1984-85, but the plant community along this transect underwent the most significant change in 1985-86, after the berm along C-38 was completed. Subsequent inundation during May-June 1986 virtually eliminated dominant xerophytic pasture species, and a wetland community dominated by Panicum hemitomon, Diodea virginiana, Juncus effusus, Alternanthera philoxeroides, Eleocharis vivipara, and Paspalum acuminatum developed during the last two years of monitoring.

TABLE 12. Dominant plant species along the Flow-Through Marsh Transect. Data show the number of quadrats in which each plant species occurred during each sampling year. A total of 41 quadrats were sampled along this 305 m transect.

Species	Sampling Year 1984	1985	1986	1987	1988
Alternanthera philoxeroides	1	0	2	18	18
Ambrosia artemisiifolia	29	0	3	11	6
Axonopus spp.	31	28	5	5	2
Centella asiatica	40	36	12	13	14
Cephalanthus occidentalis	10	8	1	5	2
Commelina diffusa	13	3	0	0	1
Cuphea carthagenensis	29	8	6	8	12
Cyperus polystachyos	15	15	2	7	6
Digitaria sp.	27	30	5	5	9
Diodea virginiana	26	25	4	7	21
Eleocharis vivipara	5	2	6	15	15
Eupatorium capillifolium	41	40	5	3	12
Hydrochloa caroliniensis	2	2	0	4	12
Hydrocotyle sp.	19	11	0	18	6
Ipomoea sagittata	3	10	0	0	0
Juncus effusus	15	14	19	19	19
Oxalis florida	3	14	1	6	4
Panicum hemitomon	14	15	13	18	24
Paspalum acuminatum	0	0	6	7	17
Paspalum notatum	8	11	3	2	1
Polygonum spp.	29	3	3	9	13
Phyla nodiflora	6	11	16	18	13
Pontederia cordata	3	2	1	7	12

FLOODPLAIN FISH

Sample data suggest that fish utilization of an existing broadleaf marsh was influenced by pool stage manipulations during the Demonstration Project. In each of the three annual fall-winter sampling periods, highest fish densities were found and maintained during peak pool stages (Figure 23). Mean sample densities ranged from 7-14 fish/m^2 during peak stages, while only 1-4 fish/m^2 were caught during rising and falling legs of the hydrographs. This relationship resulted in a high correlation ($r = .72$) between average marsh fish density and water depth (Figure 24). Most of the fish collected were small forage fish. Seventy-nine percent were mosquitofish, Gambusia affinis, and 18 percent were least killifish, Heterandria formosa. The remaining 3 percent were found only in samples collected during peak stages and included bluefin killifish, Lucania goodei, sailfin molly, Poecilia latipinna, flagfish, Jordanella floridae, golden topminnow, Fundulus chrysotus, pirate perch, Aphredoderus sayanus, bluespotted sunfish, Enneacanthus gloriosus, warmouth, Lepomis gulosus, bluegill, Lepomis macrochirus, brown bullhead, Ictalurus nebulosus, and tadpole madtom, Noturus gyrinus.

FLOODPLAIN INVERTEBRATES

Marsh Transect

Numerically dominant invertebrate groups along this transect included Chironomidae larvae, Amphipoda, Isopoda, small Crustacea (Copepoda, Cladocera, and Ostracoda), Coleoptera (larvae and adults) and Oligochaeta (Table 13). Sample densities showed seasonal differences and were influenced by the length of inundation prior to sampling. Sample densities during spring were much higher than densities in fall samples, particularly when spring samples were preceded by a prolonged period of inundation. For example, during March 1987, after sample elevations had been inundated for approximately 255 days, samples yielded a mean of 17,480 ($s = 3675$) invertebrates per square meter of marsh, including almost 10,000 chironomid larvae/m^2 and over 4,000 cladocerans and copepods/m^2. In April 1988, following 289 days of inundation, samples contained a mean of 6,716 ($s = 3156$) chironomid larvae, 7,450 ($s = 4605$) amphipods, 1,874 ($s = 2131$) isopods, 4,229 ($s = 4196$) cladocerans and copepods, and 2,752 ($s = 1371$) other invertebrates per square meter of marsh. Sample densities were considerably lower in spring 1985 (3,708 total invertebrates/m^2), when sample elevations had been inundated for only 40 days prior to sampling, and during all fall sampling dates regardless of the length of inundation preceding the sampling period. Mean invertebrate densities in September 1984 samples (2,394 total invertebrates/m^2), for example, were similar to densities found in fall 1985-87 (2,332-3,550 total invertebrates/m^2) even though the period of inundation preceding 1984 samples was three times longer (approximately 280 days) than the length of inundation preceding subsequent fall samples.

Drained Floodplain Transect

The only set of samples from the seasonally inundated elevations which typify this transect yielded a mean density of 39,438 ($s = 3345$) invertebrates/m^2 - the highest floodplain invertebrate density found during the monitoring period. The most abundant invertebrate taxa were chironomid larvae (15,737/m^2), copepods (7,023/m^2) and cladocerans (10,851/m^2) (Table 14). Sample elevations on this late fall sampling date (10 December 1987) had been inundated for 72-145 previous days. Chironomid larvae were twice as abundant at the sample elevation that had been inundated for 145 days compared to samples from elevations that had been inundated for 72 - 80

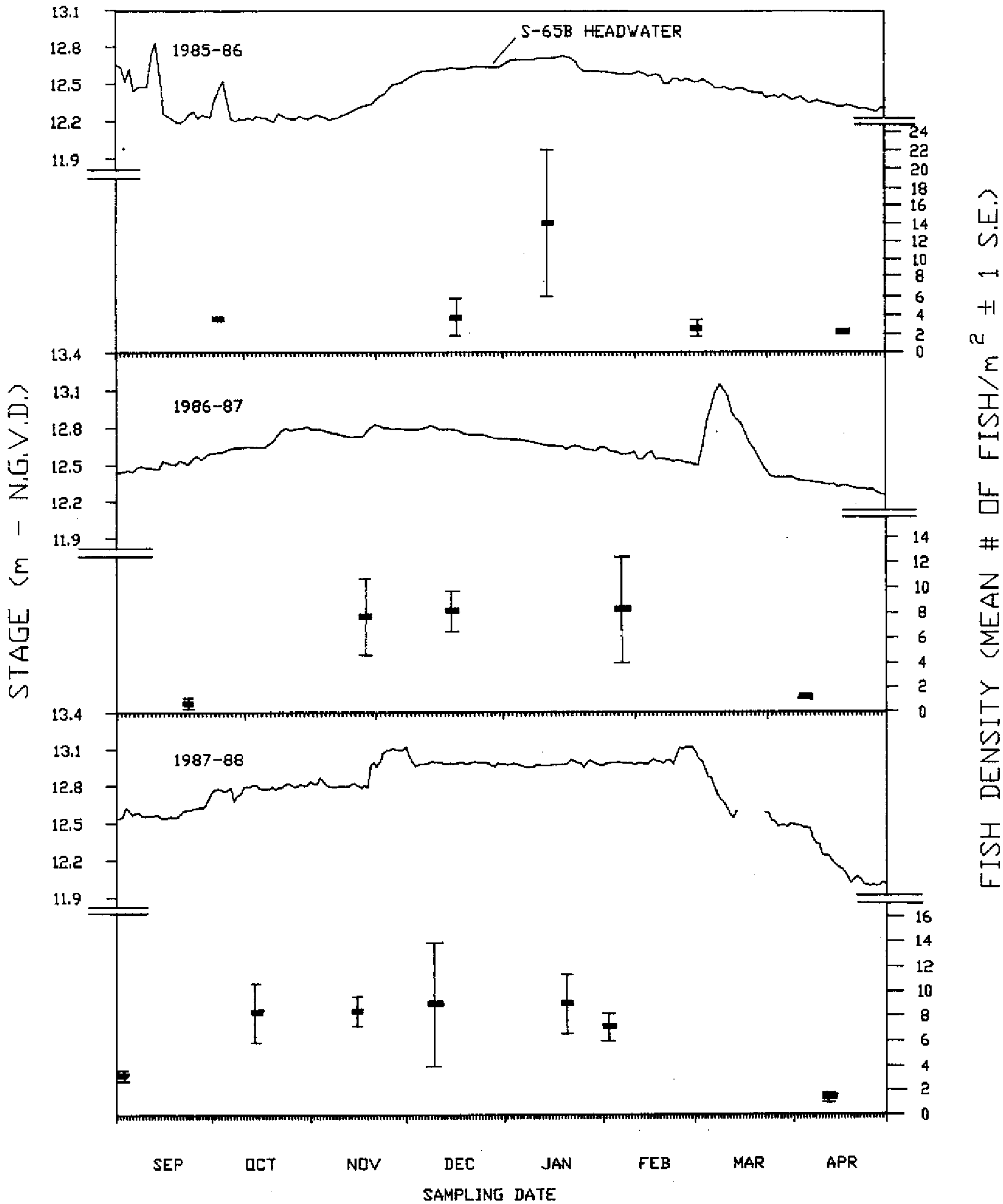

FIGURE 23. Marsh fish densities in relation to S-65B stage hydrographs.

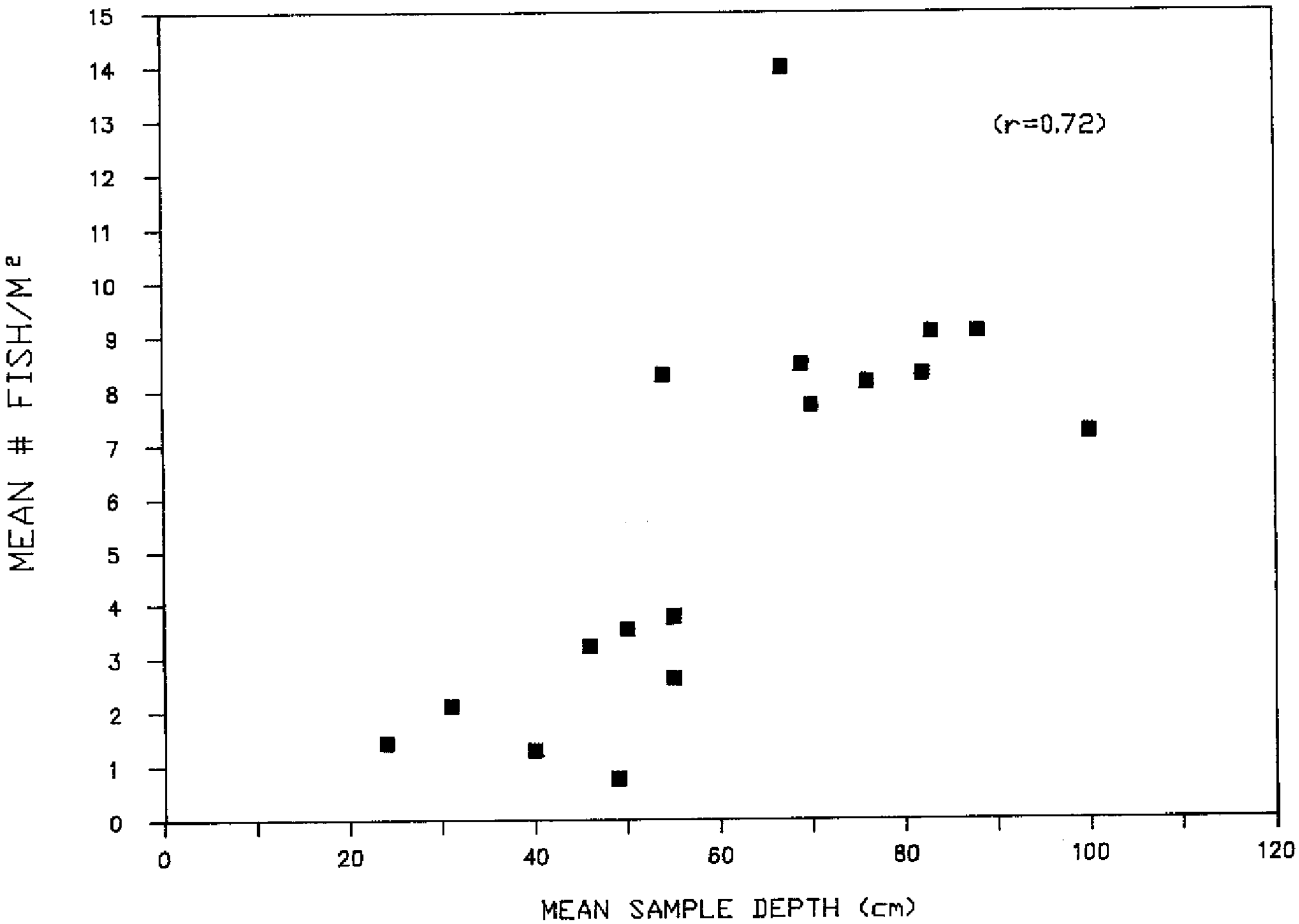

FIGURE 24. Marsh fish density in relation to sample depth. Data points show means of triplicate samples from 17 sampling dates during 1985-88.

days. Mean sample density was almost two times higher than invertebrate densities found four months later along the adjacent Marsh Transect, where sample elevations had been inundated for 289 days. However, compared to these marsh samples, large crustaceans, particularly isopods, comprised a smaller proportion of the invertebrate community of this seasonally inundated section of floodplain.

Brush Transects

Sample densities and invertebrate community structure were similar along the two seasonally inundated Brush Transects (Tables 15 and 16). Total densities ranged from 6,305-16,247 invertebrates per square meter of inundated floodplain. Chironomid larvae were the dominant (numerically) invertebrate group along both transects; however, significantly (Date * Site ANOVA: $p(F) < 0.05$) higher densities were found along Transect B than A. Average chironomid densities along Transect A ranged from 3,565-6,468 larvae/m^2 (Table 15) compared to 5,368-9,147 larvae/m^2 along Transect B (Table 16). Chironomid and total invertebrate densities generally

TABLE 13. Mean invertebrate densities ($\#/m^2$) in samples taken along Marsh Transect.

Taxa	Sampling Date	9/84	1/85	4/85	9/85	9/86	3/87	10/87	4/88
	Water Depth (cm)	6-25	4-12	9-10	40-58	38-55	13-31	15-75	16-27
	Inundation Duration (da)	280	70	40	109	94	255	15-91	289
Diptera	Chironomidae	192	212	1,114	1,043	1,516	9,599	1,197	6,765
	Ceratopogonidae	0	25	27	16	155	685	51	33
	Other	12	8	177	57	59	33	135	33
Trichoptera	Leptoceridae, Hydroptilidae	0	0	0	0	0	0	0	82
Ephemeroptera	<u>Caenis</u> <u>diminuta</u>	0	0	0	16	49	660	0	750
	Baetidae	0	0	0	0	0	98	3	0
Odonata	Anisoptera	4	16	0	8	90	187	66	32
	Zygoptera	0	0	0	8	41	49	0	65
Coleoptera	Dytiscidae Larvae	86	0	14	253	171	25	48	221
	Hydrophilidae Larvae	12	0	41	65	302	57	43	41
	Other Larvae	33	8	14	41	49	0	43	16
	Dytiscidae Adults	0	25	109	25	49	41	36	114
	Noteridae Adults	0	139	258	8	33	82	26	0
	Other Adults	0	8	109	8	8	0	15	8
Hemiptera	Nymphs and Adults	8	0	28	16	26	114	51	179
Lepidoptera	Larvae	0	0	0	25	8	0	10	187
Isopoda	Asellidae	501	465	869	25	114	65	18	1,874
Amphipoda	<u>Hyallela</u> <u>azteca</u>	86	8	0	8	0	685	5	3,293
	Gammaridae	1,231	742	666	122	147	0	46	4,157
Decapoda	Astacidae	167	16	0	1	11	8	0	5
	<u>Palaemonetes</u> <u>paludosus</u>	0	17	41	0	0	1	5	84
Ostracoda		4	0	0	163	310	98	26	171
Copepoda		0	130	217	8	16	2,624	329	3,724
Cladocera		0	0	0	49	130	1,418	115	505
Oligochaeta		33	228	883	310	49	416	74	82
Gastropoda	Ancylidae	0	16	27	8	8	171	0	98
	Planorbidae	0	0	14	8	131	49	3	62
	Other	0	0	27	16	33	90	3	164
Hydracarina, Hirudinea, Nematoda, Collembola		12	16	28	16	49	122	51	262

TABLE 14. Mean invertebrate densities ($\#/m^2$) in samples taken along Drained Floodplain Transect on 10 December 1987.

Taxa	Water Depth (cm) Sample Elevation (m) Inundation Duration (da)	31 12.63 72	47 12.47 145	40 12.54 80
Diptera	Chironomidae	14,347	25,307	10,970
	Ceratopogonidae	173	545	907
	Other Larvae	265	550	194
Ephemeroptera	*Caenis diminuta*	418	764	509
	Baetidae	51	41	51
Trichoptera	Hydroptilidae	132	132	224
	Leptoceridae	41	41	0
Odonata	Anisoptera	10	51	61
	Zygoptera	214	377	41
Coleoptera	Dytiscidae Larvae	346	351	41
	Hydrophilidae Larvae	71	102	234
	Other Larvae	20	71	0
	Adults	61	20	0
Hemiptera	Nymphs	20	306	10
	Adults	51	31	31
Lepidoptera	Larvae	31	82	0
Isopoda	Asellidae	0	10	0
Amphipoda	*Hyalella azteca*	438	346	61
	Gammaridae	168	66	0
Decapoda		36	5	10
Ostracoda		41	82	214
Copepoda		6,667	9,300	5,103
Cladocera		11,948	12,182	8,424
Oligochaeta		978	234	1,049
Gastropoda	Ancylidae	148	31	20
	Planorbidae	143	132	407
	Physidae	204	66	71
	Other	92	194	224
Collembola		0	31	0
Hydracarina		183	61	31
Turbellaria		143	255	51
Hirundinea		20	20	0
Nematoda		41	31	61

TABLE 15. Mean invertebrate densities ($\#/m^2$) in samples taken near Brush Plots A.

Taxa	Sampling Date Water Depth (cm) Inundation Duration (da)	1/16/86 22-53 70-86	2/17/86 13-55 105-222	1/15/87 23-65 13-185	2/25/87 25-45 218-257	1/15/88 14-68 58-181	4/14/88 13-38 272-290
Diptera	Chironomidae	6,860	3,708	4,092	5,348	3,706	4,477
	Ceratopogonidae	71	82	644	92	33	3
	Other	153	31	227	10	54	10
Trichoptera	Leptoceridae, Hydroptilidae	71	92	28	92	110	79
Ephemeroptera	<u>Caenis</u> <u>diminuta</u>	112	122	558	1,742	1,342	703
	Baetidae	20	0	0	0	10	0
Odonata	Anisoptera	61	61	0	31	33	17
	Zygoptera	71	41	76	143	199	60
Coleoptera	Dytiscidae Larvae	112	20	43	10	3	1
	Hydrophilidae Larvae	10	41	82	0	0	0
	Other Larvae	173	92	239	10	0	0
	Adults	10	10	30	10	20	0
Hemiptera	Nymphs and Adults	71	31	0	20	9	20
Lepidoptera	Larvae	61	0	8	10	0	0
Isopoda	Asellidae	0	0	0	0	0	41
Amphipoda	<u>Hyallela</u> <u>azteca</u>	489	591	586	1,854	1,797	1,082
	Gammaridae	143	0	10	0	0	377
Decapoda		0	0	9	30	4	1
Ostracoda		31	20	51	51	2,385	513
Copepoda		1,599	285	357	214	1,314	43
Cladocera		3,922	306	364	479	1,123	285
Oligochaeta		122	82	466	41	323	55
Gastropoda	Ancylidae	357	163	26	10	69	10
	Planorbidae	285	31	28	31	71	0
	Physidae	20	51	3	10	117	0
	Other	367	122	12	31	74	0
Pelecypoda	<u>Corbicula</u>	0	0	0	0	0	54
Nematoda		102	204	48	71	176	86
Hydracarina, Hirudinea, Collembola		122	92	31	71	109	268

TABLE 16. Mean invertebrate densities ($\#/m^2$) in samples taken near Brush Plots B.

Taxa	Sampling Date Water Depth (cm) Inundation Duration (da)	1/16/86 20-51 69-84	2/17/86 20-33 110-118	1/15/87 45-69 149-179	2/25/87 23-34 209-224	1/15/88 62-65 181	4/14/88 12-24 271
Diptera	Chironomidae	8,434	5,460	9,616	6,122	7,479	6,612
	Ceratopogonidae	112	102	998	153	61	20
	Other	173	20	122	20	20	0
Trichoptera	Leptoceridae, Hydroptilidae	15	71	82	122	122	61
Ephemeroptera	<u>Caenis</u> <u>diminuta</u>	41	214	407	1,151	1,431	713
	Baetidae	31	0	0	0	6	0
Odonata	Anisoptera	61	112	122	41	54	3
	Zygoptera	71	82	61	71	247	92
Coleoptera	Dytiscidae Larvae	92	10	20	10	13	0
	Hydrophilidae Larvae	163	163	31	0	6	0
	Other Larvae	31	20	10	10	26	10
	Adults	0	10	0	20	3	0
Hemiptera	Nymphs and Adults	112	92	20	20	112	31
Lepidoptera	Larvae	71	20	0	31	48	0
Isopoda	Asellidae	31	0	0	0	0	0
Amphipoda	<u>Hyallela</u> <u>azteca</u>	1,070	1,049	917	907	787	509
	Gammaridae	31	0	0	0	0	0
Decapoda		0	12	31	62	1	10
Ostracoda		51	41	41	31	670	234
Copepoda		2,526	132	255	163	586	20
Cladocera		1,477	10	418	214	227	183
Oligochaeta		82	51	51	20	59	31
Gastropoda	Ancylidae	866	143	143	183	125	10
	Planorbidae	234	82	10	10	61	3
	Physidae	10	0	0	20	301	20
	Other	275	71	31	51	56	20
Nematoda		122	31	143	41	329	224
Hirudinea		0	31	20	10	155	126
Hydracarina,	Collembola	41	10	20	20	69	82

were highest during the first sampling date of each year. The decline in sample densities between the two sample dates of each year occurred in a 1-3 month interval during which overbank flow ceased, and stages along the transects decreased. The Ephemeroptera nymph, Caenis diminuta, and amphipod, Hyalella azteca, also were abundant along both transects during most sampling periods. Except for an absence of isopods, overall invertebrate community structure along these transects was similar to that found along the Marsh Transect.

Duck Slough Transect

The variable periods of inundation preceding Duck Slough sampling dates (Table 17) facilitated tracking of invertebrate colonization patterns. January 1986 samples, for example, reflected an invertebrate assemblage that had developed after only 19 days of inundation. These samples were dominated by oligochaetes (2,876/m^2), and low densities of chironomid larvae (720/m^2), copepods (832/m^2), and cladocerans (683/m^2). In subsequent samples, which were preceded by longer periods of inundation, numerical importance of oligochaetes declined, and densities of chironomids, copepods and cladocerans comprised at least 80 percent of the total number of sampled invertebrates. In February 1987, following 39 days of inundation, samples yielded an average of 6,220 (s = 2179) chironomid larvae, 3,507 (s = 3370) cladocerans and 2,750 (s = 2971) copepods per square meter of wetted area. Although mean chironomid density was lower (1,793/m^2) in November 1987 samples, densities of chironomid larvae and copepods were at least two times greater in the sample taken from an elevation that had been inundated for 121 previous days, than in samples from elevations that had been inundated for 48 days prior to this sampling date (Table 17). Overall, invertebrate composition of samples collected along this transect was similar to that found at seasonally inundated elevations along the Drained Floodplain Transect.

Flow-Through Marsh Transect

Aquatic invertebrate community composition in this newly reestablished wetland (Table 18) was similar to invertebrate assemblages along the Drained Floodplain and Duck Slough Transects. Initial samples (January 1987) followed approximately 200 days of inundation and were dominated by chironomids (3743 larvae/m^2) and Caenis diminuta (1431 nymphs/m^2). In December 1987, when sample elevations had been exposed to 91-143 days of preceding inundation, chironomid larvae and particularly Caenis nymphs were less abundant, while densities of copepods (5765/m^2), cladocerans (1823/m^2), and oligochaetes (810/m^2) were higher. Following another four months of additional flooding (i.e., a total of 283 days) samples were numerically dominated by chironomids (3630 larvae/m^2) again.

TABLE 17. Invertebrate densities (#/m^2) in samples along Duck Slough Transect.

Taxa		1/86 (Means)	2/87 (Means)	11/87	11/87	11/87
	Sampling Date	1/86 (Means)	2/87 (Means)	11/87	11/87	11/87
	Water Depth (cm)	12-20	13-26	8	28	11
	Inundation Duration (da)	19	39	47	121	48
Diptera	Chironomidae	720	6,220	397	3,097	1,884
	Ceratopogonidae	112	82	0	41	10
	Other	58	24	245	61	41
Trichoptera	Leptoceridae	0	0	0	10	20
Ephemeroptera	*Caenis diminuta*	0	136	295	51	112
	Baetidae	0	37	20	41	41
Odonata	Anisoptera	3	7	0	31	41
	Zygoptera	0	0	21	21	51
Coleoptera	Dytiscidae Larvae	34	88	194	71	255
	Hydrophilidae Larvae	8	31	346	61	61
	Other Larvae	71	44	0	10	41
	Adults	122	27	295	51	143
Hemiptera	Nymphs and Adults	0	0	21	0	0
Lepidoptera	Larvae	3	0	0	0	10
Amphipoda	*Hyallela azteca*	17	642	214	509	173
	Gammaridae	27	0	132	0	82
Decapoda		3	3	0	31	31
Ostracoda		24	41	10	112	31
Copepoda		832	2,750	2,985	11,429	4,665
Cladocera		683	3,507	407	907	1,100
Oligochaeta		2,876	336	366	845	71
Gastropoda	Ancylidae	3	27	0	82	0
	Planorbidae	17	635	183	71	367
	Physidae	0	3	112	10	51
	Other	20	14	0	0	0
Nematoda		228	153	61	0	20
Hirudinea		10	3	0	0	0
Hydracarina, Collembola		17	187	295	112	173

TABLE 18. Mean invertebrate densities (#/m^2) in samples taken along Flow-Through Marsh Transect.

Taxa	Sampling Date Water Depths (cm) Inundation Duration (da)	1/87 30-69 206	12/87 37-46 91-143	4/88 24-42 283
Diptera	Chironomidae	3,779	1,891	3,643
	Ceratopogonidae	145	34	217
	Other Larvae	76	156	3
Ephemeroptera	<u>Caenis diminuta</u>	1,431	65	801
	Baetidae	15	93	27
Trichoptera	Hydroptilidae	23	14	31
	Leptoceridae	48	3	17
Odonata	Anisoptera	64	10	0
	Zygoptera	13	44	14
Coleoptera	Dytiscidae Larvae	3	0	3
	Hydrophilidae Larvae	262	71	68
	Other Larvae	23	20	14
	Adults	3	15	41
Hemiptera	Nymphs and Adults	18	180	122
Lepidoptera	Larvae	0	3	14
Amphipoda	<u>Hyalella azteca</u>	8	0	401
	Gammaridae	0	0	14
Decapoda		13	14	0
Ostracoda		0	37	0
Copepoda		593	5,765	109
Cladocera		448	1,827	48
Oligochaeta		36	812	535
Gastropoda	Ancylidae	18	112	0
	Planorbidae	26	177	0
	Physidae	10	22	54
	Other	117	0	0
Collembola		3	48	3
Hydracarina		3	163	58
Hirundinea		5	0	0
Nematoda		3	0	0

RIVER CHANNEL HABITAT CHARACTERISTICS

Bottom Sediments

Preliminary sampling revealed bottom deposits consisting of mixtures of condensed to flocculent organic material, marl, and sand, which generally overlaid a homogenous, sand substrate. Based upon nine random samples from R3, organic matter content of these deposits varied between 2-26 percent of the dry weight of samples; total Kjeldahl nitrogen concentrations averaged 0.60 percent, and the mean phosphorus concentration was 0.107 percent.

Organic deposits in remnant river channels adjacent weir locations (R1-3) were greater than accumulations in the run (R4) draining Pine Island Slough. Prior to weir construction, mean depth of organic deposits along cross-sections in R1-3 ranged from 14.8 - 16.8 cm while accumulations in R4 averaged 6.9 cm (Figure 25). Deposits greater than 5 cm deep accounted for 64-74 percent of point samples in R1-3, compared to 40 percent of the measurements taken in R4 (Figure 26).

Final sediment samples (November 1988) showed that organic deposits had been reduced along 23 of 25 cross-sections in river channels adjacent the three Demonstration Project weirs (Table 19). Along these 23 cross-sections, an average of 66 percent of the cross-sectional area of organic deposits was either swept away by reintroduced flow or covered with a layer of clean sand. Mean depths of organic deposits fell to 7.1 cm in R1, 7.8 cm in R3 and 3.1 cm in R2 (Figure 25); no measurable organic deposits were found at 60 percent of point samples in R1 and R2, and 46 percent of bottom samples from R3 (Figure 26).

Organic deposits along sampled cross-sections in the run draining Pine Island Slough (R4) did not show significant change during the course of the Demonstration Project, while slight accretion occurred along two cross-sections at the mouth of R3 (Table 19). Both of these R3 cross-sections were located upstream of weir #3 and were not subject to flushing like the other 11 sampled cross-sections in this run.

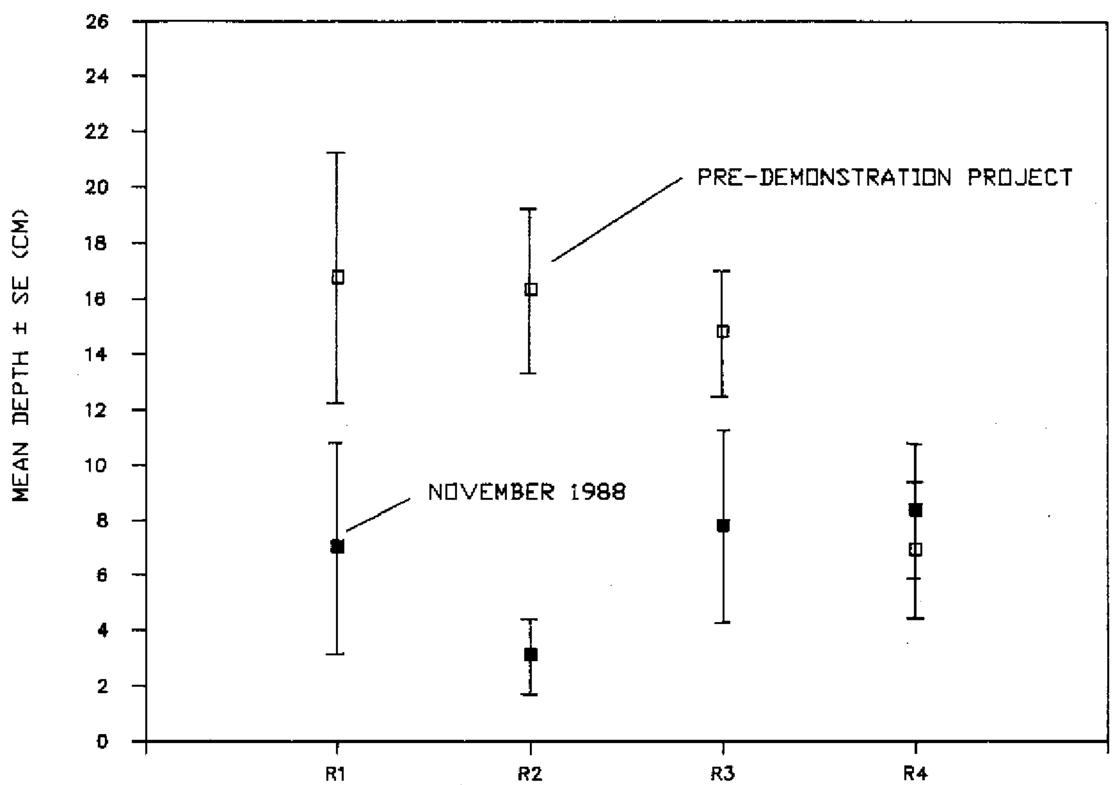

FIGURE 25. Mean depths of organic deposits in sampled river runs. Pre-demonstration project samples were taken prior to reestablishment of flow, while November 1988 samples were taken following three years of restored flow.

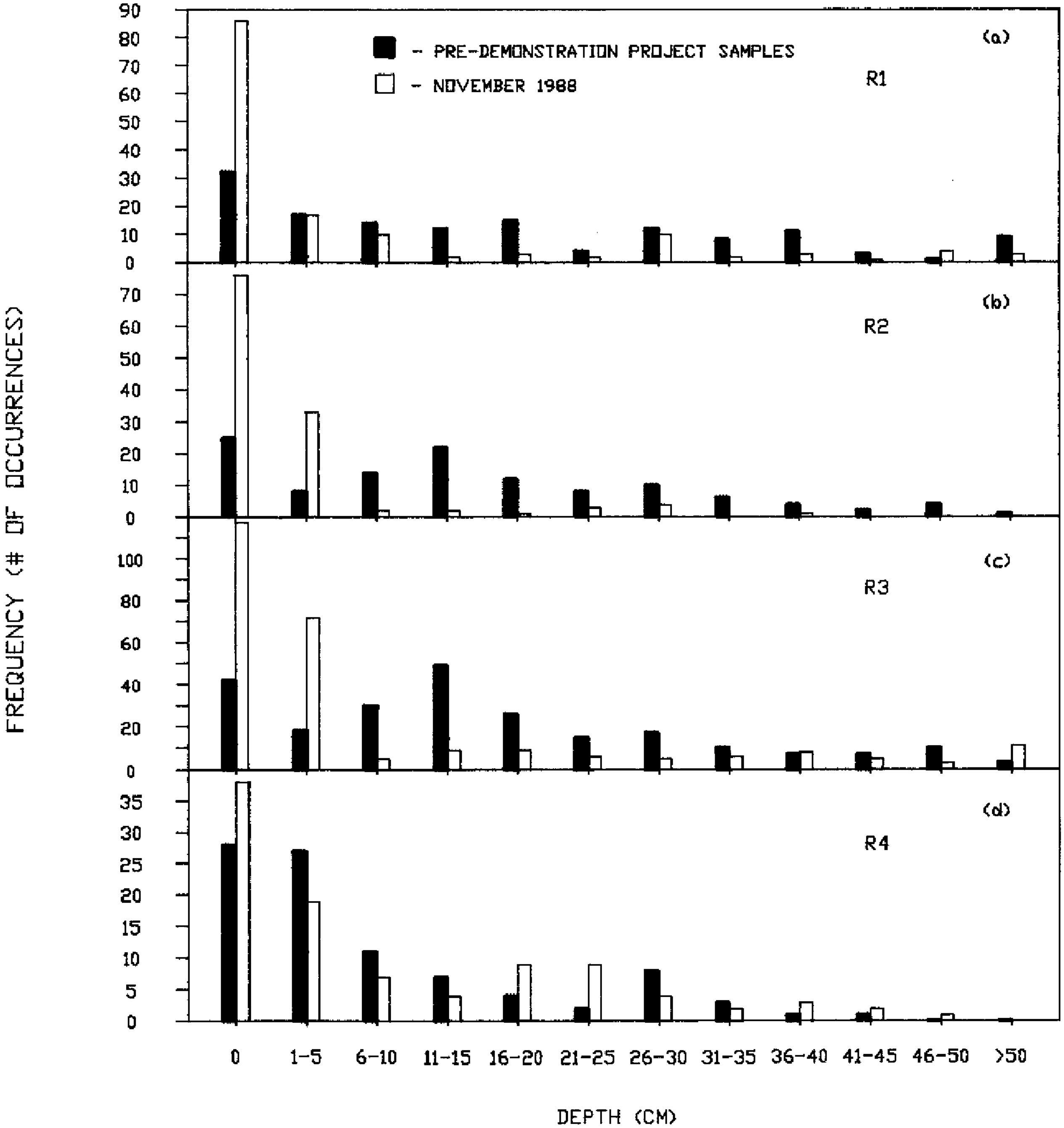

FIGURE 26. Frequency distributions of depths of organic deposits in sampled river runs. Solid bars represent samples taken prior to reestablishment of flow and open bars show distributions following three years of restored flow.

TABLE 19. Change in cross-sectional areas of the river channel and organic matter layers during the Demonstration Project (1984-88). River channel cross-sectional areas were computed from the water surface to hard bottom (see Methods).

Site	Cross-Sectional Area (m^2): River Channel 1984/85	River Channel 1988	River Channel Change	Organic Layer 1984/85	Organic Layer 1988	Organic Layer Change
R1						
1	72.43	76.05	2.02	14.51	9.85	-4.66
2	61.56	71.11	9.55	5.84	2.05	-3.79
2.5	50.17	63.50	13.33	3.19	0.00	-3.19
3	69.38	78.49	9.11	4.84	0.88	-3.96
3.5	66.03	64.54	-1.49	2.84	2.12	-0.72
4	103.99	108.42	4.43	5.94	4.08	-1.86
R2						
1	28.73	30.45	1.72	7.07	1.42	-5.66
1.5	29.53	46.42	16.89	4.89	0.44	-4.45
2	28.61	27.12	-1.49	4.58	2.19	-2.39
2.5	42.65	52.76	10.11	3.53	0.02	-3.51
3	43.76	46.73	2.97	2.10	0.98	-1.12
4	72.38	63.27	-9.11	3.76	0.82	-2.94
R3						
1	46.91	47.25	0.34	15.13	19.86	4.73
2	42.19	45.66	3.47	4.69	5.08	0.39
3	44.05	55.88	11.83	7.95	0.91	-7.04
4	107.87	118.88	11.01	6.15	1.33	-4.81
5	24.78	34.20	9.42	4.44	0.15	-4.28
6	32.32	50.09	17.77	3.10	0.00	-3.10
7	49.91	56.97	7.06	1.92	1.49	-0.43
8	54.83	55.28	0.45	2.36	1.88	-0.47
9	32.31	37.12	4.81	3.16	1.37	-1.79
10	29.23	29.94	0.71	2.72	2.32	-0.40
11	24.53	25.02	0.49	2.48	0.50	-1.98
12	28.49	29.66	1.17	1.15	0.44	-0.71
13	24.53	38.24	13.71	2.18	0.00	-2.18
R4						
1	4.51	11.35	6.84	0.04	0.42	0.37
2	27.12	25.51	-1.61	2.59	3.20	0.61
2.5	52.34	49.55	-2.79	4.72	4.52	-0.20
2.8	40.52	37.68	-2.84	2.09	3.26	1.17
3	24.59	25.72	1.13	1.19	1.14	-0.05
4	24.29	23.42	-0.87	0.47	0.57	0.10

Channel Morphology

Due to a reduction of organic deposits, most sites showed an increase in cross-sectional area (Table 19). In addition, reintroduced flow significantly altered channel shape at 10 sampled cross-sections, including 5 of 6 sites in R2, 4 sites in R3, and 1 site in R4. At these sites, channel morphology typically changed from a uniformly shallow, saucer-shaped cross-section, to a deeper, U-shaped channel (Figure 27b), or a deep channel opposite a sand-bar (Figure 27a).

RIVER CHANNEL INVERTEBRATES

Mid-Channel Sites

Invertebrate densities and community composition of the five sampled runs displayed somewhat different patterns of change during the monitoring period, particularly during the first six sampling periods (i.e., July 1984 - August 1986) (Figure 28a-e). Variability of average sample densities from the run adjacent Weir #1 was due primarily to the abundance of <u>Chaoborus</u> <u>punctipennis</u> larvae (Figure 28a). <u>Chaoborus</u> and total sample densities from R1 increased from December 1984 - April 1985, decreased sharply in August 1985, and remained at lower densities through April 1987. Large numbers of <u>Chaoborus</u> (mean = 9,391 larvae/m^2) reappeared in mid-channel samples from R1 in August 1987.

Invertebrate densities and community structure in the run adjacent Weir #2 (Figure 28b) displayed a pattern similar to R1, except density continued to increase from April - August 1985 due to increased abundance of oligochaetes (mean = 7,456/m^2). Invertebrate density in R2 fell in December 1985 and stayed fairly constant until August 1987, when <u>Chaoborus</u> larvae again dominated the mid-channel benthic community.

As in R1 and R2, <u>Chaoborus</u> larvae dominated initial samples (July and December 1984), as well as August 1987 samples, from the run adjacent Weir #3 (Figure 28c). However, from April 1985 - April 1987 other taxa formed major components of the R3 mid-channel benthic community. The increase in total invertebrate densities from December 1984 - August 1985, for example, was due to increased abundance of Chironomidae larvae in April, and oligochaetes and ostracods in August. Mid-channel invertebrate density in R3 decreased slightly in December 1985, but, unlike the lower two runs, increased in April 1986. Between these sampling periods, ostracods and the amphipod, <u>Hyalella</u> <u>azteca</u>, increased in abundance and, together with Chironomidae larvae and oligochaetes, dominated April 1986 samples. Although density of oligochaetes increased in August 1986, total invertebrate densities in R3 decreased from April - November 1986. In April 1987, increased abundance of the clam, <u>Corbicula</u> <u>fluminea</u>, brought the average density back up to a level comparable to April 1986 samples. This density was maintained in August 1987 when <u>Chaoborus</u> replaced <u>Corbicula</u> as the dominant invertebrate taxon in R3 samples.

Total invertebrate densities in mid-channel samples from R4 (Figure 28d) showed a pattern of variation similar to R3 samples. A fairly constant density of approximately 8,700 invertebrates/m^2 were found at R4 sites during each sampling

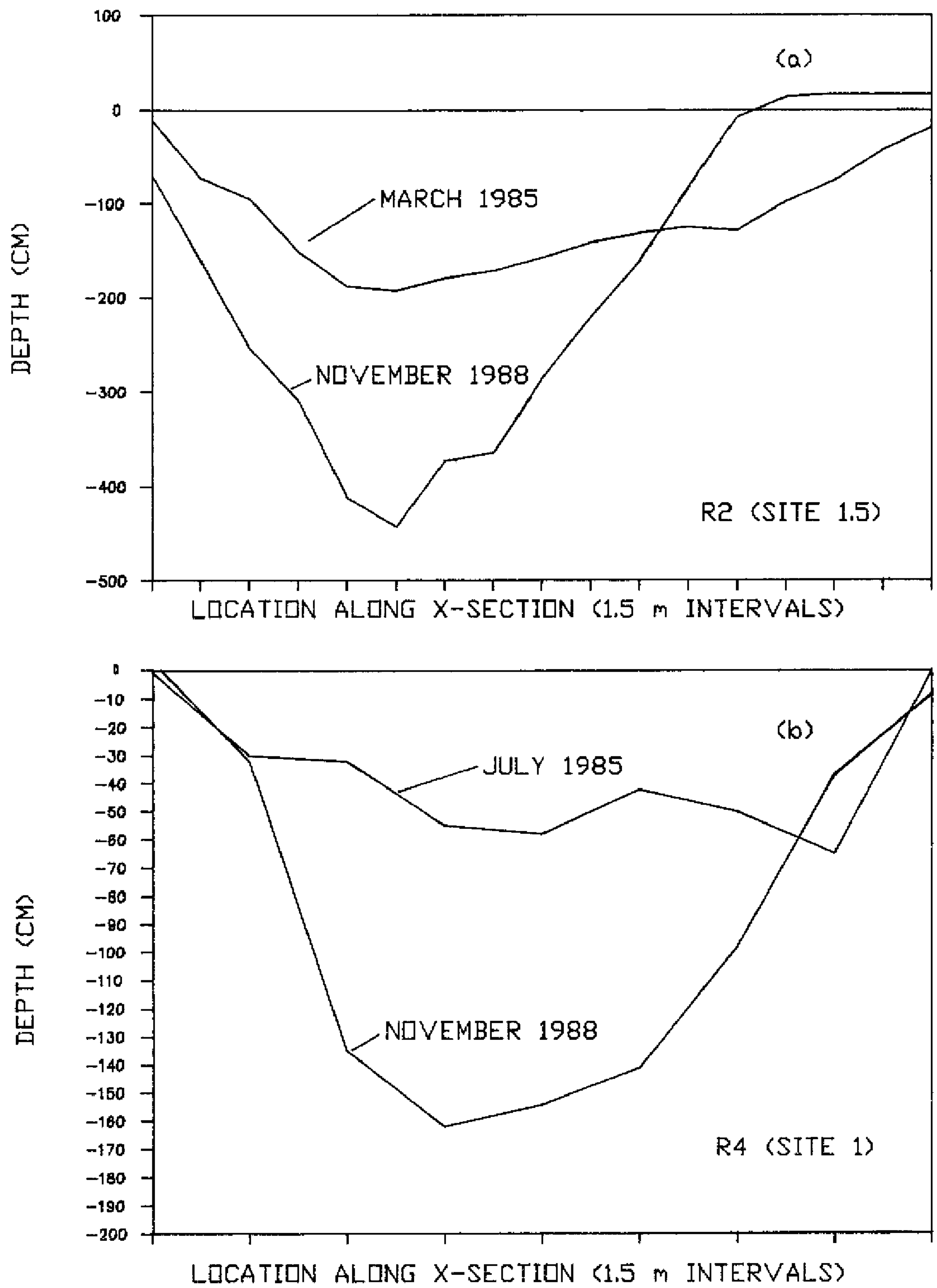

FIGURE 27. Change in river bottom profiles following reestablishment of flow.

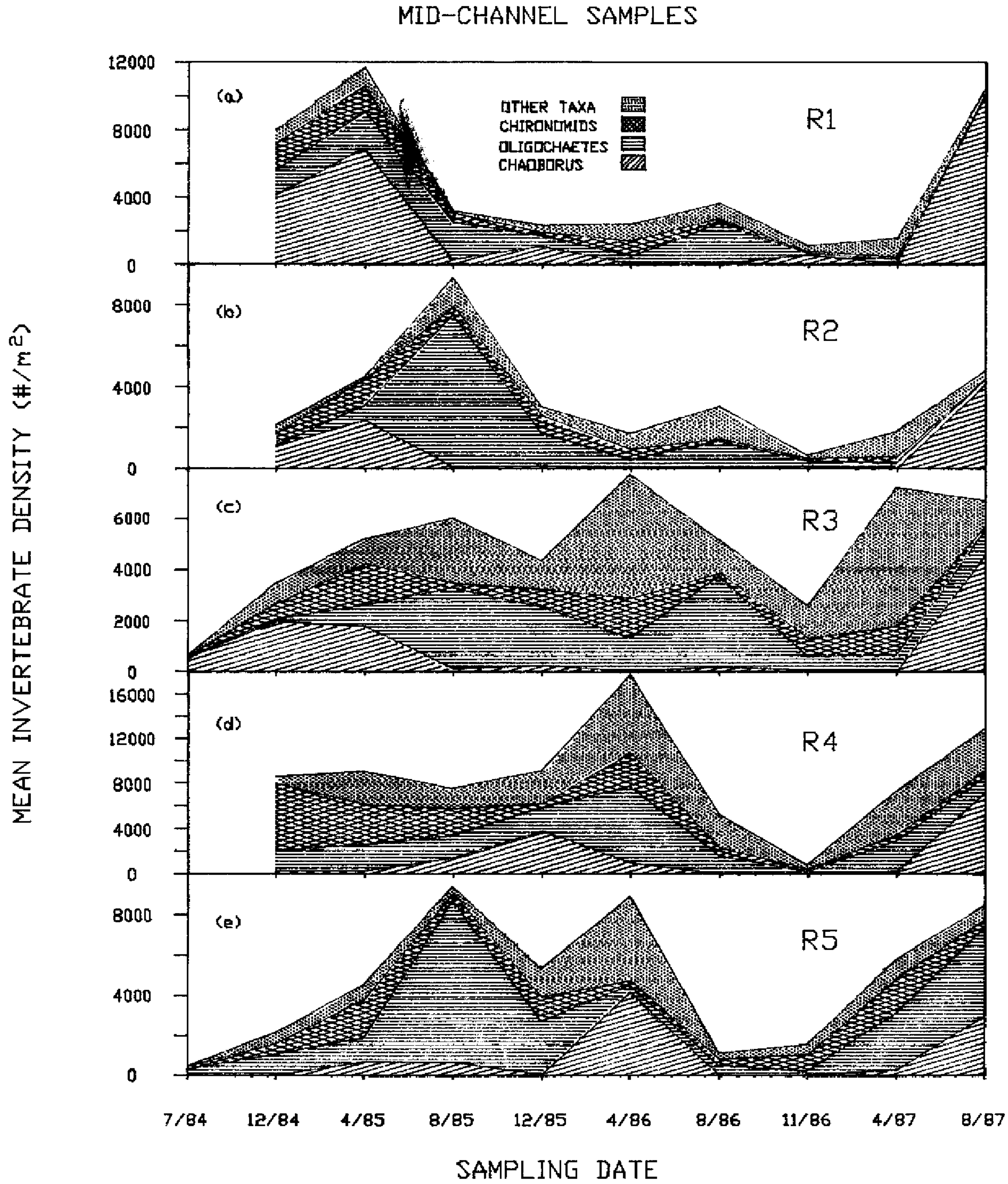

FIGURE 28. Benthic invertebrate densities in mid-channel samples from river runs adjacent weirs (R1-3), downstream of Pine Island Slough (R4), and in Pine Island Slough tributary (R5). A detailed table showing taxonomic composition and densities of mid-channel invertebrate samples is given in Appendix A.

date from December 1984 - December 1985. Chironomidae larvae were the dominant invertebrate group during December 1984; however, during the next three sampling periods, chironomid densities steadily declined, and oligochaetes, ostracods, copepods, nematodes and/or Chaoborus larvae were either equally or more abundant than chironomids. The highest average mid-channel invertebrate density found during the study (17,690 invertebrates/m^2) was recorded at R4 sites in April 1986. During this sampling period, densities of oligochaetes, ostracods and chironomids increased substantially. Subsequently, from April - November 1986, invertebrate densities declined to very low levels (783 invertebrates/m^2 in November 1986), but increased during the last two sampling dates. Ostracods and oligochaetes were the most abundant invertebrate groups in April 1987, and Chaoborus larvae were dominant in August 1987.

The pattern of variation of total invertebrate abundance in R5 (Pine Island tributary) (Figure 28e) was very similar to that displayed by R3 and R4 sites, but taxonomic composition was somewhat different. Total densities in R5 tracked changes in abundance of oligochaetes from July - December 1984, April - December 1985, and November 1986 - April 1987. Chaoborus and Cladocera dominated the community in April 1986 and Chaoborus and oligochaetes were the most abundant taxa in August 1987. Densities were low (< 2,150 invertebrates/m^2) in July and December 1984, and August and November 1986.

Among less abundant taxa collected in mid-channel samples, there was one potentially significant change detected during the monitoring period. At least three species of Sphaeriacean clams, Corbicula fluminea, Musculium transversum and Pisidium casertanum, were found in R5 samples during every sampling date except July 1984, but first appeared as consistent components of the benthic community in runs adjacent weirs in December 1985 (Table 20). In subsequent samples through April 1987, the abundance and frequency of occurrence of these clams increased in samples from runs adjacent weirs, particularly in R2 and R3.

TABLE 20. Densities (#/m^2) and frequency of occurrence (samples with clams/# of samples taken) of Sphaeriacean clams in mid-channel samples from R1-5.

Sampling	Density					Frequency				
Date	R1	R2	R3	R4	R5	R1	R2	R3	R4	R5
7/84	-	-	0	-	0	-	-	0	-	0
12/84	0	0	0	0	470	0	0	0	0	6/8
4/85	13	0	9	109	257	1/8	0	1/10	4/8	7/8
8/85	0	0	0	17	35	0	0	0	2/8	3/8
12/85	4	13	200	57	383	1/8	2/8	3/10	2/8	6/8
4/86	100	144	326	156	222	6/8	6/8	7/10	6/8	5/8
8/86	257	896	343	22	26	7/8	8/8	8/10	2/8	2/8
11/86	91	17	809	13	4	4/8	1/8	9/10	2/8	1/8
4/87	396	891	3035	13	47	8/8	6/8	10/10	2/8	4/8
8/87	109	35	39	0	87	1/8	2/8	3/10	0	2/8

Near-Bank Sites

Compared to mid-channel samples, patterns of variation of invertebrate densities in near-bank samples were more consistent among sampled river runs (Figure 29a-e). From December 1984 - December 1985, for example, total invertebrate densities gradually increased in all runs except R1. During these four sampling dates, oligochaetes and/or chironomid larvae were the only numerically dominant invertebrate groups in R2, R3, and R5, while several other taxa, including Hyalella azteca, Chaoborus, Cladocera, Ancylidae (limpets), and the Ephemeroptera nymph, Caenis diminuta, were as abundant as chironomids and oligochaetes in R4. In contrast to samples from R2-5 (Figure 29b-e), average invertebrate density at R1 sites (Figure 29a) increased dramatically from December 1984 - April 1985, and declined in August and December 1985. During these sampling dates, the near-bank benthic community in R1 was composed primarily of chironomid larvae.

During the next five sampling periods invertebrate densities in all runs typically fluctuated according to the same seasonal pattern that was displayed at R1 sites from December 1984 - December 1985. Between December and April 1986, for example, invertebrate densities increased sharply in all runs except R5, where the average density remained the same during these sampling dates. In R1-3 (Figure 29a-c) this increase was due primarily to increased abundance of Hyalella, ostracods, and nematodes (R1 only), and estimates of total invertebrate densities ranged from 17,000-19,500 invertebrates/m^2. In R4 (Figure 29d), where average invertebrate density was slightly lower (13,000 invertebrates/m^2), ostracods, oligochaetes and chironomids contributed to the increase. Invertebrate densities in all sampled runs decreased in August and December 1986, but peaked again during the following April, when chironomids, oligochaetes, Hyalella, ostracods, cladocerans, nematodes (R4 only), and copepods (R5 only) increased in abundance. On this sampling date, near-bank samples from R1 yielded estimates of almost 22,000 invertebrates/m^2, including 5,300 oligochaetes, 4,600 chironomids, 4,100 cladocerans and 3,200 ostracods. As in 1986, invertebrate densities in R1, R2, and R5 declined between April and August sampling periods in 1987. In R3 and R4, chironomid and oligochaete densities continued to increase in August 1987, and total invertebrate densities from these runs remained the same as in April. On the last sampling date, an average of 3221 Chaoborus/m^2 were collected in samples from R1. This was the only time large numbers of Chaoborus larvae were found in near-bank samples during the monitoring period.

Like mid-channel sites, Sphaeriacean clams also colonized near-bank sites of runs adjacent weirs beginning in December 1985 (Table 21). During subsequent sampling periods, the frequency of occurrence of Sphaeriacean clams increased, and densities as high as 850 clams/m^2 were found in R2 and R3.

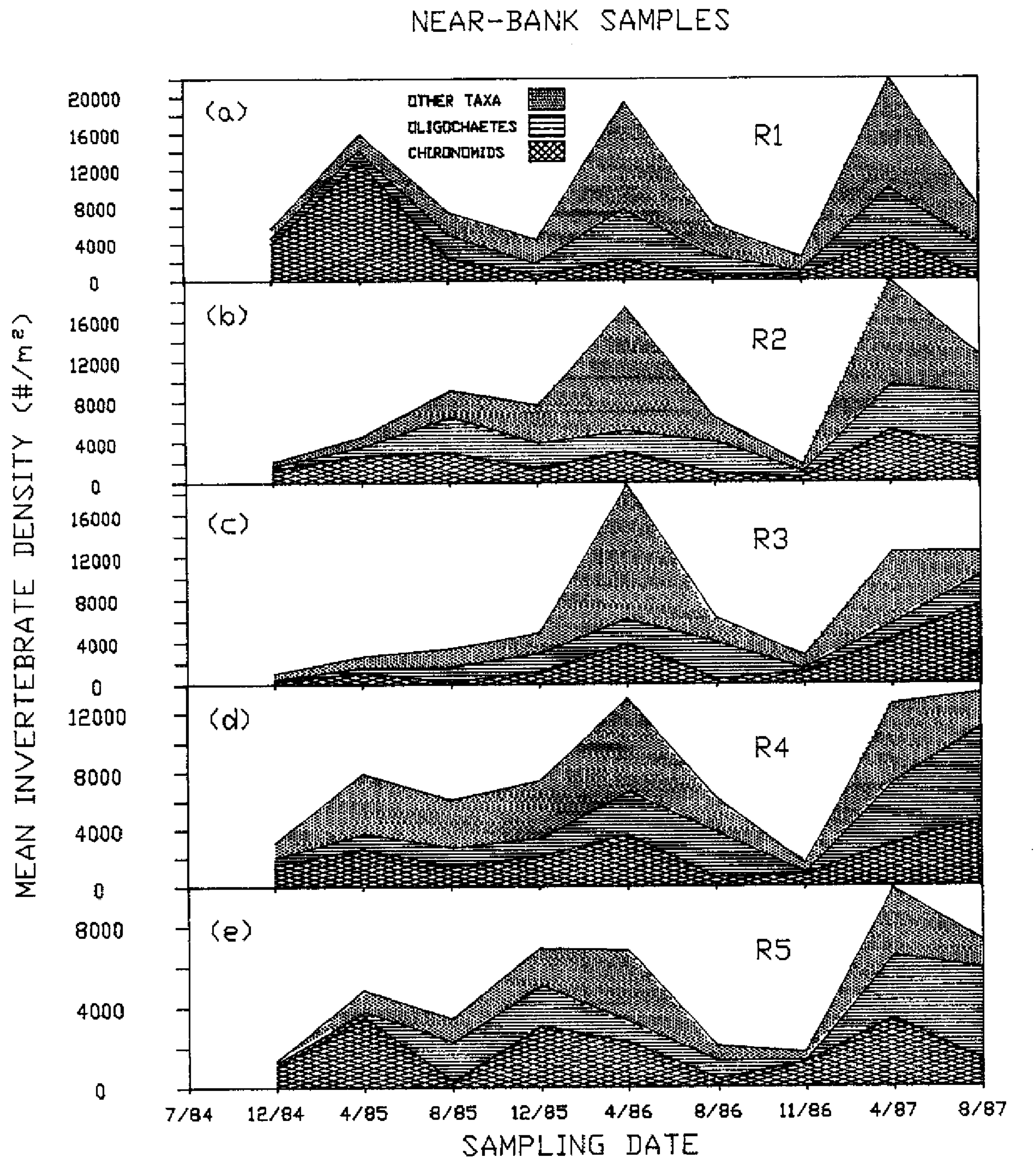

FIGURE 29. Benthic invertebrate densities in samples taken near channel banks of river runs adjacent weirs (R1-3), downstream of Pine Island Slough (R4), and in Pine Island Slough tributary (R5). A detailed table showing taxonomic composition and densities of near-bank invertebrate samples is given in Appendix B.

TABLE 21. Densities (#/m^2) and frequency of occurrence (samples with clams/# of samples taken) of Sphaeriacean clams in near-bank samples from R1-5.

Sampling	Density					Frequency				
Date	R1	R2	R3	R4	R5	R1	R2	R3	R4	R5
7/84	-	-	-	-	-	-	-	-	-	-
12/84	0	0	0	17	26	0	0	0	2/8	2/8
4/85	22	0	0	44	235	2/8	0	0	2/8	5/8
8/85	44	0	9	35	61	1/8	0	1/10	1/8	5/8
12/85	0	13	83	17	491	0	1/8	4/10	1/8	7/8
4/86	296	144	852	417	1017	7/8	4/8	6/10	6/8	7/8
8/86	352	78	178	17	430	6/8	5/8	8/10	3/8	6/8
11/86	26	83	144	0	13	3/8	3/8	3/10	0	1/8
4/87	561	844	617	26	261	6/8	7/8	9/10	3/8	4/8
8/87	17	22	35	0	126	3/8	2/8	6/10	0	3/8

DISCUSSION

HYDROLOGY

Because channelization impacted the Kissimmee River ecosystem largely by altering hydrologic regimes, an effective restoration program must reestablish hydrologic characteristics that formerly shaped the ecological structure and function of the river. Integral features of pre-channelization hydrology included extensive floodplain inundation, slow stage recession rates, and continuous river discharge with frequent overbank flow. Extended hydroperiods and stage recession rates were a product of protracted basin inflows, and slow drainage imparted by dense floodplain vegetation and a low gradient, meandering/braided river channel. Seventy to eighty percent of the floodplain was typically exposed to prolonged inundation, with irregular drying patterns that varied widely in timing, duration and spatial extent. However, higher elevations, particularly along the periphery of the floodplain, frequently were subjected to seasonal wet-dry cycles. Like stage hydrographs, river discharge had a seasonal pattern and a wide range of variability. Overbank flow was common during wet season months and base flow persisted through periodic droughts. A relatively narrow range of flow velocities lended stability and predictability to river channel habitat.

Effects of Demonstration Project components on floodplain hydroperiods varied along the length of the pool and reestablished pre-channelization inundation patterns over only a limited portion of the floodplain. In the lower 20 percent of Pool B, where 70 percent of the floodplain had remained inundated since channelization, controlled pool stage fluctuations reflooded a large portion of drained floodplain for at least brief periods from October 1985 through 1988. The greatest inundation occurred during 1987-88, when deviations from the 11.9 - 12.8 m schedule produced higher stages, and 80-90 percent of the lower Pool B floodplain had inundation frequencies comparable to pre-channelization records. Although pool stage manipulations produced higher average wet season stages and failed to inundate only the peripheral 10 percent of the floodplain at the lower end of the pool, stages did not get high enough to reproduce water depths that typically occurred on most of the floodplain prior to channelization. The pool stage fluctuation schedule also did not result in significantly more drying than that which occurred due to evapotranspiration and a 12.2 m regulation stage (i.e., prior to the Demonstration Project). In addition, because year to year stage variability remained considerably lower than ranges displayed by pre-channelization records, the floodplain was not exposed to the diverse inundation and drying patterns which were once a critical component of ecosystem hydrology.

In the middle 40 percent of the pool, where 80 percent of the floodplain was drained by channelization, the combined influence of controlled stage fluctuations and backwater effects of project weirs resulted in prolonged flooding of only 20 percent of the floodplain, while at least 75 percent of the floodplain was subjected to periodic inundation. These inundation patterns remained drastically different from pre-channelization hydrologic regimes, which consistently flooded most of the floodplain with water depths of 1-3 feet or deeper. Floodplain inundation in this section of the pool was restricted by incongruity between the pool stage fluctuation schedule and floodplain ground elevations. The peak stage of the fluctuation schedule, for example, inundates only 40-45 percent of the middle Pool B floodplain. However, during discharge periods, backwater effects of project weirs produced four, stepped water surface profiles (i.e., between S-65A, the weirs, and S-65B) which increased incrementally from the lower to upper end of the pool. Backwater effects increased floodplain inundation primarily during January - April and September, when regulatory water releases from the upper Kissimmee lakes generated high discharges through the C-38 system. Because backwater effects of weirs reestablished a relationship between stage and discharge, the range of stage variability also increased (i.e., as a function of discharge variability). However, due to operational rules governing discharge from the upper Kissimmee lakes, and the drainage capacity of C-38, stage variability in Pool B remained considerably lower than pre-channelization ranges during all months except March and April.

The influence of backwater effects on floodplain inundation was limited by the rapid rate at which water drained off the floodplain. Rapid stage recession rates were caused by the consummate drainage capacity of C-38, and operational rules that frequently produced abrupt declines in discharge. Thus, while stages were elevated upstream of each weir during discharge periods, particularly when flows were high, inundated floodplain was quickly drained by the canal as soon as discharges ceased. Resultant "spiked" hydrographs contrasted sharply with the gradual rates at which water levels on the floodplain typically receded prior to channelization.

In the northern 40 percent of Pool B, backwater effects of weirs only periodically reflooded about 30-35 percent of the completely drained floodplain. More consistent flooding occurred within the Flow-Through Marsh, where pre-channelization inundation frequencies were restored on approximately 35 percent of the floodplain, and at least 55 percent of the area was inundated seasonally. However, because drainage impediments caused this area to function like an impoundment, hydrologic consequences of the "flow-through" marsh concept could not be evaluated.

Re-introduction of flow through river runs adjacent weirs was a major feature of the Demonstration Project. Resultant river flow regimes were a function of upper basin discharge characteristics and the flow diversion efficiency of weirs. Notched weirs diverted up to 60 percent of C-38 flow through adjacent floodplain and river channels during high discharge periods, but diverted considerably lower proportions of C-38 flow when discharges were $< 28\ m^3/s$ (Loftin et al., 1988). This inefficiency, coupled with upper basin regulation schedules and operation rules, produced river flow regimes that contrasted greatly with key pre-channelization discharge characteristics. Highest discharges, for example, occurred from January - April, during the drawdown phase of the upper lakes schedule, rather than during the wet season months. Extended no flow periods were common from June - December each year, and compared to river discharge records from 1930-60, typical pre-channelization base flows (i.e., discharges exceeding $11\ m^3/s$) were generated through river runs adjacent weirs only half as frequently during the Demonstration Project.

FLOODPLAIN STRUCTURE AND FUNCTION

A primary objective of the Demonstration Project was to determine if hydrologic modifications could be used to rejuvenate remaining floodplain wetlands, and reestablish lost wetlands on drained floodplain. The simplest means of evaluating the efficacy of using hydrology to achieve wetland restoration was to monitor changes in plant species composition.

Plant community responses to Demonstration Project components showed that restoration of wetland communities on the Kissimmee River floodplain is feasible. Pool stage manipulations, backwater effects of weirs, and impoundment of water in the Flow-Through Marsh produced hydrologic changes that affected plant species distributions and community composition. At the lower end of the pool, stage fluctuations affected the permanently inundated wetlands primarily through increased water depths. Key changes in plant community composition included increased growth of Polygonum punctatum and Ludwigia peruviana, which increased the patchiness and semi-woody component of the uniform and predominantly herbaceous, broadleaf marshes. Mitchell (1976) has demonstrated that Polygonum punctatum is capable of adaptive vegetative responses to increased water depths, including vigorous growth and production of floating leaves. At higher elevations in the lower section of the pool, restoration of intermittant or seasonal inundation reestablished a diverse complement of wetland species, including Alternanthera philoxeroides, Hydrochloa caroliniensis, Eleocharis vivipara, Panicum repens, Polygonum punctatum, Panicum hemitomon, Ludwigia repens and Bacopa monneri. In the brush zone in the middle portion of the pool, the combined influence of pool stage fluctuations and backwater effects of weirs led to increased dominance by vines (Ampelopsis arborea, Ipomoea alba, and Sarcostemma clausa) and select hydrophytic species (Alternanthera philoxeroides, Panicum hemitomon, and Salix caroliniana), a closed canopy, lower diversity (species richness) in the understory, and largely eliminated the shrub, Sambucus canadensis. Sambucus occurs in habitats ranging from moist, mesic forests to bay swamps (Clewell et al., 1982), but apparently is less suited for wet conditions than Salix caroliniana (Patton and Judd, 1988). Though not documented quantitatively, Salix also replaced Myrica cerifera as the dominant riparian species along river banks in the middle portion of the pool. A threatened species, the climbing dayflower, Commelina gigas, also appeared to benefit from increased inundation frequencies in brush habitats. In the Flow-Through Marsh, prolonged annual hydroperiods led to replacement of a mesophytic pasture community by a wetland community dominated Panicum hemitomon, Alternanthera philoxeroides, Diodea virginiana, Juncus effusus, Eleocharis vivipara and Paspalum acuminatum.

Although pool stage fluctuations and backwater effects of weirs clearly improved conditions for hydrophytic species on about 60 percent of the Pool B floodplain, some mesophytic and xerophytic species such as Ambrosia artemisiifolia, Eupatorium capillifolium, Myrica cerifera, Sesbania punicea, and Urena lobata persisted in areas subject to only periodic or seasonal inundation. These "weedy" species commonly invade disturbed wetland habitats representing a wide range of edaphic (e.g., soil moisture) conditions (Clewell et al., 1982; Lowe, E.F., 1986; Marshall et al., 1985; Patton and Judd, 1988). Weed species typically are capable of rapid colonization and growth, and may present formidable competition for reestablishment of hydrophytic species (Baird, 1989). Thus, where drained floodplain was subject to only periodic reflooding, restoration of the structural integrity of floodplain plant communities (i.e., species composition representative of hydrologic regimes with periodic inundation) was not completely successful. Monitoring data from the Flow-Through Marsh indicate that a 1-2 year period of constant inundation may be the most effective hydrologic regime for rapid reestablishment of a wetland plant community on drained floodplain. Although mesophytic and xerophytic species were not eliminated entirely (their seed bank appeared to remain viable and individuals colonized during dry periods), prolonged annual hydroperiods (approximately 250 days) shifted the competitive environment in favor of hydrophytic species.

Monitoring data indicate that significant changes in plant community composition can be affected on both drained and inundated floodplain in just one year, and occur in response to subtle, as well as, major changes in hydrology. Plant species responded to several hydrologic factors, including water depths, inundation frequencies, and temporal inundation patterns. In general, Demonstration Project-related modifications of these hydrologic factors led to expanded distributions by hydrophytic species, particularly Alternanthera philoxeroides, Panicum hemitomon, and Polygonum punctatum, and decreased frequencies of occurrence of mesophytic and xerophytic taxa. These results indicate that many of the remaining complement of species on the channelized floodplain are sensitive to hydrologic change, and have the reproductive potential, including a viable seed bank, to rapidly colonize and expand their distribution into habitats with favorable hydrology. Wienhold and van der Valk (1989) have demonstrated that both vestigial seed banks and surviving populations of wetland species can have a significant role in restoration of drained wetlands.

Hydrologic modifications also may have affected plant species distributions indirectly, by altering the competitive structure and dynamics of floodplain communities. In brush habitats, increased inundation frequencies enhanced canopy growth of Salix caroliniana, causing additional shading, which likely contributed to decreased frequencies of understory species. A dense Salix canopy has influenced plant community structure similarly in other floodplain wetlands (Hall and Williams, 1984; Muzika et al., 1987). Effects of increased inundation on growth (density) of Panicum hemitomon appeared to have had a similar impact on low-growing plant species (e.g., Centella asiatica, Hydrochloa caroliniensis, Hydrocotyle sp., and Ludwigia repens) in Pine Island Slough. Lowe (1986) found that a diverse wet prairie community graded into a pure Panicum hemitomon stand as inundation frequencies increased along a lakeshore elevation gradient. In addition to gaining an advantage in competitive interactions, hydrophytic species benefit when hydrologic conditions affect overall competitive regimes by eliminating mesophytic species and associated competition for resources. Elimination of Sambucus canadensis, for example, influenced the competitive structure of brush communities in the middle portion of the pool.

While observed floodplain vegetation changes are indicative of a trend toward reestablishment of wetland plant communities, species composition of sampled communities at the end of monitoring period may not represent a stable community structure. The plant community of the Flow-Through Marsh, for example, clearly has not reached a "climax" state. Because of the inundation characteristics of this impoundment, increased dominance by typical broadleaf marsh species, such as Pontederia cordata and Sagittaria lancifolia, is imminent, and was evident in post-monitoring observations of this area during 1989 and 1990. The long-term stability of other observed plant community changes on the Pool B floodplain will be dependent primarily on characteristics of future hydrologic regimes relative to floodplain hydrology during the Demonstration Project monitoring period. On floodplain dominated by wetland species, such as the lower end of the pool, it is likely that further development of spatial heterogeneity of wetland communities will be induced through continuation of the Demonstration Project water level fluctuation schedule. On drained floodplain subjected to periodic inundation, longer hydroperiods could allow hydrophytic species to eventually outcompete mesophytic species. However, stable plant community structure should not be a restoration goal. A restored floodplain requires hydrologic characteristics that reestablishes and maintains a mosaic of wetland plant communities, with distributions of individual patches of wetland community types in a dynamic state of flux in both time and space (Pickett and White, 1985). In the pre-channelization ecosystem, these wetland community diversity characteristics were maintained by highly stochastic and widely varying stage and discharge regimes. Similar (i.e., unpredictable) hydrologic regimes have been shown to maximize diversity in the Okefenokee Swamp (Greening and Gerritsen, 1987).

While reestablishment of hydrophytic vegetation is a simple indicator of wetland restoration, reestablishment of the functional integrity of the floodplain is a more remunerative goal. Prior to channelization, floodplain wetlands provided breeding or feeding habitat for wading birds, waterfowl, bald eagles, and game fish species, maintained riverine water quality, and served as a factory for production of forage organisms that formed the base of river and floodplain food chains (Dineen et al., 1974; Perrin et al., 1982). Due to the limited scope of the Demonstration Project, complete restoration of these and other potential functional values of the floodplain was not expected. However, both qualitative (numerous field observations) and quantitative monitoring data indicate that the functional integrity of the floodplain wetland system can be recovered.

Wading bird and waterfowl sightings (Toland, 1988; personal observations) provided the most visible qualitative evidence of the potential for restoration of floodplain functionality. Flocks of up to 500 white ibis (Eudocimus albus) and 100 glossy ibis (Plegadis falcinellus), as well as, several wood storks (Mycteria americana) were observed on the Pool B floodplain during high stages. Moreover, during winter 1988, white ibis established a rookery in the willow thicket surrounding the Brush Transects. Willow thickets are important feeding and roosting habitats for many herons, egrets and other wading bird species (Bent, 1963). Waterfowl were less numerous, but Florida ducks (Anas fulvigula fulvigula) and small flocks of ring-necked ducks (Aythya collaris) commonly were sighted on inundated portions of the floodplain. Waterfowl may have benefited from reestablishment and spread of Polygonum punctatum throughout the Pool B floodplain. Polygonum is an important food item of many waterfowl species (Beckwith and Hosford, 1957; Stieglitz, 1972; Landers et al., 1976).

There also was evidence of the role of the floodplain in maintaining river water quality. By filtering sediments and associated nutrient loads from overbank discharge, floodplain wetlands can have a major influence on river water quality characteristics (Boto and Patrick, 1978). In the Kissimmee, reestablishment of this floodplain function may be of considerable significance during high discharge periods when potentially high volumes of sediment and nutrients are transported by the river. Grab samples (water column) taken from the river channel during the high discharge test in February 1988 revealed suspended solids concentrations as high as 41 mg/l, with associated total phosphorus levels of 0.131 mg/l; samples taken at a location where water was draining back into the river from the floodplain had suspended solids concentrations $<$ 1.0 mg/l and total phosphorus levels of 0.042 mg/l. Following the discharge test, thick deposits of organic sediment were observed on the floodplain along the Brush Transects.

Floodplain fish community responses to pool stage manipulations were similar to typical annual population fluctuations of Everglades fishes (Kushlan, 1980). In both marshes, maximum fish densities were found when water levels (depths) peaked at the end of the wet season. However, unlike fish populations in the Everglades (Kushlan, 1974), fish did not become concentrated in the Kissimmee floodplain marsh as water levels receded. During slow drawdown periods, fish utilizing the Kissimmee floodplain may emigrate into adjoining deep-water river habitat, a refuge not available to fish in Everglades marshes.

The observed relationship between water depths and densities of floodplain fishes, particularly Gambusia affinis, may represent a response to increased habitat space during high pool stages. Open water habitat for fish appears to be extremely limited in the densely vegetated broadleaf marsh that has been maintained by stabilized water levels in the lower end of the Pool B impoundment. Increased pool stages may allow resident fish populations to expand into open water habitat space which is created above marsh vegetation. Ovoviviparous species such as Gambusia have the reproductive potential to rapidly colonize this newly created aquatic habitat.

While higher pool stages appeared to lead to expansion of resident, forage fish populations in the marsh, there was no evidence of increased utilization of this portion of floodplain by other fish species which are common in the adjacent river channel. Although enclosure traps can produce some sampling bias (Jacobsen and Kushlan, 1987), vegetation density within the sampled marsh, and the size of the enclosures used in this study, should have provided adequate sampling methodology for capture of at least some representatives of most fish species present. It appears likely that water levels in the marsh did not get deep enough, or were not deep long enough, to accomodate immigration of typical, riverine fish species, particularly large predatory species. Kushlan (1976) found that density and diversity of large carnivorous species, including largemouth bass (Micropterus salmoides), sunfish (Lepomis spp.), and yellow bullhead (Ictalurus natalis), were significant in Everglades marshes only during extended high water periods. Fish utilization of the Pool B floodplain also may have been limited by chronic low dissolved oxygen levels in the marsh, which were not improved by pool stage manipulations. Prior to channelization, fish immigration onto the floodplain probably was tied to, perhaps stimulated by, annual wet season flooding, which flushed deoxygenated water out of the marsh much like wet season pulses of water rejuvenate the Sudd swamps of the African Nile (Howell et al., 1988). Simple manipulations (rise) of water levels in a stagnant impoundment does not reproduce the ecological functionality of flood pulses over the floodplain landscape.

Aquatic invertebrates form the trophic link between wetland vegetation and higher level consumers like fish (Lamberti and Resh, 1985; McIvor and Odum, 1988; Welcomme, 1979), waterfowl (Dubowy, 1988; Murkin and Kadlec, 1986, Wheeler and March, 1979) and wading birds (Hafner et al., 1986; Kushlan and Kushlan, 1975). This link may be direct (via consumption of marsh invertebrates by small forage fish like Gambusia) or indirect (as in consumption of floodplain forage fish by wading birds), and may occur through predation on the floodplain, or following export to adjoining river channels. Export of invertebrates from marshes has not been studied extensively, but can be an important source of colonists for connected streams and rivers (McKillop and Harrison, 1982). Export rates may be most significant during drying periods when water drains off the floodplain. Drift samples during the Demonstration Project revealed that small rivulets draining Pine Island Slough carried up to 4800 small fish and invertebrates per hour to an adjacent river tributary (Toth, 1988).

At a broad taxonomic scale, aquatic invertebrate assemblages in sampled floodplain habitats were similar. However, isopods and gammarid amphipods were a dominant component of the invertebrate fauna along the Marsh Transect, but were not abundant in samples from re-inundated sections of drained floodplain. Amphipods and isopods can be dominant invertebrate groups in wetlands (Martien and Benke, 1977), and are likely the principals[5] "shredders" on the Kissimmee River floodplain. Invertebrate shredders are adapted for ingesting dead plant litter and play a key functional role in the trophic ecology of aquatic ecosystems. Through their feeding

5 A limited number of chironomid species may perform this trophic function (Merritt and Cummins, 1978). Because chironomids were not identified to the species level, the importance of chironomid shredders could not be assessed.

activity (shredding and excretion of feces), coarse detritus is fractionated to fine particulate organic matter and thereby converted to an energy source which is usable by other invertebrate groups in the food web (Cummins, 1974; Cummins and Klug, 1979; Cummins et al., 1989). High densities of isopods and amphipods were accommodated along the Marsh Transect by a large accumulation of detritus, which is typically found in broadleaf marshes, particularly with continuous inundation. Voigts (1976) found that isopods were most common in emergent prairie marshes with abundant dead vegetation. Although Hyalella azteca, a ubiquitous amphipod species (Pennak, 1978), was abundant in other inundated sections of the floodplain, this species can be predominantly a deposit feeder (Hargrave, 1970). Several factors may have contributed to low abundances of isopod and gammarid shredders in re-inundated sections of the Pool B floodplain: (1) the habitat may not have been inundated long enough for significant colonization by these taxa, (2) a source of colonists may not have been available nearby, or had a suitable hydraulic pathway (connection) for colonization, (3) accumulations of detrital material may have been too sparse to support large shredder populations, (4) available litter may have been unpalatable or undigestable (detritus of plant species that have invaded the drained floodplain was not a food source for shredders during their evolution), and/or (5) available litter may not have been conditioned (colonized by microorganisms) sufficiently.

Paucity of isopods and gammarid amphipods on newly inundated floodplain habitats could have several trophic and ecosystem consequences. An incomplete complement of shredders may limit breakdown of dead plant litter (Barnes et al., 1986) and prevent transfer of this source of energy to other components of the floodplain food web (Short and Maslin, 1977). Unless alternative energy sources or pathways are utilized, productivity of another major group of invertebrate consumers, "collectors", may be affected (Cummins et al., 1973; Barnes et al., 1986). During the Demonstration Project, densities of the primary collectors - chironomids, copepods and cladocerans - were comparable among sampled floodplain habitats, and in some samples, were greater in newly inundated sections of floodplain than in the broadleaf marsh. Maximum sample densities of chironomids, cladocerans and copepods were at least four times lower than reported peak abundances of these groups in a California Potamogeton marsh (Lamberti and Resh, 1984), but there was no evidence that productivity of invertebrate collectors on re-inundated sections of the Pool B floodplain was limited by a depauperate shredder fauna. However, low abundances of isopods and gammarid amphipods could have affected fish populations in these reflooded habitats. Although chironomids, cladocerans and copepods can be important food items of small fishes (Fleming and Schooley, 1984; Hynes, 1970), McIvor and Odum (1988) found higher densities of microcarnivorous fishes in marsh habitats with high relative abundances of amphipods than in areas that were dominated by chironomids.

Demonstration project sampling revealed several other characteristics of floodplain invertebrate populations. Broadleaf marsh samples indicated that densities of invertebrates, especially chironomids and microcrustaceans, undergo a consistent seasonal increase between fall and spring, and are higher when the marsh is continuously inundated during this period. Peak invertebrate densities in spring also occur in littoral marshes of lakes (Smock et al., 1981), and can be timely for many waterfowl and river fish species that produce young during this period (Teels et al., 1976; Krumholz, 1980). Invertebrate colonization patterns indicated colonization of re-inundated portions of drained floodplain, as well as, existing marsh habitat was rapid. Oligochaetes were initially a dominant component of newly reflooded habitats, but a representative community structure typically was attained after about 40 days of inundation. Highest densities of invertebrates were found in reflooded areas that were in close proximity to, or hydraulically connected with, aquatic habitats. This included the Drained Floodplain Transect, a periodically reflooded, isolated section of floodplain, which was surrounded by broadleaf marsh, and the Brush Transects, where colonization was facilitated by overbank flows from the river. The importance of overbank flows was illustrated by higher chironomid densities in brush habitats that were exposed to direct overbank flow (Brush Transect B) than at sites that were simply in the path of return flow (drainage) from the floodplain to canal (Brush Transect A). Moreover, because total invertebrate densities along both of these transects declined when overland flow ceased, it appears that floodplain habitats with overland flow may sustain higher invertebrate densities than habitats without flow (e.g., impounded floodplain wetlands).

RIVER HABITAT

Physical habitat attributes such as sorting of substrates, longitudinal and cross-sectional depth profiles, and spatial and temporal velocity distributions are primary determinants of the structure of river biological communities (Allan, 1975; Gorman and Karr, 1978; Harman, 1972; Meffe and Sheldon, 1988). Anthropogenic alteration of these habitat components has led to degradation of biological resources in streams and rivers nationwide (Karr and Schlosser, 1978; Karr et al., 1985); channelization of the Kissimmee River has not been an exception (Perrin et al., 1982, Toth, 1988). The Demonstration Project showed that reintroduction of flow and associated fluvial processes can enhance diversity and quality of degraded river habitat by restoring natural substrate characteristics and channel morphology. A predominately sand substrate was restored through gradual flushing and/or covering of bottom organic deposits without any detectable impacts on water quality characteristics, including turbidity, and nitrogen and phosphorus loads (Rutter et al., 1989). Thus, concerns regarding potential downstream impacts of restoration measures that may re-suspend and transport organic deposits which have accumulated in remnant river runs since channelization were largely repudiated. Bar formation and increased cross-sectional and longitudinal depth diversity in runs adjacent weirs indicated reintroduced flows had reinstituted fluvial/meandering processes (Leopold et al., 1964, Heede, 1980), which are necessary to restore and maintain biologically important stream habitat parameters, including appropriate velocity distributions (Nunnally, 1978; Nunnally and Keller, 1979; Karr et al., 1983).

Although these habitat improvements were encouraging, the integrity of river channel habitat was not restored during the Demonstration Project. Due to regulation of the upper Kissimmee basin, flows through remnant river runs were discontinuous, with extended low or no flow periods during summer and fall months. Low flows can be an (if not the most) important limiting factor for stream and river fish populations (Stalnaker, 1981). No-flow periods continued to contribute to low dissolved oxygen levels in remnant river channels and, except for a more uniform vertical (surface to bottom) distribution of dissolved oxygen during high flows, diversion of C-38 discharges did not lead to consistent improvements in dissolved oxygen regimes in river runs adjacent weirs. Either discharges were not high enough, and/or the length of river through which flow was diverted was not long enough, to allow physical processes to aerate water that was diverted from the canal. Monitoring data (Rutter et al., 1989) indicate that dissolved oxygen concentrations in these canal sections, and hydraulically connected river runs, remained extremely low (i.e., < 3.0 mg/l) during summer and fall months.

During high flows, the drainage capacity of the canal produced an unnaturally steep gradient in river runs adjacent weirs. Resultant high velocities[6] accelerated geomorphic processes and led to a premature meander cutoff in the run adjacent Weir #3. Meander cutoffs are not uncommon in the evolutionary history of natural river systems (Leopold et al., 1964), but increased frequencies of cutoffs would reduce limited availability of river habitat even further (i.e., in addition to the 56 km of river channel that was destroyed by excavation of canal and placement of spoil) in the channelized Kissimmee River. High river channel velocities also could impact river biota directly (Sagar, 1986; Scrimgeour and Winterbourn, 1989). Kissimmee River fish species, for example, are not adapted for survival in high flow velocities (Florida Game and Fresh Water Fish Commission, 1957).

RIVER CHANNEL INVERTEBRATES

Two major changes in benthic invertebrate communities were detected during the Demonstration Project: elimination of Chaoborus punctipennis as the dominant mid-channel invertebrate species, and colonization of runs with restored flow by Sphaeriacean clams. Both of these changes appeared to be related to reintroduction of flow through remnant river channels.

Chaoborus larvae commonly are abundant in Florida lakes (Hunt, 1958; Nordlie, 1976; Schramm and Jirka, 1989), and oxbow lakes (Cooper, 1987) and backwater habitats of large rivers (Sheaffer and Nickum, 1986), but typically are not a dominant member of stream or river invertebrate communities (Hynes, 1970). The ability of Chaoborus to exist in stagnant, anaerobic habitats (Hunt, 1958) likely accounts for its dominance in C-38 (Toth, 1986) and remnant Kissimmee River channels, including pre-Demonstration Project Pool B runs. Compared to pre-Demonstration Project samples, densities of Chaoborus larvae were reduced during most sampling periods following reestablishment of flow, and represented only a minor (numerical) component of the benthic fauna in river channels adjacent weirs during this period. Although maintenance (diversion) of at least intermittent flow appeared to limit Chaoborus abundance in these revitalized runs, high densities of Chaoborus reappeared following a three month period of no flow during summer 1987.

The three species of Sphaeriacean clams collected during the monitoring period, Corbicula fluminea, Musculium transversum and Pisidium casertanum, have somewhat ubiquitous distributions, tend to be most productive in stream and river habitats with fine, sandy substrates and low to moderate flow velocities (Hamill et al., 1979; Heard, 1979; Elstad, 1986; Kilgour and Mackie, 1988), and generally are intolerant of chronic, low dissolved oxygen levels (Mackie, 1979; Jonasson, 1984). Although Pisidium casertanum has wide ecological tolerances (Bishop and Hewitt, 1976; Holopainen and Hanski, 1979), including some capability of survival in anaerobic environments (Hornbach, 1985), none of these species sustained sizable populations in the degraded, remnant river runs which occurred in Pool B prior to reintroduction of flow. Sphaeriacean clams attain high densities and are of considerable ecological importance in other river ecosystems. In Pool 19 of the Upper Mississippi River, for example, densities of Musculium transversum may exceed 100,000/m^2, and provide an annual harvest of approximately 5,000,000 kg for diving ducks (Gale, 1969). In fact, throughout the Upper Mississippi River, dense beds of Musculium are the principle component of "hot spots" of secondary productivity,

[6] During the 1988 discharge test, flows of 60 m^3/sec generated average velocities as high as 0.9m/sec in the river channel adjacent Weir #2.

which have been discovered (located) by tracking waterfowl flocks (Sparks, 1990). Although historic benthic invertebrate data is not available, macroscopic examination of sediment cores indicated Corbicula (at least) was common in the river prior to channelization. Re(?)-colonization of river channels adjacent weirs was indicative of reestablishment of suitable substrate characteristics, seasonal improvement in dissolved oxygen regimes, and reintroduction of flow. Flow is most important to Corbicula and Musculium, epifaunal filter-feeders (Cohen et al., 1984; Lopez and Holopainen, 1987; Way, 1989) which rely on at least some flow or turbulence to suspend or supply their food resources. Pisidium casertanum is an infaunal, deposit (Bailey and Mackie, 1986; Bishop and Hewitt, 1976) and/or interstitial suspension feeder (Lopez and Holpainen, 1987), and was the least abundant of three colonizing clam species. Densities of all three species declined during the last sampling period, following three months of no flow.

Demonstration Project sampling also emphasized the importance of near-bank (littoral) habitats. As in other systems (Egglishaw, 1964; Smock et al., 1981), density and diversity of benthic invertebrates were greater in samples from the littoral zone than in the center of channel. Monitoring data suggest that density and diversity of near-bank invertebrate communities were enhanced by reintroduction of flow, particularly during spring months. Peak invertebrate abundance during spring is timely for many river fish species that produce young during this period (Krumholz, 1980).

CONCLUSIONS AND RECOMMENDATIONS

While clearly demonstrating the feasibility of using hydrology to reestablish floodplain wetland communities, plant community monitoring documented structural characteristics of most Kissimmee River floodplain plant communities, and produced quantitative data on species-hydroperiod relationships and key indicators of appropriate floodplain hydrology. The Demonstration Project also provided evidence of the feasibility of restoring several components of floodplain functionality, including waterfowl and wading bird utilization, small fish and invertebrate productivity, and processes that could enhance river water quality. However, because key hydrologic characteristics were not adequately reestablished, most structural and functional aspects of floodplain ecosystem integrity were affected temporarily and/or only partially restored. The Demonstration Project did not restore the Pool B floodplain; it only proved that restoration of ecological integrity of the floodplain is possible.

Similar conclusions are derived from river channel monitoring studies. Values of flow as a component and determinant of river habitat quality, and organizer of the structure of river biological communities, were verified. The Demonstration Project showed that reintroduction of flow can be used to restore river habitat without causing water quality problems. Results also indicated that direct and/or indirect (i.e., through improved substrate and dissolved oxygen regimes) effects of flow can lead to incipient changes in degraded, river channel invertebrate communities, which appeared to reestablish a species composition with at least rudimentary characteristics of a natural (pre-channelization) river invertebrate community. Although reintroduced flows provided precursors of ecological integrity, the inadequacy of current river inflows precluded more complete and meaningful restoration. In fact, prolonged periods of no flow tended to reverse any restoration progress (derived ecological benefits of reintroduced flow). The Demonstration Project also raised doubt (at best) concerning the value of weirs as a restoration tool. River channel monitoring indicated that potential habitat and biological responses were limited, or even diminished, by the inefficiency with which weirs diverted water during low discharges, and the unnaturally high gradients that they created in adjacent river channels during high flows.

These ecological monitoring studies provide rigorous support for the objectives, guidelines and criteria that recently have been used in evaluations of Kissimmee River restoration alternatives (Loftin et al., 1990). Monitoring results confirm that ecological integrity (Karr and Dudley, 1981) - the primary goal of Kissimmee River restoration - can be achieved only with a holistic approach which succeeds in restoring both the form and function of the former ecosystem. This requires reestablishment of pre-channelization hydrologic characteristics on both the river and floodplain, including river channel and floodplain habitat that was physically destroyed by channelization.

Integration of these monitoring results with hydrologic modeling studies (Loftin et al., 1990, Shen et al., 1990) firmly establishes that restoration of the Kissimmee River ecosystem can be accomplished only by backfilling long, continuous reaches of C-38. Thus, the primary recommendation of this study is to implement the Level II Backfilling Plan described in Loftin et al. (1990), and recently endorsed by Governor Martinez and the South Florida Water Management District Governing Board. This plan will restore the ecological integrity of approximately 9000 ha of river/floodplain ecosystem, including 84 contiguous km of river channel.

Kissimmee River restoration would be an unprecedented project of global significance. However, the scope of its value will be determined largely by the quality and rigor of the ecological studies and monitoring program that are conducted in association with the restoration project. These studies will be subject to close scientific scrutiny. Ecological monitoring studies must be the "heart" of this model restoration program.

Restoration monitoring, like the restoration approach, must have an ecosystem perspective. An effective monitoring program should have the following features:

(1) It must provide a thorough understanding of the ecosystem - with and without restoration.

(2) It should show direct cause-effect relationships between restoration measures and ecological responses.

(3) It should include quantifiable biological responses.

(4) It should document changes that are of societal, as well as, scientific importance.

Demonstration Project results refined direction and provided guidelines for evaluating future restoration efforts. Because plant community monitoring clearly showed that, except for a few persistent weedy species, xerophytic taxa will be replaced by wetland species when hydrologic regimes inundate the floodplain, and the focus of Kissimmee River restoration goals is the river/floodplain landscape rather than individual habitats (Loftin et al., 1990), future restoration studies need to track the overall diversity and spatial juxtaposition of restored floodplain wetland communities. Thus, future monitoring data must be collected at scales that provide detailed characteristics of the distribution of major communities types (e.g., brush, broadleaf marsh, and wet prairie), as well as, habitat patchiness within these communities (e.g., clumps of Polygonum and Ludwigia peruviana in broadleaf marsh). In more diverse floodplain communities, such as seasonally inundated wet prairie (where weedy, mesophytic or xerophytic species tend to persist), restoration evaluation metrics should include measurements of standing crop biomass of key indicator species (e.g., Panicum hemitomon).

Expanded monitoring of the functional utility of the floodplain is warranted because functional characteristics such as wading bird, waterfowl, and fish utilization provide an integrated (i.e., dependent upon an array of structural features) measure of floodplain integrity which is meaningful in both a scientific and general (e.g., understandable by, and of value to society) sense. Monitoring studies that focus on differentiating functional values of representative habitats of the floodplain mosaic, and characteristics that confer these functional attributes, will guide the restoration process.

Studies also should be undertaken to more thoroughly evaluate interactions between the floodplain and river. Contributions (to ecosystem function) of key interactions, such as energy flow (export of invertebrates and dissolved and particulate organic matter) from the floodplain to river, and sediment filtration and nutrient assimilation functions of the floodplain during overbank flow periods, need to be verified to substantiate the success of any river restoration program.

Results of the Demonstration Project also provided direction for future river channel monitoring studies. Identification of invertebrate indicators of both degradation and recovery provides a basis for tailoring at least one component of future restoration monitoring. Results also justify a more thorough study of the littoral zone and its contribution to the trophic ecology of the river. This component would have implications for weed management (control) in the river - a somewhat controversial issue that surfaced several times during the course of the Demonstration Project. Future river channel monitoring should also include comprehensive studies on fish community (i.e., not limited to game fish species) structure and dynamics, and quantitative analyses of river habitat structure.

This comprehensive, ecosystem monitoring program would require at least 2-3 times the level of effort (both manpower and resources) employed during the Demonstration Project. A lesser effort would make it impossible to address all components of the ecosystem, and open monitoring studies to criticism that could discredit the entire restoration program.

The highest immediate priority is to begin to collect baseline data in the section of river and floodplain that will be affected by Part 1 of the Level II Backfilling Plan. This area will include most of Pools A, B and C, and, to achieve the required ecosystem perspective, must involve all components of the ecosystem (e.g., wading birds, waterfowl, fisheries, fish and invertebrate communities, habitat, water quality, and ecosystem function). Two to three years of pre-construction studies, followed by a five-year post-construction evaluation phase are needed.

Two additional measures are recommended. A modified pool stage fluctuation plan, which provides for more complete (i.e., than the Demonstration Project schedule) floodplain inundation using stochastic, rainfall-based stage variability, should be developed. This should be used as an interim (i.e., prior to implementation of Level II Backfilling) measure in Pools A-D, and should be incorporated into the overall restoration plan as a means of reestablishing floodplain wetlands in Pool E. Secondly, to facilitate future restoration efforts, all levees and berms should be degraded, and drainage ditches filled, immediately following purchase (Save Our Rivers) of floodplain parcels.

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Appendix A. Mean invertebrate densities (#/m^2) in mid-channel samples from R-1.

Taxa		1984	1985			1986			1987	
		DEC	APR	AUG	DEC	APR	AUG	NOV	MAY	AUG
Turbellaria, Hirudinea		-	5	-	-	27	82	-	33	-
Oligochaeta		1484	2190	2261	636	467	2484	98	288	109
Nematoda		-	5	49	11	71	22	16	5	16
Cladocera		5	22	-	38	22	5	54	120	-
Copepoda		408	761	11	228	147	-	65	11	71
Ostracoda		380	212	16	65	337	419	16	283	-
Isopoda	_Asellus_	-	-	-	-	-	-	-	-	-
Amphipoda	_Hyalella azteca_	-	5	-	11	71	11	44	272	-
Amphipoda	Gammaridae	-	-	-	-	-	-	-	-	-
Decapoda	_Palaemonetes paludosus_	-	-	-	-	-	-	-	-	-
Hydracarina		5	-	22	22	136	44	11	27	141
Ephemeroptera	_Caenis diminuta_	-	5	-	5	16	27	-	5	-
Collembola		-	-	-	5	5	-	-	-	-
Odonata	Anisoptera	-	5	-	-	-	-	5	-	-
Trichoptera	Hydroptilidae	-	-	-	-	5	-	-	-	-
Trichoptera	Leptoceridae	-	-	-	-	38	-	-	5	-
Trichoptera	Polycentropodidae	-	-	-	-	5	-	-	-	-
Coleoptera	Chrysomelidae (L)	-	5	-	-	-	-	-	-	-
Diptera	Chironomidae (P & L)	1544	1495	587	196	902	239	92	125	5
Diptera	Ceratopogonidae	22	49	-	16	11	22	-	-	-
Diptera	_Chaoborus punctipennis_	4158	6913	261	1098	27	27	576	-	9951
Gastropoda	Ancylidae	-	-	5	-	11	33	-	-	-
Gastropoda	Hydrobiidae	-	-	-	-	-	5	5	-	-
Gastropoda	Planorbidae, Physidae	-	5	-	-	11	22	-	5	-
Gastropoda	Unidentifiable	5	-	-	-	-	-	-	-	-
Pelecypoda	_Corbicula fluminea_	-	-	-	-	-	71	11	266	-
Pelecypoda	Sphaeriidae	-	5	-	5	98	185	82	130	109
Pelecypoda	Unidentifiable	-	5	-	-	-	-	-	-	-

Appendix A. Mean invertebrate densities ($\#/m^2$) in mid-channel samples from R-2.

Taxa		1984	1985			1986			1987	
		DEC	APR	AUG	DEC	APR	AUG	NOV	APR	AUG
Turbellaria, Hirudinea, Nematoda		-	-	60	65	125	169	22	16	38
Oligocheata		60	717	7457	1598	413	1658	239	250	250
Cladocera		5	16	-	16	60	5	16	82	11
Copepoda		261	33	33	16	5	44	11	27	130
Ostracoda		87	11	1130	201	207	342	11	38	125
Amphipoda	_Hyalella azteca_	-	5	22	22	152	54	11	158	-
Hydracarina		-	-	-	5	33	-	-	27	49
Ephemeroptera	Baetidae	-	-	-	-	-	-	5	5	-
Ephemeroptera	_Caenis diminuta_	-	-	11	5	22	-	27	5	5
Ephemeroptera	Unidentifiable	-	-	-	-	-	5	-	-	-
Odonata	Anisoptera	-	-	-	16	-	-	-	-	-
Odonata	Coenagriidae	-	-	-	-	-	5	-	-	-
Trichoptera	Hydroptil., Hydropsych.	-	-	-	-	16	-	-	5	-
Trichoptera	Leptoceridae	-	-	-	5	11	-	-	-	-
Coleoptera	Dytiscidae (A)	11	-	-	-	-	-	-	-	-
Diptera	Chironomidae (P & L)	549	1288	402	842	527	141	103	294	16
Diptera	Ceratopogonidae	65	44	136	49	5	44	27	5	11
Diptera	_Chaoborus punctipennis_	1087	2364	65	136	-	27	130	-	4092
Diptera	Miscellaneous	-	-	-	-	-	-	-	-	-
Gastropoda	Ancylidae	-	-	-	27	-	16	5	-	11
Gastropoda	Planorbidae, Physidae	-	-	11	-	-	5	-	16	-
Gastropoda	Unidentifiable	-	5	11	-	-	-	-	-	-
Pelecypoda	_Corbicula fluminea_	-	-	-	5	54	796	16	783	22
Pelecypoda	Sphaeriidae	-	-	-	5	87	130	-	109	11
Pelecypoda	Unidentifiable	-	-	-	-	-	11	-	-	-

Appendix A. Mean invertebrate densities ($\#/m^2$) in mid-channel samples from R-3.

Taxa		1984	1985			1986			1987	
		DEC	APR	AUG	DEC	APR	AUG	NOV	MAY	AUG
Turbellaria, Hirudinea, Nematoda		-	9	9	161	804	217	130	322	187
Oligocheata		65	830	3200	2291	1352	3574	544	639	1013
Cladocera		39	87	9	57	178	-	30	317	13
Copepoda		420	461	87	52	157	65	122	96	266
Ostracoda		174	296	2257	165	1391	291	44	200	526
Amphipoda	Hyalella azteca	17	22	9	104	1630	4	-	1104	-
Amphipoda	Gammaridae	4	-	-	-	-	-	-	4	-
Decapoda	Palaemonetes paludosus	-	-	-	-	17	-	-	-	-
Hydracarina		-	4	48	13	35	148	4	52	30
Ephemeroptera	Caenis diminuta	4	35	17	48	187	22	17	22	-
Ephemeroptera	Unidentifiable	4	-	-	-	-	-	-	-	-
Collembola		-	4	-	-	-	-	-	-	-
Odonata	Anisoptera	4	-	-	26	13	9	17	-	-
Odonata	Coenagriidae	4	-	9	52	-	-	4	-	-
Trichoptera	Hydroptil., Hydropsych.	-	-	-	-	4	-	4	26	4
Trichoptera	Leptoceridae	-	-	-	-	52	9	17	61	-
Trichoptera	Polycentropodidae	-	-	-	-	-	-	4	22	-
Coleoptera	Larvae and Adults	4	-	9	-	-	4	-	4	-
Diptera	Chironomidae (P & L)	613	1609	170	700	1539	96	700	1135	117
Diptera	Ceratopogonidae	109	44	87	83	39	152	83	183	22
Diptera	Chaoborus punctipennis	1991	1796	104	252	-	196	57	9	4544
Diptera	Miscellaneous	4	-	-	-	9	-	-	-	-
Gastropoda	Ancylidae	-	-	17	109	-	4	-	-	-
Gastropoda	Hydrobiidae, Pilidae	-	-	-	-	-	4	9	4	4
Gastropoda	Planorbidae, Physidae	-	-	-	26	9	-	-	22	9
Gastropoda	Unidentifiable	4	-	4	-	9	-	-	-	-
Pelecypoda	Corbicula fluminea	-	-	-	-	87	57	383	2757	9
Pelecypoda	Unionidae	-	-	-	-	-	4	-	-	-
Pelecypoda	Sphaeriidae	-	-	-	200	239	287	422	278	30
Pelecypoda	Unidentifiable	-	9	-	-	-	-	-	-	-

Appendix A. Mean invertebrate densities (#/m^2) in mid-channel samples from R-4.

Taxa		1984	1985			1986			1987	
		DEC	APR	AUG	DEC	APR	AUG	NOV	MAY	AUG
Turbellaria, Hirudinea, Nematoda		5	125	500	880	1016	783	-	511	266
Oligochaeta		1804	2359	1978	2011	6739	1625	196	2690	1408
Cladocera		98	261	294	207	266	38	-	82	44
Copepoda		70	794	109	239	522	-	-	429	1065
Ostracoda		245	919	228	957	3620	1717	60	2337	2370
<u>Isopoda</u>	<u>Asellus</u>	-	-	-	-	-	-	-	-	-
Amphipoda	<u>Hyalella azteca</u>	-	49	44	22	5	44	11	-	-
Decapoda	<u>Palaemonetes</u>, Astacidae	-	-	-	-	5	-	-	-	-
Hydracarina		5	22	22	-	16	-	-	27	22
Ephemeroptera	<u>Caenis fluminea</u>	5	538	201	239	544	38	54	5	-
Ephemeroptera	Baetidae	-	-	-	-	-	-	-	-	5
Odonata	Coenagriidae	-	5	11	-	11	-	-	-	-
Odonata	Libellulidae	-	-	22	11	5	5	-	-	-
Odonata	Unidentifiable	-	-	-	-	5	-	-	-	-
Hemiptera	Assorted	-	33	-	11	-	-	-	-	5
Trichoptera	Lepto.,Polycen.,Hydrop.	-	16	22	-	38	33	16	-	5
Coleoptera	Larvae and Adults	5	5	22	33	-	60	49	5	5
Diptera	Chironomidae (P & L)	5902	3516	2304	337	2984	679	223	522	554
Diptera	Ceratopogonidae	207	65	114	141	228	212	163	402	27
Diptera	<u>Chaoborus punctipennis</u>	261	261	1440	3837	1011	5	-	223	7136
Diptera	Miscellanous	5	-	-	-	-	-	-	-	-
Gastropoda	Ancylidae	-	-	98	44	22	11	-	-	-
Gastropoda	Planorbidae, Physidae	-	33	98	54	22	11	-	16	-
Gastropoda	Viviparidae, Pilidae	-	-	-	-	-	-	-	-	-
Gastropoda	Unidentifiable	-	-	22	-	-	-	-	-	-
Pelecypoda	<u>Corbicula fluminea</u>	-	-	-	-	-	-	-	11	-
Pelecypoda	Sphaeriidae	-	87	16	54	158	22	11	5	-
Pelecypoda	Unidentifiable	-	22	-	-	-	-	-	-	-

Appendix A. Mean invertebrate densities (#/m^2) in mid-channel samples from R-5.

Taxa		1984	1985			1986			1987	
		DEC	APR	AUG	DEC	APR	AUG	NOV	MAY	AUG
Turbellaria, Hirudinea		<1	-	-	11	22	44	5	16	33
Oligochaeta		1071	1185	8054	2707	337	500	185	2859	4382
Nematoda		-	-	38	87	5	-	-	38	16
Cladocera		-	5	11	-	3065	-	-	120	11
Copepoda		-	337	49	27	544	-	5	467	97
Ostracoda		-	-	5	16	38	38	5	76	254
Isopoda	<u>Asellus</u>	22	-	-	-	-	-	-	11	-
Amphipoda	<u>Hyalella</u> <u>azteca</u>	-	-	11	-	-	-	27	-	-
Amphipoda	Gammaridae	-	-	-	-	-	-	-	16	-
Decapoda	<u>Palaemonetes</u> <u>paludosus</u>	-	-	5	-	-	-	-	-	-
Hydracarina		-	-	-	5	5	-	-	87	80
Ephemeroptera	<u>Caenis</u> <u>diminuta</u>	-	60	11	380	109	-	261	-	-
Odonata	Anisoptera	-	5	33	27	-	-	-	-	-
Trichoptera	Leptoceridae	5	-	16	60	5	11	27	5	22
Trichoptera	Polycentropodidae	-	-	-	-	-	-	-	-	5
Coleoptera	Haliplidae (L)	-	-	22	-	-	-	-	-	-
Coleoptera	Dytiscidae (A)	-	-	-	-	11	-	-	-	-
Coleoptera	Hydrophilidae (L & A)	-	-	-	5	11	11	33	-	16
Coleoptera	Elmidae (L & A)	22	-	-	22	-	22	103	44	1
Lepidoptera	Larvae	-	-	-	-	-	-	-	-	5
Diptera	Chironomidae (P & L)	516	1902	337	1179	255	310	848	1690	383
Diptera	Ceratopogonidae	125	82	87	304	60	135	54	22	38
Diptera	<u>Chaoborus</u> <u>punctipennis</u>	-	701	647	33	4163	-	-	250	2966
Diptera	Miscellaneous	-	-	-	-	-	5	-	-	11
Gastropoda	Ancylidae	-	-	49	33	5	-	-	-	11
Gastropoda	Planorbidae	-	-	-	-	11	-	-	11	11
Gastropoda	Unidentifiable	-	-	-	16	-	-	-	-	-
Pelecypoda	Unionidae	-	11	-	-	-	-	-	-	5
Pelecypoda	<u>Corbicula</u> <u>fluminea</u>	174	5	-	-	-	-	-	-	54
Pelecypoda	Sphaeriidae	5	125	33	380	223	27	5	49	33
Pelecypoda	Unidentifiable	201	125	-	-	-	-	-	-	-

Appendix B. Mean invertebrate densities (#/m^2) in near-bank samples from R-1.

Taxa		1984	1985			1986			1987	
		DEC	APR	AUG	DEC	APR	AUG	NOV	MAY	AUG
Turbellaria, Hirudinea, Nematoda		-	201	266	900	4794	554	234	1309	239
Oligochaeta		696	1235	2614	1203	5391	2304	391	5294	3272
Cladocera		413	109	49	219	250	565	190	4092	114
Copepoda		71	82	-	163	109	87	207	402	16
Ostracoda		179	277	967	22	1467	429	38	3201	174
Isopoda	Asellus	-	5	-	-	-	-	-	5	-
Amphipoda	Hyalella azteca	49	310	33	408	3478	234	130	1087	44
Amphipoda	Gammaridae	-	-	-	-	-	-	-	5	-
Decapoda	Palaemonetes, Astacidae	-	-	-	-	76	5	-	5	-
Hydracarina		5	27	120	22	43	33	-	222	130
Collembola		-	-	-	-	-	-	4	-	-
Ephemeroptera	Caenis diminuta	76	120	71	36	620	82	38	446	11
Ephemeroptera	Baetidae	-	-	-	-	-	-	5	-	-
Odonata	Anisoptera	38	16	109	67	33	87	33	54	38
Odonata	Coenagriidae	11	-	-	18	22	11	22	60	11
Hemiptera	Assorted	-	-	-	22	-	-	54	-	-
Trichoptera	Lepto.,Polyc.,Hydrop.	-	27	-	11	33	-	22	109	11
Coleoptera	Larvae and Adults	11	44	33	44	65	22	16	5	44
Diptera	Chironomidae (P & L)	3967	12962	2419	636	2294	359	500	4636	337
Diptera	Ceratopogonidae	98	49	49	120	261	22	22	65	22
Diptera	Chaoborus punctipennis	38	60	283	172	-	234	424	-	3222
Diptera	Miscellaneous	5	-	-	22	-	11	-	-	11
Gastropoda	Ancylidae, Hydrobiidae	-	16	272	216	98	315	11	83	61
Gastropoda	Planorbidae, Physidae	11	364	22	60	141	234	-	179	38
Gastropoda	Viviparidae, Pilidae	-	-	11	-	11	5	5	5	-
Gastropoda	Unidentifiable	22	60	44	-	-	-	-	49	-
Pelecypoda	Unionidae	5	-	-	-	-	-	11	-	5
Pelecypoda	Corbicula fluminea	-	-	-	-	11	-	-	317	-
Pelecypoda	Sphaeriidae	-	16	44	-	196	353	27	245	16
Pelecypoda	Unidentifiable	-	5	-	-	87	-	-	-	-

Appendix B. Mean invertebrate densities ($\#/m^2$) in near-bank samples from R-2.

Taxa		1984	1985			1986			1987	
		DEC	APR	AUG	DEC	APR	AUG	NOV	MAY	AUG
Turbellaria, Hirudinea, Nematoda		5	22	294	500	1130	408	27	1114	1658
Oligochaeta		342	870	3440	2478	2054	3120	576	4571	5549
Cladocera		49	76	120	500	250	190	5	24130	54
Copepoda		82	5	11	152	359	44	22	359	16
Ostracoda		44	27	1158	380	4190	522	33	2527	261
Isopoda	Asellus	-	-	-	-	-	5	-	-	-
Amphipoda	Hyalella azteca	11	103	33	467	4461	82	44	2016	16
Decapoda	Palaemonetes, Astacidae	-	-	-	-	65	-	11	5	5
Hydracarina		-	65	33	22	-	16	5	109	49
Ephemeroptera	Baetidae	-	-	-	-	11	-	-	5	-
Ephemeroptera	Caenis diminuta	38	158	27	402	821	196	27	413	103
Ephemeroptera	Unidentifiable	-	-	-	11	-	-	-	-	-
Collembola		-	-	-	-	-	5	11	-	-
Odonata	Anisoptera	5	27	94	163	22	71	27	27	44
Odonata	Coenagriidae	-	11	54	44	44	-	5	71	16
Trichoptera	Hydroptil., Hydropsych.	-	5	-	54	44	11	-	114	212
Trichoptera	Leptoceridae	-	-	-	-	44	5	5	16	33
Coleoptera	Larvae and Adults	5	22	44	11	98	16	22	11	5
Diptera	Chironomidae (P & L)	1342	2717	3044	1424	3071	924	457	5065	3092
Diptera	Ceratopogonidae	33	22	179	152	310	114	60	136	163
Diptera	Chaoborus punctipennis	54	33	221	44	-	299	250	11	788
Diptera	Miscellaneous	-	-	-	-	5	-	-	16	11
Gastropoda	Ancylidae	-	-	196	772	141	152	11	33	294
Gastropoda	Planorbidae, Physidae	38	315	141	11	11	92	5	212	92
Gastropoda	Hydrobiidae, Pleuroceridae	-	-	-	-	-	-	22	-	-
Gastropoda	Unidentifiable	22	49	44	-	22	-	-	5	11
Gastropoda	Pilidae, Thiaridae	-	-	-	-	11	-	5	5	-
Pelecypoda	Corbicula fluminea	-	-	-	-	-	27	22	527	-
Pelecypoda	Sphaeriidae	-	-	-	11	141	49	60	315	22
Pelecypoda	Unionidae	-	-	-	-	-	-	-	22	-

Appendix B. Mean invertebrate densities (#/m^2) in near-bank samples from R-3.

Taxa		1984	1985			1986			1987	
		DEC	APR	AUG	DEC	APR	AUG	NOV	MAY	AUG
Oligocheata		204	461	1435	1722	2252	3609	291	1404	2652
Cladocera		117	139	578	61	44	17	9	2052	109
Copepoda		222	91	44	70	17	48	22	157	65
Ostracoda		30	52	226	200	7278	200	113	830	217
Isopoda	_Asellus_	-	-	-	-	9	-	-	-	-
Amphipoda	_Hyalella azteca_	4	235	48	391	2600	370	26	1570	4
Decapoda	_Palaemonetes_. Astacidae	-	26	-	-	44	4	-	4	4
Hydracarina		-	22	44	17	83	39	22	109	135
Collembola		4	-	-	-	-	-	-	-	-
Ephemeroptera	_Caenis diminuta_	-	100	61	191	365	35	100	483	13
Ephemeroptera	Baetidae	-	-	-	-	-	-	22	-	-
Ephemeroptera	Unidentifiable	-	17	-	-	-	-	-	-	-
Odonata	Anisoptera	13	-	83	113	35	52	44	4	174
Odonata	Coenagriidae	-	30	9	26	-	30	-	13	17
Hemiptera	Assorted	9	4	-	-	-	4	-	-	-
Trichoptera	Lepto.,Polyc., Hydrop.,	-	-	-	26	152	35	57	117	204
Trichoptera	Unidentifiable pupae	-	-	-	-	-	-	-	-	13
Coleoptera	Larvae and Adults	9	148	9	57	26	22	17	26	74
Diptera	Ceratopogonidae	9	4	35	78	235	344	313	265	391
Diptera	Chaoboridae	152	9	144	9	-	196	65	9	174
Diptera	Chironomidae (P & L)	309	1200	265	1248	3935	470	1157	4222	7535
Diptera	Miscellaneous	4	9	-	9	9	-	-	-	-
Gastropoda	Ancylidae, Hydrobidiidae	-	4	244	313	44	148	30	13	65
Gastropoda	Planorbidae, Physidae	-	109	100	4	96	65	57	48	96
Gastropoda	Vivi.,Pilidae, Thiaridae	-	-	-	-	-	-	9	9	-
Gastropoda	Unidentifiable	4	-	-	17	-	-	-	-	-
Pelecypoda	Unionidae	-	-	-	-	4	4	-	4	9
Pelecypoda	_Corbicula fluminea_	-	-	-	-	17	17	-	348	30
Pelecypoda	Sphaeriidae	-	-	9	83	835	161	144	270	4

Appendix B. Mean invertebrate densities (#/m^2) in near-bank samples from R-4.

Taxa		1984	1985			1986			1987	
		DEC	APR	AUG	DEC	APR	AUG	NOV	MAY	AUG
Turbellaria, Hirudinea, Nematoda		11	196	304	299	859	815	27	1011	266
Oligochaeta		522	1082	1310	1250	2989	3158	82	4261	6386
Cladocera		294	169	413	625	163	27	-	1429	179
Copepoda		103	190	76	228	136	11	-	359	141
Ostracoda		136	207	152	136	2076	38	-	495	234
Amphipoda	_Hyalella azteca_	130	723	5	169	652	212	44	33	-
Decapoda	_Palaemonetes_, Astacidae	-	-	-	11	-	-	5	-	5
Hydracarina		-	5	11	98	60	5	-	114	185
Ephemeroptera	_Caenis fluminea_	92	1761	141	842	1103	288	103	223	114
Ephemeroptera	Baetidae	11	-	-	92	22	5	-	11	-
Ephemeroptera	Leptophlebidae	-	-	-	-	-	-	-	11	-
Ephemeroptera	Unidentifiable	5	16	-	-	-	-	-	-	-
Odonata	Coenagriidae	5	60	33	33	33	16	27	-	-
Odonata	Libellulidae	5	38	93	136	33	38	54	16	125
Hemiptera	Assorted	5	5	-	49	-	-	-	-	-
Trichoptera	Lepto.,Polyc.,Hydrop.	-	33	33	109	54	54	22	71	245
Lepidoptera	Larvae	-	-	-	11	-	-	-	-	-
Coleoptera	Larvae and Adults	11	60	49	27	33	60	65	16	185
Diptera	Chironomidae (P & L)	1538	2690	1451	2147	3652	625	848	2978	4603
Diptera	Ceratopogonidae	87	196	207	92	603	440	130	1261	386
Diptera	_Chaoborus punctipennis_	103	-	755	315	-	38	169	44	141
Diptera	Miscellaneous	5	-	11	-	-	5	-	-	-
Gastropoda	Ancylidae	5	60	348	554	120	245	5	11	87
Gastropoda	Planorbidae, Physidae	27	353	495	190	11	44	-	5	71
Gastropoda	Unidentifiable	33	82	163	-	-	-	-	-	-
Pelecypoda	Unionidae	-	-	-	-	-	5	-	-	-
Pelecypoda	_Corbicula fluminea_	-	-	-	-	-	-	-	-	-
Pelecypoda	Sphaeriidae	-	5	33	16	386	16	-	22	-
Pelecypoda	Unidentifiable	16	38	-	-	33	33	-	-	-

Appendix B. Mean invertebrate densities (#/m^2) in near-bank samples from R-5.

Taxa		1984 DEC	1985 APR	1985 AUG	1985 DEC	1986 APR	1986 AUG	1986 NOV	1987 MAY	1987 AUG
Turbellaria, Hirudinea, Nematoda		-	33	54	141	65	65	5	342	326
Oligochaeta		141	277	1973	1995	1033	908	92	3082	4516
Cladocera		-	38	5	44	33	16	-	234	54
Copepoda		5	11	44	60	294	-	-	1174	38
Ostracoda		5	11	11	5	82	-	-	130	33
Isopoda	_Asellus_	-	-	-	-	-	-	-	114	-
Amphipoda	_Hyalella azteca_	11	16	33	11	60	60	16	-	-
Amphipoda	Gammaridae	-	-	-	-	-	-	-	179	-
Decapoda	_Palaemonetes_, Astacidae	11	-	-	5	5	-	11	-	11
Hydracarina		-	11	-	5	22	-	-	109	152
Collembola		-	-	-	-	-	-	-	33	5
Ephemeroptera	_Caenis diminuta_	-	71	-	663	332	44	125	60	5
Ephemeroptera	Baetidae	-	-	-	11	-	-	-	5	-
Odonata	Anisoptera	5	11	76	82	33	27	16	-	54
Odonata	Coenagriidae	-	5	-	5	16	-	-	-	27
Hemiptera	Assorted	-	11	5	5	-	-	-	-	-
Trichoptera	Lepto.,Polyc.,Hydrop..	27	22	5	22	27	16	11	16	98
Coleoptera	Larvae and Adults	-	11	65	49	60	16	120	277	27
Diptera	Chironomidae (P & L)	1114	3560	321	3103	2283	413	1255	3457	1353
Diptera	Ceratopogonidae	27	147	60	92	495	27	76	277	109
Diptera	_Chaoborus punctipennis_	-	44	457	-	815	5	-	71	141
Diptera	Miscellaneous	-	-	-	-	-	-	-	-	11
Gastropoda	Ancylidae	-	-	-	136	22	44	-	22	114
Gastropoda	Planorbidae, Physidae	-	386	288	27	27	5	-	16	76
Gastropoda	Unidentifiable	-	-	5	-	109	-	-	-	5
Pelecypoda	Unionidae	-	-	-	-	-	5	-	-	-
Pelecypoda	_Corbicula fluminea_	16	-	11	-	-	-	-	-	5
Pelecypoda	Sphaeriidae	-	103	49	489	1016	429	11	250	120
Pelecypoda	Unidentifiable	11	130	-	-	-	-	-	11	-