Aquatic Fauna Forage Base in the Big Cypress Region

FINAL PROJECT REPORT

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EXECUTIVE SUMMARY

This report provides a summary and analyses of aquatic fauna monitoring conducted in Big Cypress National Preserve (BCNP) cypress forests in the 2006-2011 hydrologic years (5/1/05-4/30/11). These six years of continuous data collection were conducted with the express intent of evaluating the impact of CERP activities on the Big Cypress Region (BCR). We observed significant variation in the amount and timing of dry-season rainfall among study years, resulting in three hydrologic scenarios upon which fauna analyses were based: gradual dry-season recession years (received little dry season rainfall), a reversal year (received significant dry-season rainfall, but still dried prior to the subsequent hydrologic year) and a no recession year (received consistent dry-season rainfall that prevented drying prior to the subsequent hydrologic year).

Only gradual dry-season recession years (WY06, WY07, WY09) facilitated the formation of high density prey concentrations in the late dry season. Structure of prey communities shifted continuously and linearly throughout the year, dominated by crayfish in the first half of the hydrologic year and fish/shrimp in the second half of the hydrologic year. Crayfish biomass peaked in the middle of the hydrologic year, while fish and grass shrimp biomass increased exponentially throughout the year. The dry-season reversal year (WY08) followed similar community structure and standing stock patterns up until an unseasonable February rainfall event, after which community structure resembled wet/dry season transition communities and fauna densities were greatly reduced. No dry-season recession in WY10 caused prey density to remain low and crayfish to remain a significant contributor to prey biomass. When water levels finally receded the following year (WY11), prey concentrations were similar to those seen in gradual recession years.

The sensitivity of BCR cypress habitats to hydrologic variation demonstrated in this study suggests the effects of restoration activities on aquatic fauna community may be considerable. Increased water delivery to the BCNP Addition Lands from L-28-Interceptor and West Feeder canals will increase wetland hydroperiod, and perhaps more importantly, the spatial extent of inundation. While data collected in this six-year monitoring effort allow some ability to predict change, the spatial and temporal variation in data suggest more data are needed to develop robust models for this region. Furthermore, demonstrated differences between BCR and Everglades aquatic fauna communities suggest the application of models of aquatic prey response to hydrology derived for Everglades graminoid marshes are inappropriate.

Finally, we outline the systematic network of sites with varying proximity to the L-28 Interceptor and West Feeder canals identified in this study for the purpose of detecting CERP impacts in northeastern BCNP, and strongly recommend the monitoring effort be reinstated. In the absence of any wading bird monitoring in BCNP (and the difficulty of doing such aerial monitoring in a forested system), aquatic fauna serve as a critical surrogate indicator. With no other ecological monitoring currently conducted within freshwater BCNP, elimination of this monitoring effort has left no means for adaptive management and has disregarded all potential effects of hydrologic change from CERP activities on BCNP cypress forests.

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INTRODUCTION

Forested freshwater wetlands provide critical habitat for fish and macroinvertebrate communities, but they have been poorly studied compared to other wetland ecosystems. These systems are often seasonally inundated, with aquatic fauna moving into the system when it becomes inundated and then seeking deeper-water refuges when waters recess. Timing and delivery of water to freshwater forested wetlands is critical to proper ecosystem functioning. Fish in permanent freshwater systems are often so dependent upon forested floodplains for feeding, spawning, and rearing of young that it has been argued the permanently flooded aquatic systems and their neighboring temporarily- or seasonally-flooded forested habitats should be considered as a single system (Hall and Lambou 1990). Adjacent, temporarily- or seasonally-flooded forests have been found to provide important habitat for fish in large rivers (Kwak 1988; Turner et al. 1994), streams (Ross and Baker 1983), and coastal wetlands (Poulakis et al. 2002). Fish in these forested systems may actually depend on annual water level fluctuation to limit intra- and interspecific competition for food, space, and spawning grounds (Lambou 1959), although the relative importance of habitat structure and hydrologic variability is likely species-specific (Ross and Baker 1983). Because seasonally-flooded forested systems are cut-off from continuously inundated areas as water levels recede, competition for food in these systems may become increasingly important as the dry season progresses. Lambou (1959) found that approximately 44% of fish collected in a Mississippi River floodplain forest were predators. Lambou further concluded that predaceous fish must be partially dependent upon some type of forage other than fish, suggesting that swamp crayfish are likely important in their diet. Studies have also suggested strong aquatic-terrestrial food web links in these systems, as terrestrial invertebrates may provide a significant proportion of fish diet in some cases (Woodall et al. 1975; Arner et al. 1976).

Ecological role of cypress forests in the Greater Everglades system

Although the Greater Everglades ecosystem contains thousands of hectares of freshwater forested wetlands, a paucity of information exists on their aquatic fauna communities. Prior to the start of this study (2005), the few studies that had previously been conducted in South Florida cypress forests focused primarily on fish and spanned relatively small spatial and temporal scales, leaving large-scale spatial variation, inter- and intra-annual variation, and overall faunal community structure virtually undescribed. Several groups in the 1960s and 1970s made fish collections to describe prey populations for wading birds (Kahl 1964; Kushlan 1974; Browder 1976). More recent studies have described the composition and distribution of native and non-indigenous fishes in southern areas of Big Cypress National Preserve (BCNP) (Loftus and Kushlan 1987) and Picayune Strand (formerly Southern Golden Gate Estates) (Addison et al. 2006), produced inventories of fishes found in the different habitats of BCNP (Ellis et al. 2004), and described fish assemblages in isolated cypress wetlands (Main et al. 2007) in Southwest Florida. Limited records of fish collections in the Big Cypress Region (BCR) have also been reported by Loftus (1987, written communication) and Dalrymple (1995). Small quantitative studies describing fish community dynamics in shallow- and deep-water habitats in the Corkscrew region of Big Cypress (Carlson and Duever 1979) and comparing fish communities in natural and restored habitats within the Big Cypress Seminole Indian Reservation (Dunker 2003) have also been reported.

Forested wetlands provide critical nesting and foraging habitat for wading birds in South Florida. In Florida, Wood Storks (*Mycteria americana*) have historically nested primarily in the Everglades, most often in the tops of cypress and mangrove trees. Wood Stork populations declined from 20,000 nesting pairs in the 1930s to 5,000 nesting pairs in the late 1970s, prompting their addition to the federal register of endangered species in 1984 (US Fish & Wildlife Service 1996). Declines in Wood Stork populations have been linked to a decline in small fish that serve as their prey-base, due primarily to loss of wetland habitat and detrimental changes in hydrology (U.S. Fish & Wildlife Service 1996). Wood Storks feed primarily on fish (mosquitofish, sunfish, killifish) but also commonly consume crayfish and other aquatic prey (Kahl 1964). White Ibis (*Eudocimus albus*), identified as a Species of Special Concern by the State of Florida, feed on fish and crayfish and were the most abundant birds observed during 2005 systematic reconnaissance flights (SRF) (Nelson and Metzger 2005). Ibis were observed to use trees in BCNP for roosting. Reports from 2005 SRF flights indicate that the east-central area of BCNP (the L28 gap area) is one of three regions of the Preserve with the highest densities of wading birds, and that the areas with the highest densities were those that tended to be wetter (Nelson and Metzger 2005). Density of wading birds was lower in BCNP than the adjacent Water Conservation Areas (WCAs), also likely due to water levels, period of inundation, and the ability to census nests under dense canopy. Four metrics involving Wood Stork and Ibis populations have been suggested as indicators of Everglades restoration success: (1) timing of nesting by Wood Storks, (2) the ratio of nesting White Ibis + Wood Storks to Great Egrets, (3) the proportion of all nests located in the estuarine/freshwater ecotone, and (4) the interval between years with exceptionally large White Ibis nestlings (Frederick et al. 2009).

Potential impact of CERP activities on the Big Cypress ecosystem

Several CERP projects have the potential to impact cypress forests of BCNP, both in the very near and more distant future. Most immediately, hydrologic restoration of Picayune Strand (Southern Golden Gates Estates) has begun. While hydrological alteration of BCNP is not expected from this project, comparisons with baseline and concurrent data from aquatic communities in un-impacted portions of the region will be very important in assessing the effect of restoration efforts on this ecosystem. Furthermore, the development of robust, quantitative sampling methods for Southwest Florida's forested wetlands is essential to the monitoring efforts that will accompany this restoration project. Several other CERP projects will be beginning in the next few years, and it is imperative that baseline data describing the aquatic faunal community are collected prior to these restoration efforts. The WCA 3 Decompartmentalization and Sheetflow Enhancement project, Seminole Big Cypress Plan, Big Cypress/L-28 Interceptor Modifications and Western Tamiami Trail Culverts will all change hydrology in BCNP (particularly in the northeastern and eastern regions). Furthermore, concerns exist about additional nutrient delivery to the highly-oligotrophic system with the proposed changes in water delivery. The impact of increased nutrient load on the cypress forest aquatic food web is unknown.

Numerous studies in Everglades wetland habitats have found fish and macroinvertebrate communities to be quite sensitive to changes in hydrology and nutrient status. In estuarine forested wetlands, small fish community dynamics (community structure and density) have been directly linked with variation in annual rainfall, freshwater in-flows and salinity (Lorenz and Serafy 2006; Rehage and Robblee 2009) and indicators of roseate spoonbill (*Platalea ajaia*) nesting success (Lorenz et al. 2009). In gramminoid marshes, altered hydrology has been seen to correlate with changes in fish community structure (Chick et al. 2004; Ruetz et al. 2005; Trexler et al. 2005) and aquatic macroinvertebrate abundance (Trexler 2005). Recent studies suggest four aquatic fauna species with different life history responses to drought (bluefin killifish (*Lucania goodei*), flagfish (*Jordanella floridae*), Eastern mosquitofish (*Gambusia holbrooki*) and Everglades crayfish (*Procambarus alleni*)) can be used as indicators for Everglades restoration (Trexler and Goss 2009). Furthermore, interactions between hydroperiod and eutrophication have been linked to changes in fish and macroinvertebrate community structure (Liston 2006) and food web alterations (Williams and Trexler 2006). Because the systems have similar fauna assemblages, it is likely that fish and macroinvertebrate communities in the Big Cypress system will be equally sensitive to hydrologic and nutrient impacts, but the magnitude of these impacts is unknown. It is thought that flooded forests in Big Cypress may be even more sensitive to hydrological changes than Everglades graminoid marshes, due to greater microtopographic variation (Garrett 2007) and greater inter-annual variation in water depths.

Objectives

The objectives of this study were to:

(1) Collect quantitative baseline data on fish and macroinvertebrate communities in BCR cypress forests with the express intent of evaluating the impact of CERP activities on the wading bird prey base;

(2) Describe spatial and temporal variation in BCR aquatic fauna communities and the role of hydrology in influencing these dynamics.

BACKGROUND

Our aquatic fauna monitoring program in the BCR began in July 2005 with an effort to develop field methodology and a statistically valid sampling protocol that would provide quantitative estimates of aquatic fauna standing stock in the forested wetlands of the BCR. Following two years of methods comparisons (WY06 and WY07; methods included 9-m^2 drop trap, 6-m^2 bottomless lift net, 1-m^2 throw trap, experimental gill net and drift fence array), 1-m^2 throw trap sampling emerged as the most effective and practical aquatic fauna sampling method for this habitat. A sampling design was then developed based on statistical power analyses and some considerations of pragmatism (Liston et al. 2007). An additional year of sampling was then conducted using this sampling protocol (WY08) to provide proof-of-concept and the work plan for a long-term monitoring program was finalized (Liston and Lorenz 2008). The refined long-term monitoring effort began in WY09 and was discontinued following WY11. Analyses presented in this final report are based on all six years of continuous data collection at three sentinel sites in BCNP.

As the first aquatic fauna sampling study of its kind in the BCR, data collected in our first three years were largely descriptive. Intra-annual variation in this habitat is dramatic, with wet-season water levels in the center of cypress domes often reaching >1.5 m and frequently drying completely during the dry season. It is not uncommon for dry-season water recession rates to exceed 1 cm/day. The aquatic fauna assemblage and the factors that drive community structure and abundance are virtually identical to those of Everglades graminoid wetlands, but due to differences in microtopography, habitat structure, and other factors, the scale at which some processes operate and the resulting community dynamics appear to be different in many cases. For example, the mosaic of wetland and upland habitats characteristic of the BCR creates a patchy aquatic system (where long-distance dispersal of fish is difficult) much earlier in the dry season than in the graminoid Everglades, where fish follow a continuous drying front throughout much of the dry season before becoming confined in alligator holes and other depressions later in the dry season. Additional research is needed to explore these and other factors in an attempt to better understand the similarities and differences between these two habitats.

There is a noticeable shift in aquatic fauna community structure throughout the hydrologic year in the BCR (Figure 1). The wet-season community is dominated by crayfish (particularly *Procambarus alleni*) and the dry-season community is dominated by grass shrmip (*Palaemonetes paludosus*) and fish. Structure of the macroinvertebrate community appears to change constantly throughout the hydrologic year, while the fish community appears to assemble early in the wet season and remain relatively unchanged through the remainder of the year. Tadpoles are only a small component of overall standing stock and are only common early in the wet season. Crayfish standing stock peaks in the wet season while fish and shrimp standing stock increase exponentially throughout the year, peaking in the dry season prior to desiccation or summer re-inundation. Rainfall (or anthropogenic influxes of water during the dry season) can have a dramatic impact on the aquatic prey available for wading birds at a critical time in their nesting season.

METHODS

Study Design

Monitoring was conducted at three ~1 ha sentinel sites (Table 1) in BCNP that represented a range of forested wetland habitats, including cypress (domes and strands) and mixed swamp forest (Duever et al. 1986). Sites were generally located in the most consistently inundated forested habitats in the areas of the Preserve, and local topography resulted in each site spanning a full range of hydroperiods (from continuously- to occasionally-inundated). Each site was comprised of three replicate study plots. One of the replicate plots at BI was destroyed by lightning strike in summer 2009 and sampling was discontinued due to inaccessibility and dramatic changes in habitat structure.

Our sampling effort at each plot was stratified into sampling zones: long-hydroperiod forest (LHF; generally inundated >10mo/yr), intermediate-hydroperiod forest (IHF; generally inundated ~8 mo/yr), and wet prairie (WP; generally inundated <6 mo/yr) (Figure 2). Canopy in LHF was comprised mainly of pond apple (*Annona glabra*), bald cypress (*Taxodium distichum*) and/or pop ash (*Fraxinus caroliniana*). Canopy in IHF was comprised mainly of bald cypress, but also consisted of red maple (*Acer rubrum*), and/or cabbage palm (*Sabal palmetto*). The only large woody vegetation in WP was bald cypress (height < 3 m; very small specimens were occasionally included in throw trap samples). All plots contained LHF and IHF zones, but only 3 plots contained WP.

Field sampling

During each sampling event, triplicate 1-m^2 throw trap samples (Jordan et al. 1997) were collected in each sampling zone of each plot when present and accessible (5 cm < water depth < 90 cm). Sampling events were generally timed to correspond with the early-wet season (July/August), mid-wet season (September/October), wet/dry season transition (December), mid-dry season (January/February) and late-dry season (March/April). Actual sampling dates, however, were based on annual hydrologic conditions rather than calendar dates. From July 2005 (study start) to April 2011 (end of WY11), 34 sampling events were conducted: 4 in WY06 (no early-wet season throw trap samples were collected due to changes in methodology), 11 in WY07 (extra sampling events were conducted to increase replication for methods comparisons), 6 in WY08 (an extra sampling event was added following the dry-season reversal), 5 in WY09, and 4 in both WY10 and WY11 (sampling was limited due to failure to dry in the first year and an earlier-than-expected dry down in the second year).

During each sampling event, water depth and abundance of aquatic vegetation (counts of emergent species, estimated percent cover of floating and submerged species) were obtained from each trap. Physicochemical data (water temperature, pH, dissolved oxygen (DO; concentration and %saturation) and specific conductance were collected from each sampling zone using a *YSI Model 85 Handheld Dissolved Oxygen, Conductivity, Salinity and*

Temperature System and an *EcoSense® pH10* (YSI Incorporated, Yellow Springs, OH). Water depth was recorded from a permanently installed staff gauge located in the LHF zone at each plot.

Sample processing

All fauna specimens were fixed in 10% buffered formalin in the field and preserved in 70% ethanol after ≥48 hours. In the laboratory, all specimens were identified and enumerated, and standard length (SL; mm)(all other fishes) or carapace length (CL; mm)(*Procambarus*) and wet weight (g)(all specimens) were recorded for each specimen. Sex was recorded for crayfish (CL≥5 mm), livebearers (*Belonesox belizanus*, *G. holbrooki* (SL≥17 mm), *Heterandria formosa* (SL≥10 mm), *Poecilia latipinna* (SL≥18 mm)) and some killifish (*J. floridae* (SL≥20 mm), *L. goodei* (SL≥18 mm)).

Data analysis

Days since dry (DSD) was estimated for each plot for each sampling day to examine patterns of fauna standing stock relative to hydrologic disturbance. We used linear regression to estimate relationships between staff gauge readings at each plot throughout the study and same-day stage data at the nearest hydrologic station (EDEN: http://sofia.usgs.gov/eden/). Regression equations (Table 2) were used to predict daily water depths at each plot and plots were considered 'dry' when daily water depth was ≤5 cm. Annual re-inundation dates for each plot were identified based on predicted daily water depths (Table 3). While early wet season temporary flooding may allow fish to begin dispersing to some degree (especially small, excellent-dispersing species), the re-inundation dates used to calculate DSD were based on the first date water level rose above ground level and stayed above ground level in the deepest part of each plot. Due to low confidence in the ability of regressions to predict daily depths well before the beginning of this study, data collected in the first year of this study with predicted DSD>365 were discarded prior to any correlation with DSD.

A combination of multivariate and univariate techniques was used to examine variation in aquatic fauna communities. Community analyses focused on abundance (no./m^2, ln(y+1) transformed to fulfill assumptions of normality) of common taxa (incidence ≥10%). Temporal and spatial variation in fish and macroinvertebrate community structure was described using 1-way ANOSIM based on a standardized Bray-Curtis dissimilarity matrix (Clarke 1993; Clarke and Warwick 1994). For multivariate analyses, hydroperiod categories were used as the temporal metric: 0-60 dsd, 61-120 dsd, 121-180 dsd, 181-240 dsd, ≥241 dsd. Similarity percentage breakdown (SIMPER) was used to describe observed community variation, and non-metric multidimensional scaling (nMDS) was used to help visualize patterns. Univariate analyses focused on fauna biomass (g wet weight/m^2; ln(y+1) transformed to fulfill assumptions of normality), as this metric has the most direct application to resource availability for higher trophic levels. We used analysis of covariance (ANCOVA) to examine variation in biomass of common fauna species among study years, study sites and with DSD (DSD^2 is included in model to detect/describe an exponential function). All results reported from ANCOVA are based on type III sums-of-squares (Shaw and Mitchell-Olds 1993).

RESULTS

Hydrology

The BCR experienced significant inter-annual hydrologic variation during the 6 years of this study. While total annual rainfall was relatively consistent between study years ($\bar{X}$=54.6 ± 3.7 in at BCA-15), the amount and timing of annual dry-season rainfall created three different hydrologic scenarios, each with distinct implications for concentrating aquatic fauna (Figure 3):

- Gradual recession– Gradual water level recession resulting from little dry season rainfall was observed in the dry seasons of WY06, WY07, WY09 and WY11 (Figure 4).

- Reversal– Significant dry-season rainfall in February 2008 resulted in a notable dry-season reversal in WY08 (Figure 5). Water depth at study sites rose 40-60 cm in one day, but this reversal occurred early enough in the dry season that sites still dried prior to the onset of WY09 rains.

- No recession– Consistent rainfall throughout the WY10 dry season kept water levels near peak levels and prevented sites from drying prior to WY11 (Figure 5).

Additionally, due to local variation in rainfall, one site (BI) experienced a drought and only re-inundated for a few weeks (and only in the deepest parts of the site) in WY08. While water levels at the other two sites were relatively low in WY08, they did not experience a drought of this magnitude.

Physicochemical conditions & habitat structure

Intra-annual variation in physicochemical conditions was observed consistently throughout all years of this study, while intra-site variation was minimal. Water temperature was highest in the summer months (July-September), fell steadily in winter months (October-February), and rose again in the spring prior to desiccation (Figure 6). Specific conductance (Figure 7) and pH (Figure 8) increased throughout the hydrologic year, with specific conductance often significantly higher at BI compared to other sites. While dissolved oxygen levels displayed a high degree of temporal variability, concentration and %saturation were consistently highest at RP and lowest at BI (Figure 9).

Macrophyte community structure and coverage varied consistently among sampling zones, while variation among sites was minimal (Table 4). The WP community was primarily comprised of beaksedge (*Rhynchospora* spp), spikerush (*Eleocharis* spp.) and periphyton (benthic and floating). The IHP community was primarily comprised of lemon bacopa (*Bacopa caroliniana*), creeping primrosewillow (*Ludwigia repens*), and Chapman's arrowhead (*Sagittaria graminea var. chapmanii*). The LHP community was primarily comprised of lemon bacopa, creeping primrosewillow and water spangles (*Salvinia minima*) (site-scale spatial variation in habitat structure is detailed in the attached draft manuscript, Appendix 1).

Aquatic fauna variation with hydrology

In the six years of this study, we collected a total of 24,353 invertebrates (8.9 kg wet weight), 24, 573 fish (12.8 kg wet weight) and 323 amphibians (0.3 kg wet weight) (Table 5). We present analyses of variation in aquatic fauna community structure and density with hydrology, conducted separately for each of the three previously described hydrologic scenarios, with some exception:

- Gradual recession– Fauna analyses include all data from WY06, WY07 and WY09 and data from wet season WY10 (wet season WY10 is included as it followed similar rainfall/inundation patterns to the other years)
- Reversal– Fauna analyses include all data from WY08
- No recession– Fauna analyses include all data from WY10 and WY11 to show a full picture of the extended period of inundation

GRADUAL DRY-SEASON RECESSION. In study years that experienced a gradual dry-season recession, we observed significant spatial and temporal variation. Fauna community structure varied among study sites (Global R=0.215, P=0.001) and with duration of inundation (Global R=0.309, P=0.001). While the aquatic fauna community shifted gradually throughout the hydrologic year, these shifts were remarkably consistent among study sites (Figure 10). Communities were characterized as follows:

- Early wet season (≤60 dsd): juvenile crayfish (*Procambarus* spp.), Everglades crayfish (*Procambarus alleni*), tadpoles, and eastern mosquitofish (Gambusia *holbrooki*) (cumulative similarity=76.95%)
- Mid-wet season (61-120 dsd): eastern mosquitofish, Everglades crayfish, and flagfish (*Jordanella floridae*) (cumulative similarity=78.23%)
- Wet/dry season transition (121-180 dsd): eastern mosquitofish, Everglades crayfish, dragonfly larvae, grass shrimp (*Palaemonetes paludosus*), and slough crayfish (*Procambarus fallax*) (cumulative similarity=88.43%)
- Mid-dry season (181-240 dsd): eastern mosquitofish, grass shrimp, dragonfly larvae, slough crayfish, and least killifish (*Heterandria formosa*) (cumulative similarity=82.36%)

- Late-dry season (≥241 dsd): eastern mosquitofish, grass shrimp, bluefin killifish (*Lucania goodei*), slough crayfish, and flagfish (cumulative similarity=83.84%)

Aquatic fauna standing stock increased throughout the hydrologic year as the community shifted from one dominated by crayfish and early-dispersing fishes to one dominated by later-dispersing fish and grass shrimp (Figure 11). Eastern mosquitofish were ubiquitous and tadpoles were generally present only very early in the hydrologic year. Within the crayfish community, we observed a gradual shift from a community comprised primarily of Everglades crayfish and juveniles (likely, primarily Everglades) to one comprised primarily of slough crayfish (Figure 12).

ANCOVA of aquatic fauna standing stock indicated significant spatial, temporal and hydrologic variation (Table 6). Inter-annual and inter-site variation was observed in biomass of most common taxa, as well as total fish biomass and total crayfish biomass. Despite this variation, variation with hydrology was consistent across study years and sites for almost all common taxa. Exponential increases in biomass with duration of inundation were observed in all common fish species (Eastern mosquitofish (Figure 13), least killifish (Figure 14), flagfish (Figure 15), warmouth (Figure 16), dollar sunfish (Figure 17), bluefin killifish (Figure 18)), total fish (Figure 19) and grass shrimp (Figure 20). We observed hump-shaped relationships between duration of inundation and biomass of Everglades crayfish (Figure 21) and total crayfish (Figure 22; Everglades crayfish often dominate crayfish biomass), and an inverse-hump-shaped relationship between duration of inundation and biomass of juvenile crayfish (Figure 23).

DRY-SEASON REVERSAL. In the dry-season reversal year, we observed significant spatial and temporal variation in the aquatic fauna community. Fauna community structure varied among study sites (Global R=0.369, P=0.001) and with duration of inundation (Global R=0.104, P=0.027). Inter-site differences were driven by a higher relative abundance of flagfish and crayfish (both species) at RP, and a higher relative abundance of eastern mosquitofish and dragonfly larvae at L28. From the early-wet to mid-dry season, the aquatic fauna community shifted similar to that described in years with gradual dry-season recessions (above). Following the reversal, however, the late-dry season community was more similar to the wet/dry season transition community than expected (Figure 24). Intra-annual variation was relatively consistent among study sites, and communities were characterized as follows:

- Early wet season (≤60 dsd): not sampled in WY08
- Mid-wet season (61-120 dsd): eastern mosquitofish, Everglades crayfish, and flagfish (cumulative similarity=86.52%)
- Wet/dry season transition (121-180 dsd): eastern mosquitofish, Everglades crayfish, dragonfly larvae, flagfish, and slough crayfish (cumulative similarity=91.94%)
- Mid-dry season (181-240 dsd): eastern mosquitofish, Everglades crayfish, dragonfly larvae, flagfish, and warmouth (*Lepomis gulosus*) (cumulative similarity=86.56%)

- Late-dry season (≥241 dsd): eastern mosquitofish, Everglades crayfish, grass shrimp, flagfish, slough crayfish, and bluefin killifish (cumulative similarity=79.67%)

Aquatic fauna standing stock increased leading up to the reversal as was observed in other years of this study. Following the reversal, however, standing stock was markedly reduced compared to pre-reversal levels and the high density prey concentrations seen in gradual dry-season recession years were absent (Figure 11). Relative to gradual dry-season recession year, the late-dry season community (post reversal) had fewer fish and grass shrimp, and fish and crayfish contributions to biomass were similar.

ANCOVA of aquatic fauna standing stock indicated significant variation with hydrology (spatial variation was minimal) in a few species (Table 7). We observed hump-shaped relationships between duration of inundation and biomass of eastern mosquitofish, dollar sunfish, total fish, slough crayfish and total crayfish (Figures 25 & 26). This variation reflects a gradual increase in standing stock through the wet season and the beginning of the dry season as water levels began to recede, followed by a dramatic decrease in standing stock as water levels rose quickly in response to the unseasonable winter rainfall (the reversal).

NO DRY-SEASON RECESSION. In the two-year period with no dry-season recession, we observed significant spatial and temporal variation in the aquatic fauna community. Fauna community structure varied among study sites (Global R=0.263, P=0.001) and with duration of inundation (Global R=0.355, P=0.001). From the early-wet to mid-dry season of the first year, the aquatic fauna community shifted similarly to that described in years with gradual dry-season recessions (above). As water levels remained steady for the remainder of the first year and into the second year, community structure did not change markedly. Toward the end of the second year when water levels finally began to recede (>480 dsd), further shifts in community structure were apparent (Figure 27). While sampling was not conducted during every 60-d period, 60-d hydroperiod categories were used for consistency. Communities were characterized as follows:

- ≤60 dsd: Everglades crayfish, eastern mosquitofish, juvenile crayfish, and flagfish (cumulative similarity=88.80%)
- 61-120 dsd: eastern mosquitofish, Everglades crayfish, and juvenile crayfish (cumulative similarity=89.98%)
- 121-180 dsd: no sample
- 181-240 dsd: Everglades crayfish, eastern mosquitofish, slough crayfish, and dragonfly larvae (cumulative similarity=79.89%)
- 241-300 dsd: eastern mosquitofish, grass shrimp, bluefin killifish, slough crayfish, dragonfly larvae, and least killifish (cumulative similarity=83.33%)
- 301-360 dsd: no sample
- 361-420 dsd: no sample

- 421-480 dsd: eastern mosquitofish, grass shrimp, least killifish, bluefin killifish, dragonfly larvae, slough crayfish, and brook silversides (*Labidesthes sicculus*) (cumulative similarity=86.19%)
- 481-540 dsd: grass shrimp, eastern mosquitofish, least killifish, bluefin killifish, and dragonfly larvae (cumulative similarity=89.68%)
- ≥541 dsd: eastern mosquitofish, flagfish, juvenile crayfish, grass shrimp, Everglades crayfish, least killifish, slough crayfish, and spotted sunfish (*Lepomis punctatus*) (cumulative similarity=87.05%)

Due to the persistence of high water levels through the first hydrologic year, we did not observe the dry-season fauna concentration seen in years with gradual dry-season recession. Rather, fish and grass shrimp biomass remained relatively low and crayfish remained abundant. Toward the end of the second hydrologic year (>541 dsd), fish and grass shrimp biomass increased markedly with maximum fauna standing stock equivalent to that of high density concentrations seen in years with gradual dry-season recession (Figure 11).

Observations of crayfish response to extended dry season

Reduced rainfall in northwestern BCNP resulted in an extremely limited wet season at BI in WY08. Between March 2007 and July 2008, the deepest parts of BI were inundated a maximum of ~53 days. The maximum predicted daily water depth at our staff gauge during this time was only 29 cm (average depth for this time of year is >70 cm), leaving most of BI dry for the entire 16-month duration. While the ability of small fishes to disperse and endure multi-year droughts has been described for the Everglades, the impact on crayfish is unknown (N. Dorn, personal communication). In WY07 (prior to the extended dry season), average crayfish density at BI was 12.28 crayfish/m^2 in August and 7.11 crayfish/m^2 in October. In WY09 (following the extended dry season) crayfish density was initially extremely low, as average crayfish density at BI was 0.55 crayfish/m^2 in August and 0 crayfish/m^2 in October. We continued to see increases in crayfish standing stock during subsequent sampling events (Figure 11).

Observations of cold stress on non-native fish

Peninsular Florida experienced a severe cold snap in mid-January 2010, but large cold-stress fish kills (like those seen in other parts of the Everglades system) were not observed (personal observation and communication). Recent studies have demonstrated especially poor low-temperature tolerance in non-native fishes (Schofield et al. 2009; Schofield and Huge 2010). While the incidence (23.2%) and abundance ($\bar{X}$= 0.32 ± 0.07 fish/m^2) of non-native fishes was relatively low at study sites prior to this cold event, no non-native fish were collected at study sites in the 5 sampling events (55 throws) conducted following this event (March 2010-February 2011). With limited observations, however, observations are only anecdotal and cause-and-effect relationships cannot be established.

DISCUSSION

Aquatic prey quality, vulnerability and density during Everglades' dry seasons are critical factors for determining to what extent prey populations benefit breeding wading birds (Gawlik 2002). In this study we demonstrated that in forested wetlands of the Big Cypress Region, the wet season is characterized by significant changes in the structure of prey communities (community assembly), while the dry season is characterized by significant increases in prey density (prey concentration). With hydrology as the major driving force in these seasonal wetlands, we revealed an apparent shift in Big Cypress prey communities from a crayfish-dominated standing stock in the wet season to a fish- and grass shrimp-dominated standing stock in the dry season. Data collected in this study also reveal how variation in rainfall patterns (particularly the amount and timing of dry season rainfall) can have a dramatic impact on these communities, and thus, on the density and availability of aquatic prey for foraging wading birds.

Three of the six years of this study displayed a pattern of inundation where all cypress habitats (except the center of the deepest domes and sloughs) dried annually and received little dry-season rainfall (hereby called "typical" years). In these years, Everglades crayfish emerged immediately following inundation, dominating a relatively low fauna standing stock. This is also the only time of year that tadpoles were a significant presence in the fauna community. As the wet season progressed, slough crayfish increased in abundance, matching Everglades crayfish biomass by the wet/dry season transition. This pattern likely results from the drying of wet prairies and shorter-hydroperiod habitats where more drought-tolerant Everglades crayfish are indirectly favored (Dorn and Trexler 2007). Small fish and grass shrimp re-colonized wetlands as they do in Everglades graminoid marshes, following the flooding front as it expanded the spatial area of inundation outward from dry-season refuges (Trexler and Goss 2009; DeAngelis et al. 2010; Jopp et al. 2010). The mosaic of habitats that is characteristic of the BCR places short-hydroperiod wetlands (e.g., wet prairies, hydric pinelands) in relatively close proximity (often <100 m) to long-hydroperiod wetlands (e.g., the center of cypress domes, cypress sloughs). As dry-season surface water begins to recede, short-hydroperiod habitats quickly compartmentalize much of the system (as upland), creating barriers for the long-distance movements of fishes described in Everglades sloughs (Ruetz et al. 2005; Goss 2006; DeAngelis et al. 2010). Rather, once the BCR begins to dry, fishes likely seek dry-season refuges on a more localized scale and those that find the deepest refuges remain there (trapped) through the dry season.

In these typical years, as the dry season progresses aquatic prey concentrate, as has been well-documented in other Everglades wetlands (e.g., Loftus and Kushlan 1987; Loftus and Eklund 1994). As prey become increasingly concentrated, they become more susceptible to predation (from piscivorous fish, wading birds, alligators, mesomammals, etc.) (Ogden et al. 1976; Kushlan 1980; Bondavalli and Ulanowicz 1999; Gawlik 2002) and abiotic factors (e.g., Kushlan 1974; 1979; Kushlan and Hunt 1979; Loftus and Kushlan 1987). Studies in other Everglades habitats have shown this seasonal drying pattern results in a disturbance-limited system (Loftus and Eklund 1994; Trexler et al. 2005) that particularly devastates piscivorous fish (Chick et al. 2004). Our data suggest this is true for the BCR, as well, as recently re-inundated cypress forests were largely fishless.

The dry-season reversal experienced in WY08 resulted from an unseasonable amount of dry season rainfall. In northeastern BCNP, 5 in of February rainfall (recorded at BCA 18) resulted in a water depth increase of >29 cm in long-hydroperiod forests at L28, and wet prairies and intermediate-hydroperiod forests that had previously dried became re-inundated. This increase in water levels, and resulting spatial extent of inundation, allowed fauna to disperse widely. Fish and grass shrimp density decreased markedly, and combined with intermittent rain throughout March and April, the formation of high-density prey patches was prohibited. This dry-season re-inundation also caused Everglades crayfish to emerge from burrows and reproduce. The community succession patterns of macroinvertebrate and fish community structure observed in typical years were not observed, rather, post-reversal communities were quite similar to the communities seen in the early- to mid-wet season following the initial inundation.

The extended wet-season seen WY10-WY11 also failed to support the formation of high-density prey patches in the first hydrologic year. During the extended period of inundation, however, fauna community structure did not change significantly more than during typical wet seasons. Rather, high crayfish standing stock persisted throughout WY10 and into early WY11 as fish standing stock remained low, supporting previously described patterns of crayfish abundance in wetlands with low predation pressure from fishes (Dorn 2008). When water levels finally receded in the WY11 dry season, density of fauna concentrations was similar to that seen in typical years, suggesting maximum dry-season density is not limited by duration of inundation (time needed for species to disperse, etc.).

Several findings of this study emphasize the need for continued monitoring in this region in order to evaluate the impacts of CERP activities in the BCR:

- Spatial variation observed between sentinel sites, and PI observations of the over-dried landscape in NE Big Cypress Addition Lands, highlight the importance of increasing the spatial extent of the monitoring network by adding multiple sites in the relative vicinity of West Feeder and L28-Interceptor canals (see next section for recommendations).
- Marked hydrologic variations in the six years of this monitoring effort (resulting from annual rainfall patterns) emphasize the need for additional study years to establish a robust baseline data set.
- Differences between BCR and Everglades aquatic fauna communities suggest the application of models of aquatic prey response to hydrology derived for Everglades graminoid marshes are inappropriate.

Implications for wading birds

The annual timing of aquatic prey concentrations is of particular significance for wood stork nesting, as their foraging strategy is well-suited for exploiting such concentrations. The timing of wood stork nesting has been identified as a primary indicator of Everglades restoration success (Frederick et al. 2009). Wood storks are asynchronous nesters, historically initiating nesting in the BCR in early November and finishing in June. Foraging demand by wood storks is greatest when adults are feeding nestlings, and particularly, when chicks have reached 3-4 weeks of age (Kahl 1964). During this time (January/February), crayfish still dominated aquatic fauna biomass in most years of this study. Documentation of wood stork consumption of crayfish is sparse, however, storks are known to consume them (Kahl 1964). Difficulty in observing prey captures in forested wetlands and greatly reduced crayfish presence in wet prairies and long-hydroperiod forests documented in this study may contribute to an under-sampling of such behavior in wood storks.

Historically, abundant short-hydroperiod wetlands (primarily hydric pine flatwoods and wet prairies) supported large wood stork colonies in Southwest Florida. The spatial arrangement of these wetlands in a wetland mosaic provided ideal conditions for production of wood stork prey populations (J. Lauritsen, personal communication). A delay in nest initiation in most years since the mid-1980s is likely directly linked to the disproportionate loss of Southwest Florida's short-hydroperiod wetlands which provided foraging opportunity for wood storks during the early portion of the nesting season (J. Lauritsen, personal communication). Productivity in the well-documented wood stork colony nesting at Corkscrew Swamp Sanctuary appears to be higher when nest initiation begins early in the nesting season (November or December) (unpublished data). Interestingly, wood storks only nested at the Corkscrew colony in 2 of our 6 study years, both of which were gradual dry-season recession years where high-density prey concentrations were documented at our study sites (600 pairs in 2006 (1,428 young fledged) and 1,120 pairs in 2009 (2,570 young fledged); unpublished data). Higher topography in BCR relative to Everglades graminoid marshes facilitates the formation of high-density prey patches earlier in the hydrologic year. For this reason, a delay in nest initiation would be expected to impact wood stork colonies foraging in the BCR more than Everglades-foraging colonies. Restoration of lost short hydroperiod wetlands may aid in the recovery of the BCR wood stork population by moving nest initiation back closer to its historic range.

RECOMMENDATIONS FOR CONTINUED MONITORING TO EVALUATE POTENTIAL IMPACTS OF CERP IN NORTHEASTERN BCNP

Background

The intent of this study was to institute a monitoring program with the express purpose of detecting any potential impacts of CERP activities on the wading bird prey base in Big Cypress National Preserve. The first several years of this study focused on developing sampling methodology and gaining an understanding of the spatial and temporal variation in BCNP aquatic fauna communities at three sentinel sites that represented the range of relatively pristine freshwater forested wetland habitats in BCNP. In the last few years of this project, our effort shifted to include establishing sampling sites in the BCNP Addition Lands where impacts from CERP were expected. After extensive consultation with hydrologists and ecologists knowledgeable of this region, we identified a systematic network of study sites with varying proximity to the L-28 Interceptor and West Feeder canals (Figure 28). The idealized design consisted of a grid of 12 sites: 1 existing (sentinel) site (L28), 8 new sites in BCNP and 3 new sites on the Seminole Tribe of Florida's Big Cypress Reservation.

Establishment of the monitoring network

Accessibility (and resulting travel time) was a significant factor in development of a study design for this region. We determined that all-terrain vehicle (ATV) was the most practical method of accessing this remote backcountry, as all other modes of transportation were deemed impossible, impractical, or too costly. Assistance from the Florida Trail Association was invaluable in locating passable trails in this region, as we were not able to find reliable trail maps or first-hand knowledge of this region from NPS. Travel to study sites entailed highway vehicle travel to access points (Nobles Grade trail at I-75 MM 63 or L28I levee at I-75 MM 51), ATV travel (ranging from 1 to 9 miles) and foot travel. ATV travel was often treacherous, as many trails were not maintained and treefall across trails was frequently encountered and impassable (travel with a chainsaw is necessary in some areas). Foot travel often entailed traversing thick palmetto or Brazilian pepper stands. Total travel time to sites from Corkscrew Swamp Sanctuary (Naples, FL) ranged 1.5 to 2.5 h (each way). Due to these time considerations, the decision was made to establish only one plot at each site, trading inter-site replication for greater spatial coverage.

Prior to the premature termination of this study, we established 6 of the 8 proposed sites in BCNP: Noble's Grade (NG), Panther Camp (PC), Hunter's Hideaway (HH), Cowbell Strand (CS), Pipeline (PL) and Looneyville (LV) (Figure 29). We were in the process of trying to reach and locate sampleable areas at the two remaining locations in BCNP and obtaining access Tribal lands.

Importance of continued monitoring

The sensitivity of Big Cypress aquatic fauna communities to hydrology demonstrated through the six years of this study emphasizes the importance of including this region in CERP monitoring efforts. Reduced hydroperiod in over-drained Big Cypress wetlands (such as the BCNP Addition Lands) may cause significant reduction in the standing stock of aquatic wading bird prey (Duever 2005). While hydroperiod restoration of this region through CERP projects (e.g., Seminole Big Cypress, Big Cypress/L-28 Interceptor Modification, WCA3 Decompartmentalization) would be expected to result in recovery of fish density, size structure and relative abundance, hypothetical relationships between fish population and hydroperiod are based on those developed in Everglades graminoid wetlands, which differ considerably from the Big Cypress region in topography and hydrology (detailed in the Big Cypress Basin Conceptual Ecological Model (http://www.evergladesplan.org/pm/studies/study_docs/swfl/swffs_cems_bigcypress.pdf)). This monitoring effort could also serve as a reference for restoration of the Picayune Strand. In the absence of any wading bird monitoring in BCNP (and the difficulty of doing such aerial monitoring in a forested system), aquatic fauna serve as a critical surrogate indicator. This effort is especially important for federally endangered Wood Storks who rely on BCNP as a primary foraging ground. With no other ecological monitoring currently conducted within freshwater BCNP, elimination of this monitoring effort has left no means for adaptive management and has disregarded all potential effects of hydrologic change from CERP activities on the Preserve. We feel it is imperative to re-establish this monitoring program in order to establish baseline data at the network of sites in the BCNP Addition Lands, and to continue monitoring throughout restoration to allow a feedback mechanism (adaptive management) as CERP projects alter hydrology in this region.

CONCLUSIONS

This report demonstrates the utility of an aquatic fauna monitoring program in assessing the effect of Everglades restoration efforts on forested wetlands in the BCR, specifically, Big Cypress National Preserve. Variation in standing stocks of macroinvertebrates, fish and amphibians were demonstrated on both spatial and intra- and inter-annual temporal scales. Relationships elucidated between hydrology and aquatic fauna communities clearly demonstrate the ability of this monitoring program to assess effects of the Comprehensive Everglades Restoration Plan (CERP) on freshwater forested wetlands in southern Florida.

Of the six hydrologic years encompassed by this monitoring program, two years had dry-season rainfall patterns that were detrimental to creating the high-density prey concentrations upon which nesting wading birds rely. While these weather events are a natural component of multi-decadal patterns, they emphasize the need for additional data collection to better describe fauna response to hydrology in this region. Furthermore, these data suggest that the increased frequency of extreme weather events (rain, drought, cold, etc.) that are predicted with global climate change will likely negatively impact the aquatic prey populations on which wading birds depend.

The sensitivity of this region to hydrologic variation also suggests the effects of restoration activities on aquatic fauna community may be considerable. Increased water delivery to the BCNP Addition Lands from L-28-Interceptor and West Feeder canals will increase wetland hydroperiod, and perhaps more importantly, the spatial extent of inundation. While data collected in this six-year monitoring effort allow some ability to predict change, our ability to model these data will improve with continued monitoring of sites within the Big Cypress Addition Lands.

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REFERENCES

Addison D, Barry M, Bartoszek I, Ceilley D, Schmid J, Schuman M (2006) Pre-restoration wildlife surveys in the Southern Golden Gate Estates (2001-2004). Final Report to South Florida Water Management District.

Arner DH, Robinette JR, Frasier JE, Gray MH (1976) Effects of channelization of the Luxipalila River on fish, aquatic invertebrates, water quality and furbearers. U.S. Fish and Wildlife Services, Office of Biological Services FWS/OBS-76-08

Bondavalli C, Ulanowicz RE (1999) Unexpected effects of predators upon their prey: the case of the American alligator. Ecosystems 2: 49-63

Browder JA (1976) Water, wetlands, and wood storks in southwest Florida. Ph.D. Dissertation, University of Florida, Gainesville, FL

Carlson JE, Duever MJ (1979) Seasonal fish population fluctuations in South Florida swamp. Proc.Annual Conf.S.E.Assoc.Fish & Wildlife Agencies 31: 603-611

Chick JH, Ruetz CR, III, Trexler JC (2004) Spatial scale and abundance patterns of large fish communities in freshwater marshes of the Florida Everglades. Wetlands 24: 652-664

Clarke KR (1993) Non-parametric multivariate analyses of change in community structure. Australian Journal of Ecology 18: 117-143

Clarke KR, Warwick RM (1994) Change in Marine Communities: An Approach to Statistical Analyses and Interpretation. Natural Environmental Research Council, Plymouth Marine Laboratory

Dalrymple GH (1995) Wildlife inventory for baseline studies of the Addition Lands of the Big Cypress National Preserve: Miami, Report prepared for Florida International University.

DeAngelis DL, Trexler JC, Cosner C, Obaza A, Jopp F (2010) Fish population dynamics in a seasonally varying wetland. Ecological Modelling 221: 1131-1137

Dorn NJ (2008) Colonization and reproduction of large macroinvertebrates are enhanced by drought-related fish reductions. Hydrobiologia 605

Dorn NJ, Trexler JC (2007) Crayfish assemblage shifts in a large drought-prone wetland: the roles of hydrology and competition. Freshwater Biology 52: 2411

Duever MJ (2005) Big Cypress Regional Ecosystem conceptual ecological model. Wetlands 25: 843-853

Duever MJ, Carlson JE, Meeder JF, Duever LC, Gunderson LH, Riopelle LA, Alexander TR, Myers RL, Spangler DP (1986) The Big Cypress National Preserve. National Audubon Society, New York, NY

Dunker KJ (2003) Non-indigenous fishes in restored and natural wetlands on the Big Cypress Seminole Indian Reservation. M.S., Florida Atlantic University, Boca Raton, FL

Ellis G, Zokan M, Loftus WF, Lorenz JJ (2004) Inventory of the freshwater fishes of the Big Cypress National Preserve. NPS Inventory & Monitoring Program

Frederick PC, Gawlik DE, Ogden JC, Cook MI, Lusk M (2009) The White Ibis and Wood Stork as indicators for restoration of the everglades ecosystem. Ecological Indicators 9: S83-S95

Garrett PB (2007) Factors affecting wading bird prey concentrations in the Everglades during the dry season. M.S., Florida Atlantic University, Boca Raton, FL

Gawlik DE (2002) The effects of prey availability on the numerical response of wading birds. Ecological Monographs 72: 329-346

Goss CW (2006) Fish colonization of temporary wetlands in the context of the Everglades landscape. M.S. Thesis, Florida International University, Miami, FL

Hall HD, Lambou VW (1990) The ecological significance to fisheries of bottomland hardwood systems: values, detrimental impacts, and assessment: the report of the fisheries workgroup. In: Gosselink JG, Lee LC, Muir TA (eds.) Ecological Processes and Cumulative Impacts: Illustrated by Bottomland Hardwood Wetland Ecosystems. Lewis Publishers, Inc., Chelsea, MI, pp 481-531

Jopp F, DeAngelis DL, Trexler JC (2010) Modeling seasonal dynamics of small fish cohorts in fluctuating freshwater marsh landscapes. Landscape Ecology 25: 1041-1054

Jordan CF, Coyne S, Trexler JC (1997) Sampling fishes in heavily vegetated habitats: the effects of habitat structure on sampling characteristics of the 1-m^2 throw trap. Transactions of the American Fisheries Society 126: 1012-1020

Kahl MP, Jr. (1964) Food ecology of the wood stork (*Mycteria americana*) in Florida. Ecological Monographs 34: 97-17

Kushlan JA (1974) Effects of a natural fish kill on the water quality, plankton, and fish population of a pond in the Big Cypress Swamp, Florida. Transactions of the American Fisheries Society 103: 235-242

Kushlan JA (1979) Temperature and oxygen in an Everglades alligator pond. Hydrobiologia 67: 267-271

Kushlan JA (1980) Population fluctuations of Everglades fishes. Copeia 1980: 870-874

Kushlan JA, Hunt BP (1979) Limnology of an alligator pond in South Florida. Florida Scientist 42: 65-84

Kwak TJ (1988) Lateral movement and use of floodplain habitat by fishes of the Kankakee River, Illinois. American Midland Naturalist 120: 241-249

Lambou VW (1959) Fish populations of backwater lakes in Louisiana. Transactions of the American Fisheries Society 88: 7-15

Liston SE (2006) Interactions between nutrient availability and hydroperiod shape macroinvertebrate communities in Florida Everglades marshes. Hydrobiologia 569: 357

Liston SE, Katin N, Lorenz JJ (2007) Sampling aquatic fauna in cypress forests, Big Cypress National Preserve. In: Liston SE, Lorenz JJ, McIvor CC, Loftus WF (eds.) Development and Testing of Protocols for Sampling Fishes in Forested Wetlands of Southern Florida. pp

Liston SE, Lorenz JJ (2008) Development and testing of protocols for sampling fishes in forested wetlands in Southern Florida - Big Cypress Region. Annual Report submitted to Everglades National Park & the MAP RECOVER team, 42 pp.

Loftus WF, Eklund A-M (1994) Long-term dynamics of an Everglades small-fish assemblage. In: Davis SM, Ogden JC (eds.) Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Boca Raton, FL, pp 461-483

Loftus WF, Kushlan JA (1987) The freshwater fishes of southern Florida. Bulletin of the Florida State Museum of Biological Sciences 31

Lorenz JJ, Langan-Mulrooney B, Frezza PE, Harvey RG, Mazzotti FJ (2009) Roseate spoonbill reproduction as an indicator for restoration of the Everglades and the Everglades estuaries. Ecological Indicators 9S: S96-S107

Lorenz JJ, Serafy JE (2006) Subtropical wetland fish assemblages and changing salinity regimes: implications for everglades restoration. Hydrobiologia 569: 401-422

Main MB, Ceilley DW, Stansly P (2007) Freshwater fish assemblages in isolated South Florida wetlands. Southeastern Naturalist 6: 343-350

Nelson D, Metzger R (2005) Distribution and abundance of wading birds in the Big Cypress National Preserve, 2005. Report to the US Army Corps of Engineers by Lone Star Environmental Studies.

Ogden JC, Kushlan JA, Tilmant JT (1976) Prey selectivity by the wood stork. The Condor 78: 324-330

Poulakis GR, Shenker JM, Taylor DS (2002) Habitat use by fishes after tidal reconnection of an impounded estuarine wetland in the Indian River Lagoon, Florida (USA). Wetlands Ecology and Management 10: 51-69

Rehage JS, Robblee M (2009) Role of marsh-mangrove interface habitats as aquatic refuges for wetland fishes and other aquatic animals. p. 118 pp., Annual Assessment Report for Year 2009.

Ross ST, Baker JA (1983) The response of fishes to periodic spring floods in a southeastern stream. American Midland Naturalist 109: 1-14

Ruetz CR, III, Trexler JC, Jordan F, Loftus WF, Perry SA (2005) Population dynamics of wetland fishes: spatio-temporal patterns synchronized by hydrological disturbance? Journal of Animal Ecology 74: 322-332

Schofield PJ, Huge DH (2010) Low-temperature tolerance of two non-native fishes (*Hoplosternum littorale* [Hancock 1828], *Cichlasoma bimaculatum* [Linnaeus 1758]) established in south Florida USA. Florida Scientist 74: 73-83

Schofield PJ, Loftus WF, Kobza M, Cook MI, Slone DH (2009) Tolerance of nonindigenous cichlid fishes (*Cichlasoma urophthalmus*, *Hemichromis letourneauxi*) to low temperature: laboratory and field experiments in south Florida. Biological Invasions 12: 2441-2457

Shaw RG, Mitchell-Olds T (1993) Anova for unbalanced data: an overview. Ecology 74: 1638-1645

Trexler JC (2005) Analyzing the impact of the Intermidiate Operating Plan (IOP) in the Eastern Everglades, Everglades National Park.

Trexler JC, Goss CW (2009) Aquatic fauna as indicators for Everglades restoration: applying dynamic targets in assessments. Ecological Indicators 9: S108-S119

Trexler JC, Loftus WF, Perry SA (2005) Disturbance frequency and community structure in a twenty-five year intervention study. Oecologia

Turner TF, Trexler JC, Miller GL, Toyer KE (1994) Temporal and spatial dynamics of larval and juvenile fish abundance in a temperate floodplain river. Copeia 1994: 174-183

Williams AJ, Trexler JC (2006) A preliminary analysis of the correlation of food-web characteristics with hydrology and nutrient gradients in the southern Everglades. Hydrobiologia 569: 493-504

Woodall WR, Adams JG, Heise J (1975) Invertebrates eaten by Altamaha River fish. 39th Meeting GA Entomological Society, St. Simons Island, March 19-21

TABLES

Table 1. Universal Transverse Mercator (UTM; Zone 17R) coordinates for the three sentinel study sites (three replicate plots comprise each site).

Site		Latitude (N)	Longitude (E)
BI	Plot 1	2899800	468302
	Plot 2	2899783	470915
	Plot 3	2899883	471404
L28	Plot 1	2897378	509257
	Plot 2	2897699	509042
	Plot 3	2895157	510252
RP	Plot 1	2874238	507579
	Plot 2	2875313	507578
	Plot 3	2875984	508750

Table 2. Model parameters for linear regressions used to predict daily water depths at each study plot from water depths recorded at nearby hydrologic stations.

Site-Plot	Nearby Station	N	α	β	R^2
BI-1	BCA2	21	-373.58	1.0571	0.9040
BI-2	BCA2	8	-358.20	1.0193	0.9267
BI-3	BCA2	19	-434.44	1.2319	0.9361
L28-1	BCA18	24	-348.31	1.0665	0.9356
L28-2	*	*	*	*	*
L28-3	BCA18	20	-423.83	1.2672	0.9071
RP-1	BCA5	17	-280.95	1.2473	0.9355
RP-2	BCA5	26	-296.56	1.3006	0.9374
RP-3	BCA5	20	-313.38	1.3014	0.9061

* insufficient data were available from staff gauge at L28-2

Table 3. Estimated inundation dates for study sites (based on daily predicted water depths) during each hydrologic year of monitoring.

Site		WY06	WY07	WY08	WY09	WY10	WY11
BI	Plot 1	6/1/05	7/4/06	8/7/07	7/16/08	6/2/09	*
	Plot 2	6/1/05	7/4/06	8/7/07	7/16/08	6/2/09	*
	Plot 3	6/1/05	7/4/06	8/7/07	7/16/08	6/2/09	*
L28	Plot 1	6/13/01†	7/8/06	6/13/07	6/18/08	5/29/09	*
	Plot 2	6/13/01†	7/8/06	6/13/07	6/18/08	5/29/09	*
	Plot 3	6/14/01†	7/12/06	6/19/07‡	6/19/08	5/30/09	*
RP	Plot 1	6/2/05	7/3/06	6/13/07	6/21/08‡	5/28/09	*
	Plot 2	6/3/05	7/3/06	6/13/07	6/21/08‡	5/29/09	*
	Plot 3	6/3/05	7/3/06	7/10/07	6/21/08‡	6/2/09	*

* No re-inundation date in WY11 since study site did not dry in WY10

† Excluded from analyses, as estimated inundation date is so far prior to the start of the study (low confidence in predicted water depth outside temporal span of study)

‡ Date is approximate, as daily hydrologic data were unavailable during re-inundation

Table 4. Average abundance (±SE) of common macrophyte taxa, total submerged/ floating macrophytes and total emergent macrophytes recorded in WY06-WY11 in each sampling zone (habitat type) of each study site. 'Common' taxa were identified by incidence ≥5% and were quantified using either %cover (submerged/floating taxa) or density (no./m^2; emergent taxa). "na" indicates habitat was not present at the site, and "--" indicates species was not collected at the site.

	Site		
	BI	L28	RP
WET PRAIRIE			
Submerged/Floating (%cover)			
Bacopa caroliniana	na	1.2 ± 1.2	2.0 ± 1.2
Eleocharis vivipara	na	0.3 ± 0.3	2.6 ± 2.0
Periphyton (benthic)	na	19.7 ± 13.1	21.7 ± 7.0
Periphyton (floating)	na	3.1 ± 1.4	5.8 ± 2.5
Utricularia foliosa	na	--	0.5 ± 0.2
Utricularia purpurea	na	--	1.1 ± 0.6
Total	na	24.8 ± 12.5	34.2 ± 7.4
Emergent (no./m^2)			
Cladium jamaicense	na	--	2 ± 1
Eleocharis spp.	na	8 ± 5	2 ± 1
Oxypolis spp.	na	<1	<1
Panicum spp.	na	4 ± 3	1
Paspalidium geminatum	na	<1	<1
Pluchea rosea	na	<1	<1
Rhynchospora spp.	na	70 ± 11	35 ± 9
Stillingea aquatica	na	6 ± 2	<1
Taxodium distichum	na	<1	1
Unidentified	na	4 ± 3	1
Total	na	94 ± 11	45 ± 9
INTERMEDIATE-HYDROPERIOD FOREST			
Submerged/Floating (%cover)			
Bacopa caroliniana	--	10.3 ± 1.9	23.9 ± 3.7
Ludwigia repens	0.6 ± 0.2	27.5 ± 3.3	1.4 ± 1.3
Sagittaria graminea	5.2 ± 1.9	4.5 ± 1.1	1.7 ± 0.9
Total	7.5 ± 2.5	44.8 ± 3.9	31.0 ± 3.6
Emergent (no./m^2)			
Cladium jamaicense	<1	--	2 ± 1
Panicum spp.	<1	1	1
Taxodium distichum	<1	<1	<1

	BI	Site L28	RP
Unidentified	1	<1	<1
Total	3 ± 1	4 ± 1	5 ± 2
LONG-HYDROPERIOD FOREST			
Submerged/Floating (%cover)			
Bacopa caroliniana	<0.1	0.3 ± 0.3	17.7 ± 2.9
Ludwigia repens	0.3 ± 0.2	0.1	4.1 ± 1.2
Salvinia minima	0.1 ± 0.1	7.3 ± 2.5	--
Total	10.6 ± 3.1	9.6 ± 2.7	22.7 ± 3.4
DEEP WATER REFUGE			
Submerged/Floating (%cover)			
Salvinia minima	na	35.8 ± 30.8	--
Utricularia foliosa	na	7.5 ± 7.5	--
Total	na	43.5 ± 38.5	--

Table 5. Incidence (%I), total number (N), relative abundance (%RA), biomass (g) and relative biomass (%) of invertebrate, fish and amphibian taxa collected throughout this study (WY06-WY11). Superscripts indicate adult (A) and larval (L) life-stages of insects.

Latin name	Common name	I (%)	N	RA (%)	Biomass (g)	Relative Biomass (%)
Invertebrates						
Gastropoda						
Physella spp.	Physid snail	2.5	38	0.2	1.8	0.0
Planorbella spp.	Planorbid snail	10.4	257	1.1	79.2	0.9
Pomacea paludosa	Florida apple snail	2.1	22	0.1	154.4	1.7
	Unidentified gastropod	0.5	6	0.0	1.2	0.0
Bivalvia						
Sphaeridae	Fingernail clam	6.0	464	1.9	25.4	0.3
Unionidae	Freshwater mussel	0.1	1	0.0	44.6	0.5
---	Unidentified bivalve	0.1	1	0.0	0.1	0.0
Oligochaeta	Earthworm	0.7	8	0.0	1.5	0.0
Hirudinea	Leech	1.2	12	0.0	3.0	0.0
Ephemeroptera^{L}	Mayfly	0.2	2	0.0	0.5	0.0
Odonata						
Anisoptera^{L}	Dragonfly	32.6	853	3.5	131.8	1.5
Coenagrionidae^{L}	Damselfly	4.2	82	0.3	1.7	0.0
Heteroptera						
---	Unidentified	0.2	3	0.0	0.0	0.0
Belostoma spp.	Giant waterbug	3.0	48	0.2	8.2	0.1
Lethocerus spp.^{A}	Toe biter	2.1	28	0.1	44.4	0.5
Corixidae^{A}	Water boatman	2.9	65	0.3	0.8	0.0
Pelocoris femoratus^{A}	Alligator flea	4.7	85	0.3	2.5	0.0
Ranatra spp.^{A}	Water scorpion	1.1	12	0.0	2.0	0.0
Coleoptera						
---	Unidentified adult	5.3	282	1.2	3.8	0.0
---	Unidentified larvae	0.5	5	0.0	0.1	0.0
Dyticidae^{A}	Predaceous diving beetle	1.6	33	0.1	1.2	0.0

Latin name	Common name	I (%)	N	RA (%)	Biomass (g)	Relative Biomass (%)
Cybister spp.[L]		1.9	48	0.2	3.5	0.0
Gyrinidae[A]	Whirligig water beetle	9.7	629	2.6	47.5	0.5
Dryopidae[A]	Long-toed water beetle	1.9	193	0.8	3.5	0.0
Diptera						
Tipulidae[L]	Crane fly	0.1	1	0.0	0.1	0.0
Stratiomyidae[L]	Soldier fly	0.2	2	0.0	0.1	0.0
Crustacea						
Isopoda		0.1	1	0.0	0.0	0.0
Palaemonetes paludosus	Riverine grass shrimp	34.0	14,727	60.5	1,398.7	15.8
Procambarus alleni	Everglades crayfish	46.5	2,955	12.1	5,639.9	63.7
Procambarus fallax	Slough crayfish	28.9	963	4.0	877.5	9.9
Procambarus spp.	Unidentified crayfish	23.8	2,519	10.3	377.5	4.3
	Unidentified invert	0.7	8	0.0	1.0	0.0
---	**Total invertebrates**		**24,353**		**8,857.5**	
Fish						
Lepisosteus platyrhincus	Florida gar	0.3	3	0.0	919.8	7.2
Esox niger	Chain pickerel	0.1	1	0.0	2.1	0.0
Notemigonus crysoleucas	Golden shiner	0.8	15	0.1	29.8	0.2
Notropis petersoni	Coastal shiner	0.4	6	0.0	1.9	0.0
Ameiurus natalis	Yellow bullhead	1.3	246	1.0	1,393.1	10.9
Ameiurus nebulosus	Brown bullhead	0.1	1	0.0	9.5	0.1
Noturus gyrinus	Tadpole madtom	0.3	4	0.0	7.2	0.1
Clarius batrachus	Walking catfish	0.2	36	0.1	938.9	7.4
*Hoplosternum littorale**	Brown hoplo	2.7	100	0.4	1,810.1	14.2
Fundulus chrysotus	Golden topminnow	4.6	64	0.3	32.6	0.3
Fundulus confluentus	Marsh killifish	3.1	72	0.3	71.1	0.6
Jordanella floridae	Flagfish	28.9	4,823	19.6	2,126.3	16.7
Lucania goodei	Bluefin killifish	22.9	1,185	4.8	204.5	1.6
*Belonesox belizanus**	Pike killifish	0.3	3	0.0	2.5	0.0
Gambusia holbrooki	Eastern mosquitofish	63.4	13,188	53.7	2,570.8	20.1
Heterandria formosa	Least killifish	19.3	2,810	11.4	135.9	1.1

Latin name	Common name	I (%)	N	RA (%)	Biomass (g)	Relative Biomass (%)
Poecilia latipinna	Sailfin molly	5.2	199	0.8	106.4	0.8
Labidesthes sicculus	Brook silverside	4.9	272	1.1	74.5	0.6
Elassoma evergladei	Everglades pygmy sunfish	4.3	114	0.5	15.4	0.1
Enneacanthus gloriosus	Bluespotted sunfish	3.9	86	0.3	61.1	0.5
Lepomis spp.	Sunfish (unidentified)	6.6	96	0.4	9.3	0.1
Lepomis gulosus	Warmouth	12.3	385	1.6	718.9	5.6
Lepomis macrochirus	Bluegill sunfish	1.0	11	0.0	43.2	0.3
Lepomis marginatus	Dollar sunfish	13.5	512	2.1	461.6	3.6
Lepomis microlophus	Redear sunfish	0.9	11	0.0	23.6	0.2
Lepomis punctatus	Spotted sunfish	6.1	123	0.5	163.7	1.3
Etheostoma fusiforme	Swamp darter	1.6	19	0.1	13.9	0.1
*Cichlasoma bimaculatum**	Black acara	5.6	95	0.4	182.6	1.4
*Tilapia mariae**	Spotted tilapia	0.6	10	0.0	29.1	0.2
*Cichlasoma urophthalmus**	Mayan cichlid	1.9	26	0.1	601.1	4.7
*Hemichromis letourneuxi**	African jewelfish	0.1	1	0.0	1.2	0.0
Cichlidae*	Cichlid (unidentified)	0.1	1	0.0	0.1	0.0
---	Unidentified fish	3.2	54	0.2	1.2	0.0
---	Larval fish (unidentified)	0.1	1	0.0	0.0	0.0
---	**Total fish**		**24,573**		**12,762.9**	
Amphibians						
Notophthalmus viridescens	Peninsula newt	2.5	43	13.3	52.8	19.6
---	Tadpole	10.2	275	85.1	216.7	80.3
Sirenidae	Siren	0.2	5	1.5	0.4	0.1
---	**Total amphibians**		**323**		**269.8**	

*non-indigenous species

Table 6. ANCOVA of the biomass (g./m^2) of common amphibian, fish and invertebrate taxa among study years, study sites and with hydroperiod (DSD=days since dry) during gradual dry-season recession years.

	Year		Site		DSD		DSD^2		
	$F_{4,231}$	P	$F_{2,231}$	P	$F_{1,231}$	P	$F_{1,231}$	P	R^2
AMPHIBIANS									
Tadpoles	0.931	0.447	**5.627**	**0.004**	**5.498**	**0.020**	1.358	0.245	0.119
FISHES									
Eastern mosquitofish	**15.323**	**<0.001**	2.776	0.064	2.561	0.111	**34.237**	**<0.001**	0.518
Least killifish	1.462	0.215	0.388	0.679	2.573	0.110	**9.413**	**0.002**	0.161
Flagfish	**10.467**	**<0.001**	**11.978**	**<0.001**	**10.545**	**0.001**	**35.736**	**<0.001**	0.371
Warmouth	**8.970**	**<0.001**	**3.903**	**0.022**	0.579	0.448	**8.886**	**0.003**	0.263
Dollar sunfish	**2.884**	**0.023**	**6.482**	**0.002**	1.628	0.203	**11.639**	**0.001**	0.253
Bluefin killifish	1.184	0.319	**3.055**	**0.049**	**8.031**	**0.005**	**23.496**	**<0.001**	0.277
Total fish	**10.262**	**<0.001**	0.132	0.877	**3.931**	**0.049**	**34.858**	**<0.001**	0.480
INVERTEBRATES									
Grass shrimp	2.336	0.056	0.880	0.416	**37.474**	**<0.001**	**89.672**	**<0.001**	0.530
Everglades crayfish	**2.813**	**0.026**	**20.103**	**<0.001**	**13.672**	**<0.001**	**16.805**	**<0.001**	0.267
Slough crayfish	**7.395**	**<0.001**	**5.254**	**0.006**	2.126	0.146	1.115	0.292	0.240
Crayfish (juvenile)	**16.536**	**<0.001**	**3.494**	**0.032**	**32.751**	**<0.001**	**13.475**	**<0.001**	0.419
Total Crayfish	**4.515**	**0.002**	**11.334**	**<0.001**	**9.361**	**0.002**	**12.333**	**0.001**	0.177

Table 7. ANCOVA of the biomass (g./m^2) of common amphibian, fish and invertebrate taxa among study sites and with hydroperiod (DSD=days since dry) during the dry-season reversal year.

	Site		DSD		DSD^2		
	$F_{1,52}$	P	$F_{1,52}$	P	$F_{1,52}$	P	R^2
AMPHIBIANS							
Tadpoles	1.278	0.263	0.358	0.552	0.386	0.537	0.030
FISHES							
Eastern mosquitofish	0.240	0.626	**6.604**	**0.013**	**4.377**	**0.041**	0.200
Least killifish	2.948	0.092	0.543	0.465	0.184	0.670	0.106
Flagfish	**6.000**	**0.018**	3.025	0.088	2.013	0.162	0.186
Warmouth	0.059	0.810	0.757	0.388	0.225	0.637	0.097
Dollar sunfish	1.938	0.170	**5.532**	**0.022**	**4.998**	**0.030**	0.126
Bluefin killifish	0.079	0.779	2.513	0.119	1.461	0.232	0.112
Total Fish	0.013	0.911	**6.240**	**0.016**	3.495	0.067	0.252
INVERTEBRATES							
Grass shrimp	0.129	0.721	1.822	0.183	1.143	0.290	0.073
Everglades crayfish	3.298	0.075	2.758	0.103	3.648	0.062	0.134
Slough crayfish	1.078	0.304	**5.577**	**0.022**	**4.389**	**0.041**	0.138
Crayfish (juvenile)	1.194	0.280	0.057	0.812	0.106	0.746	0.030
Total Crayfish	3.752	0.058	**3.997**	**0.051**	**4.823**	**0.033**	0.147

FIGURES

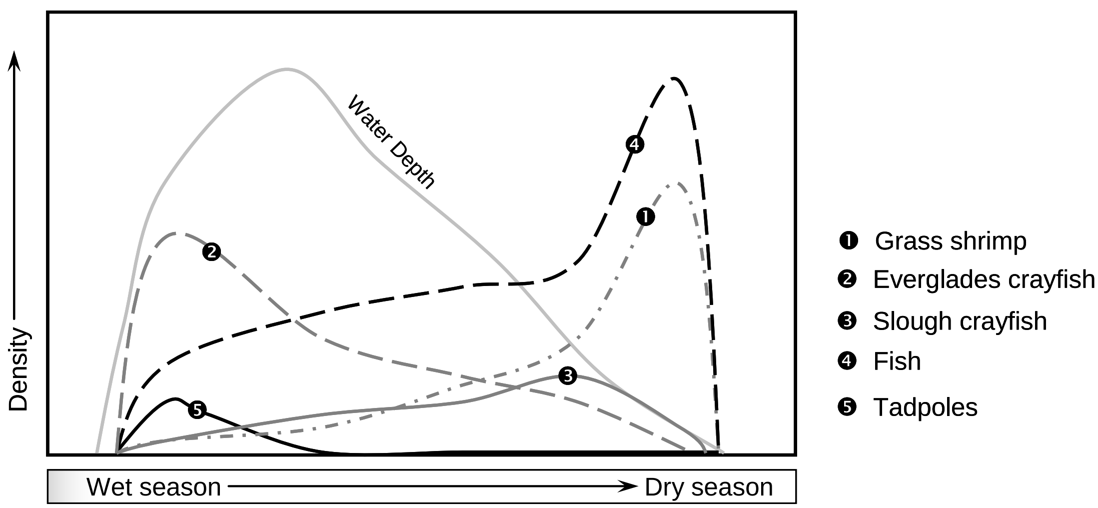

Figure 1. Conceptual model depicting relative variation in standing stock of key fauna groups (grass shrimp, Everglades crayfish, slough crayfish, fish, tadpoles) throughout the hydrologic year, developed from data collected in the first two years of this study (WY06 and WY07). Gray line indicates the relative water depth in long-hydroperiod cypress forests during this time.

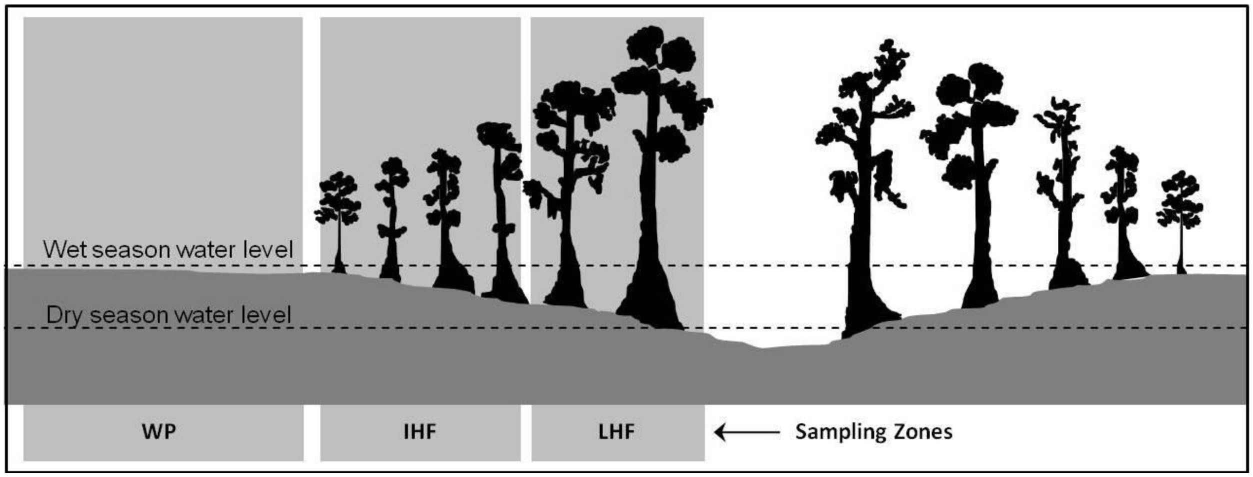

Figure 2. Schematic representation of the location and proximity of sampling zones (WP=wet prairie, IHF=intermediate-hydroperiod forest, LHF=long-hydroperiod forest) within a ~1 ha study plot. In most years, all sampling zones dried during the course of the dry season and were re-inundated early in the wet season.

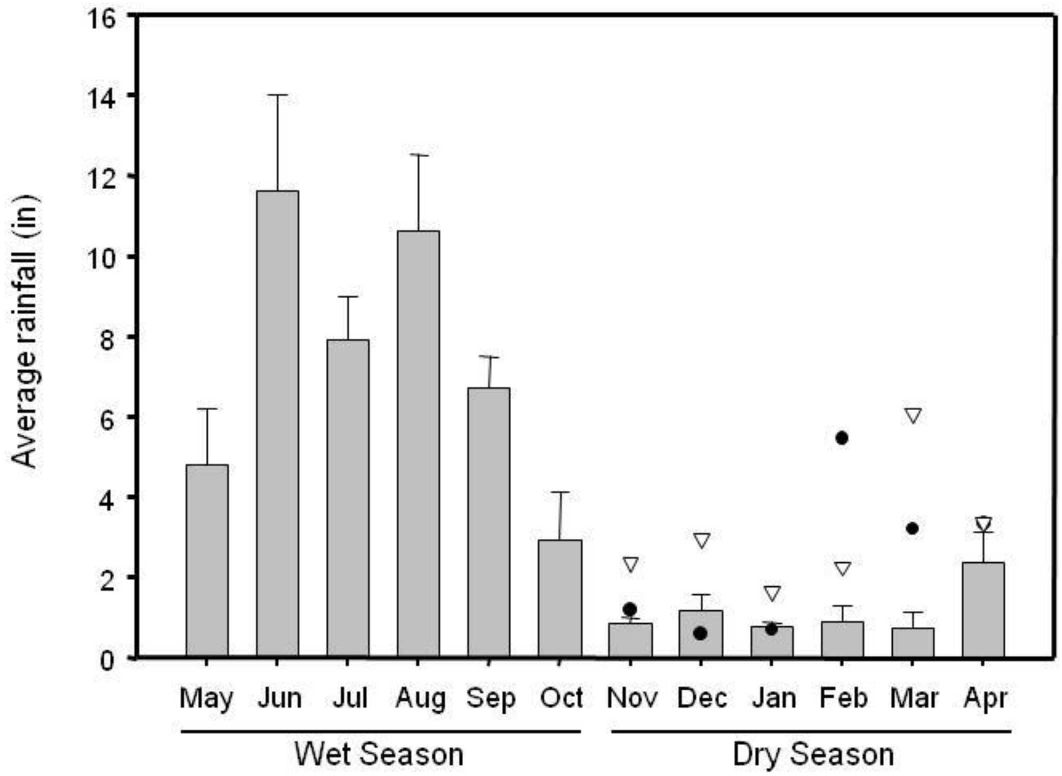

Figure 3. Average monthly rainfall (inches; error bars represent ±1 SE) in central Big Cypress National Preserve (hydrologic station BCA 15 (26°02'22.542", 81°01'37.624")) throughout this study (WY06-WY11). Bars represent wet season months of all years and dry season months of years with typical rainfall patterns (WY06, WY07, WY09, and WY11). Dry season months with unseasonable rainfall are indicated (WY08: closed circle; WY10: open triangle).

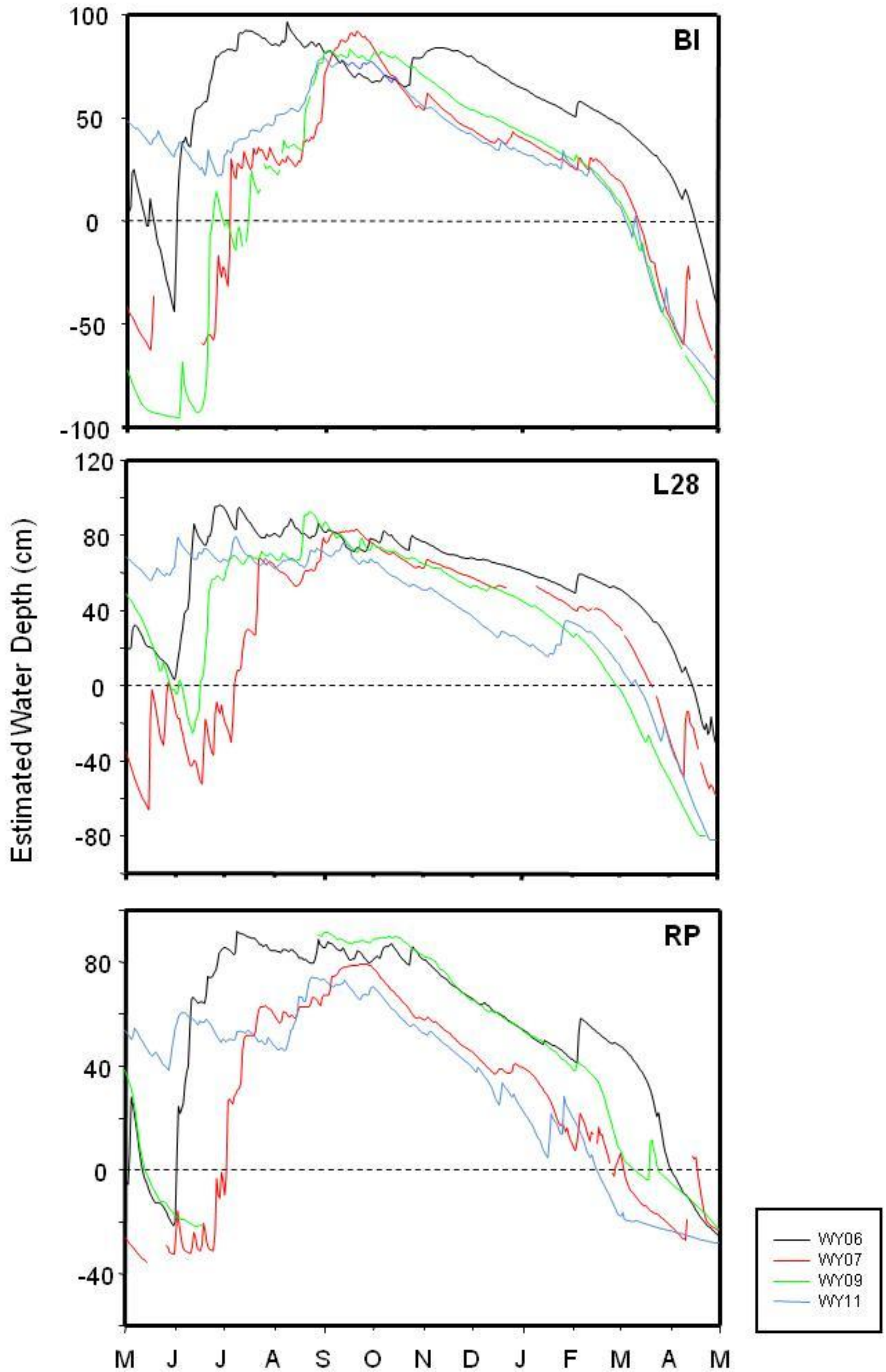

Figure 4. Estimated daily water depth (cm) at staff gauges in one plot of each sentinel study site WY06, WY07, WY09 and WY11. Zero represents approximate ground level in long-hydroperiod forest zone.

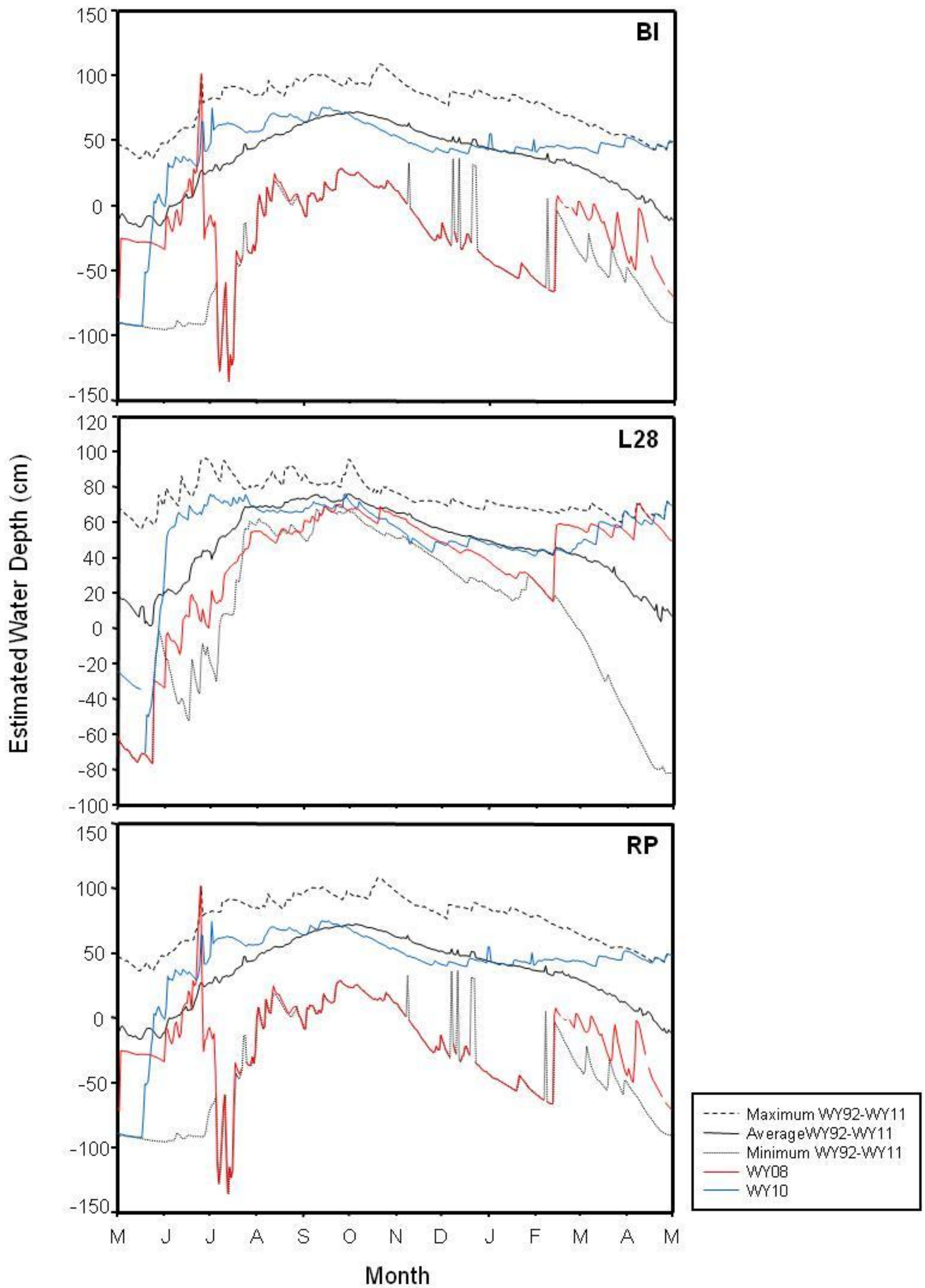

Figure 5. Average, maximum and minimum estimated daily water depth (cm) at staff gauges in one plot of each sentinel study site WY92-WY11 (5/1/91-4/30/11). L28-1 data were unavailable prior to 12/5/00. The two atypical dry season years, WY08 and WY10, are highlighted in red and blue, respectively. Zero represents approximate ground level in long-hydroperiod forest zone.

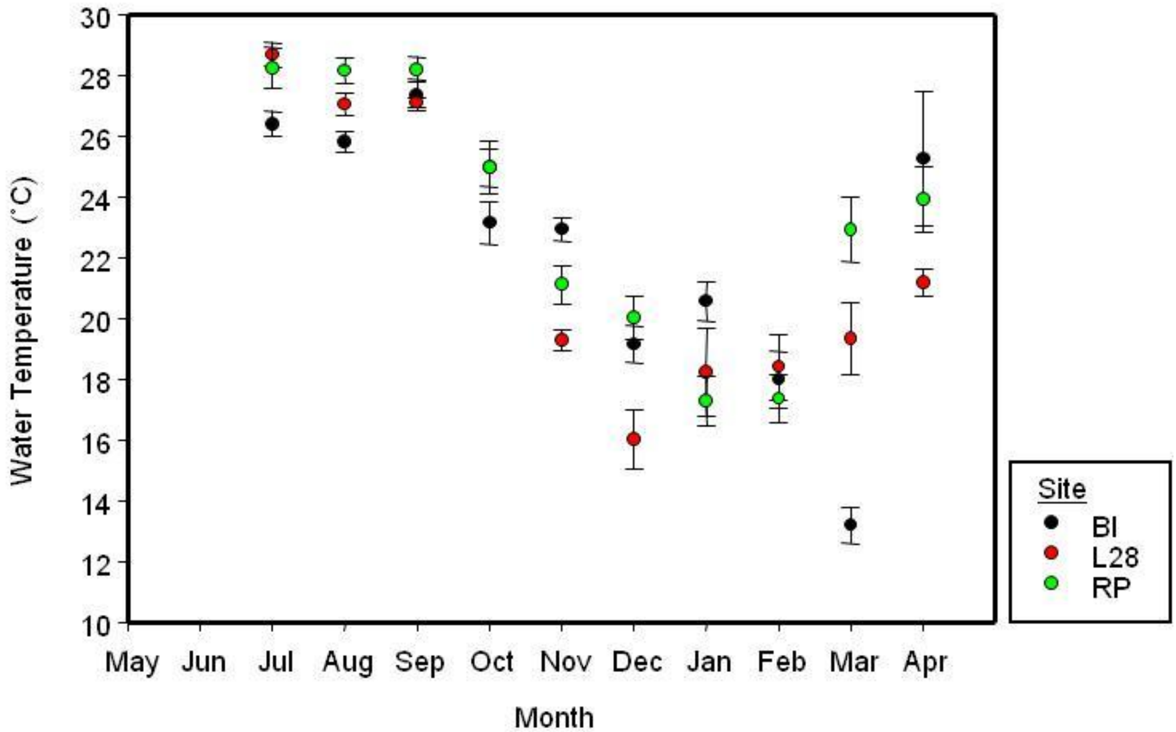

Figure 6: Average monthly water temperature (°C)(±SE) recorded at study sites during sampling events through the course of this study (WY06-WY11).

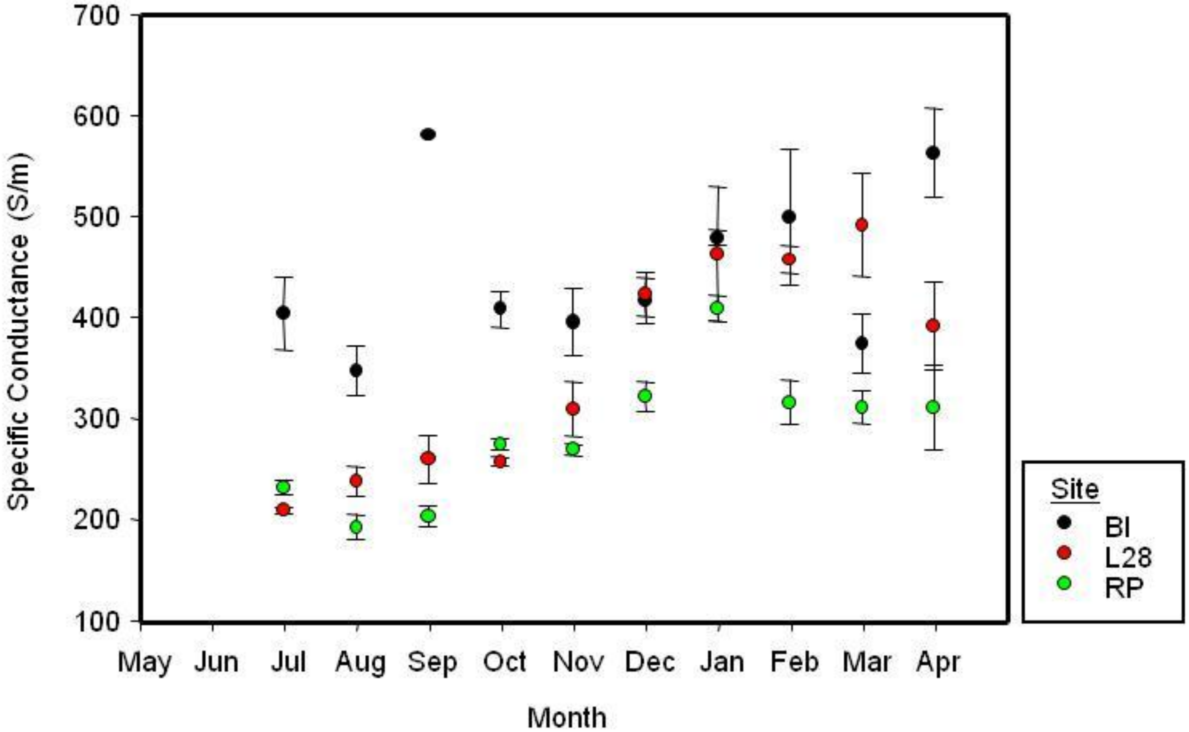

Figure 7: Average monthly specific conductance (S/m)(±SE) recorded at study sites during sampling events through the course of this study (WY06-WY11).

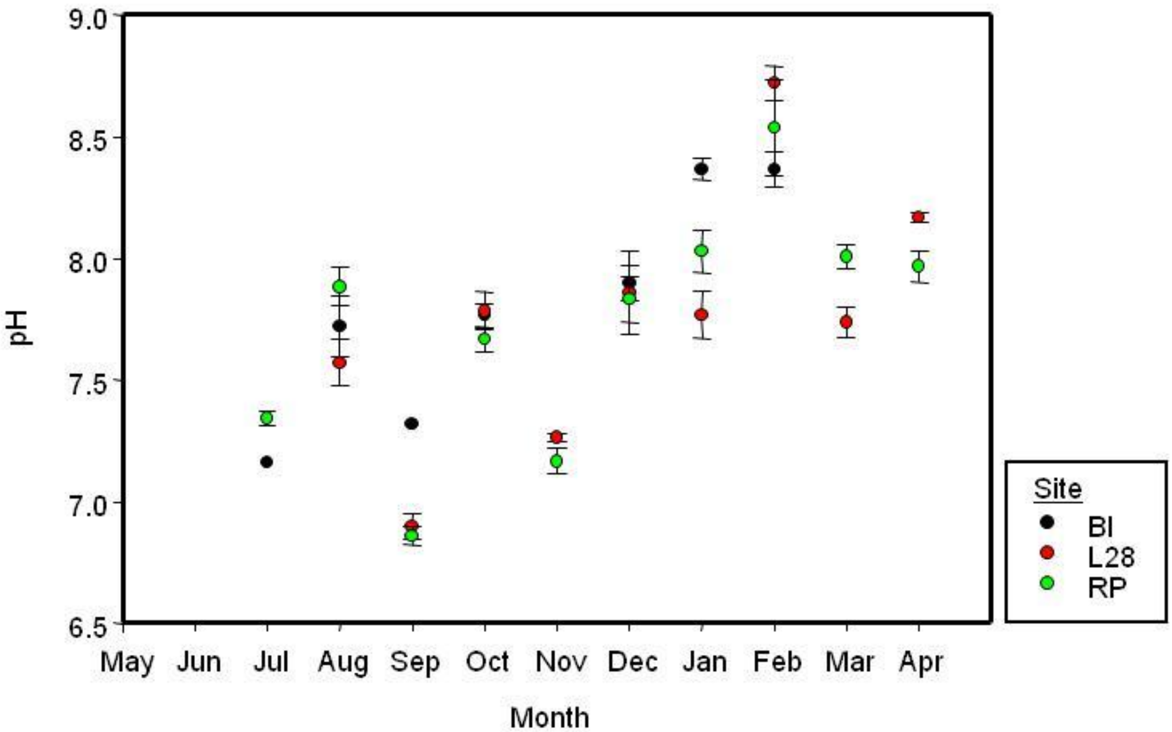

Figure 8: Average monthly pH (±SE) recorded at study sites during sampling events through the course of this study (WY06-WY11).

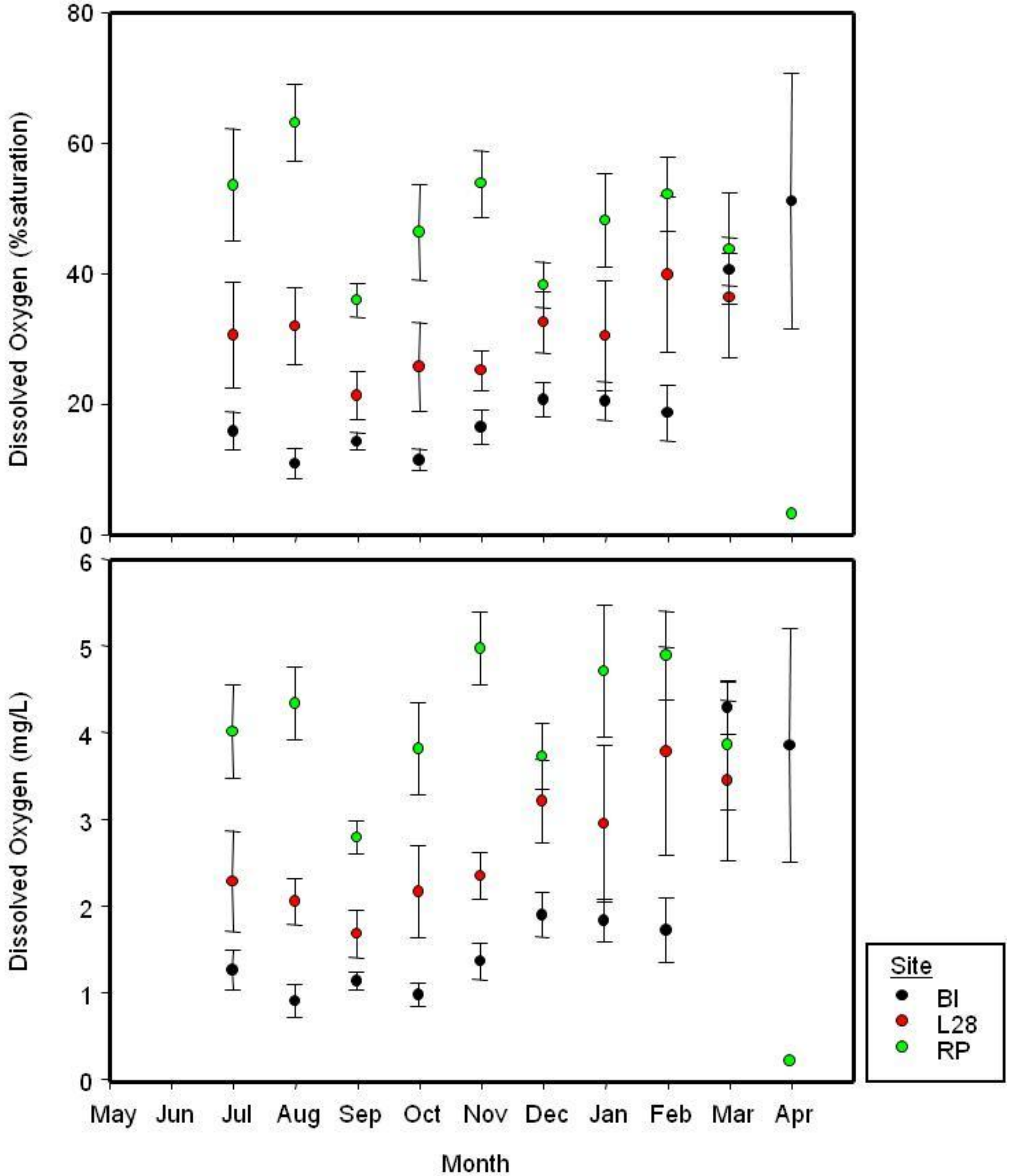

Figure 9: Average monthly dissolved oxygen (%saturation: top; mg/L: bottom)(±SE) recorded at study sites during sampling events through the course of this study (WY06-WY11).

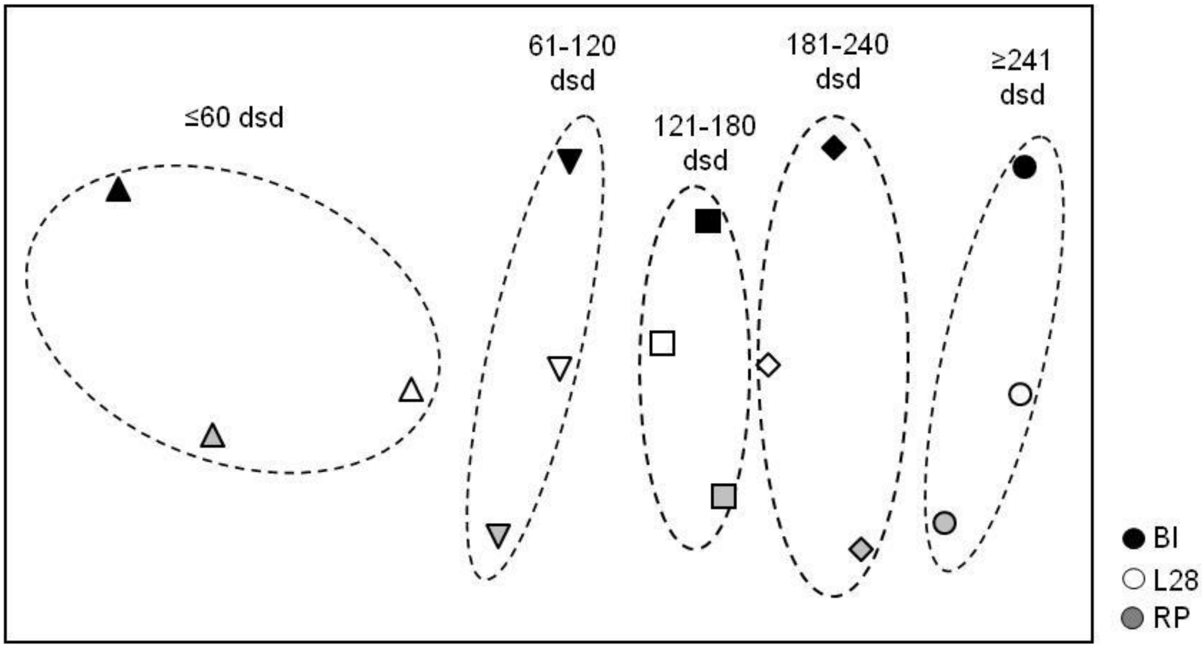

Figure 10. Non-metric multidimensional scaling plot of aquatic fauna communities throughout the hydrologic year in study years with gradual dry-season recession (stress=0.06).

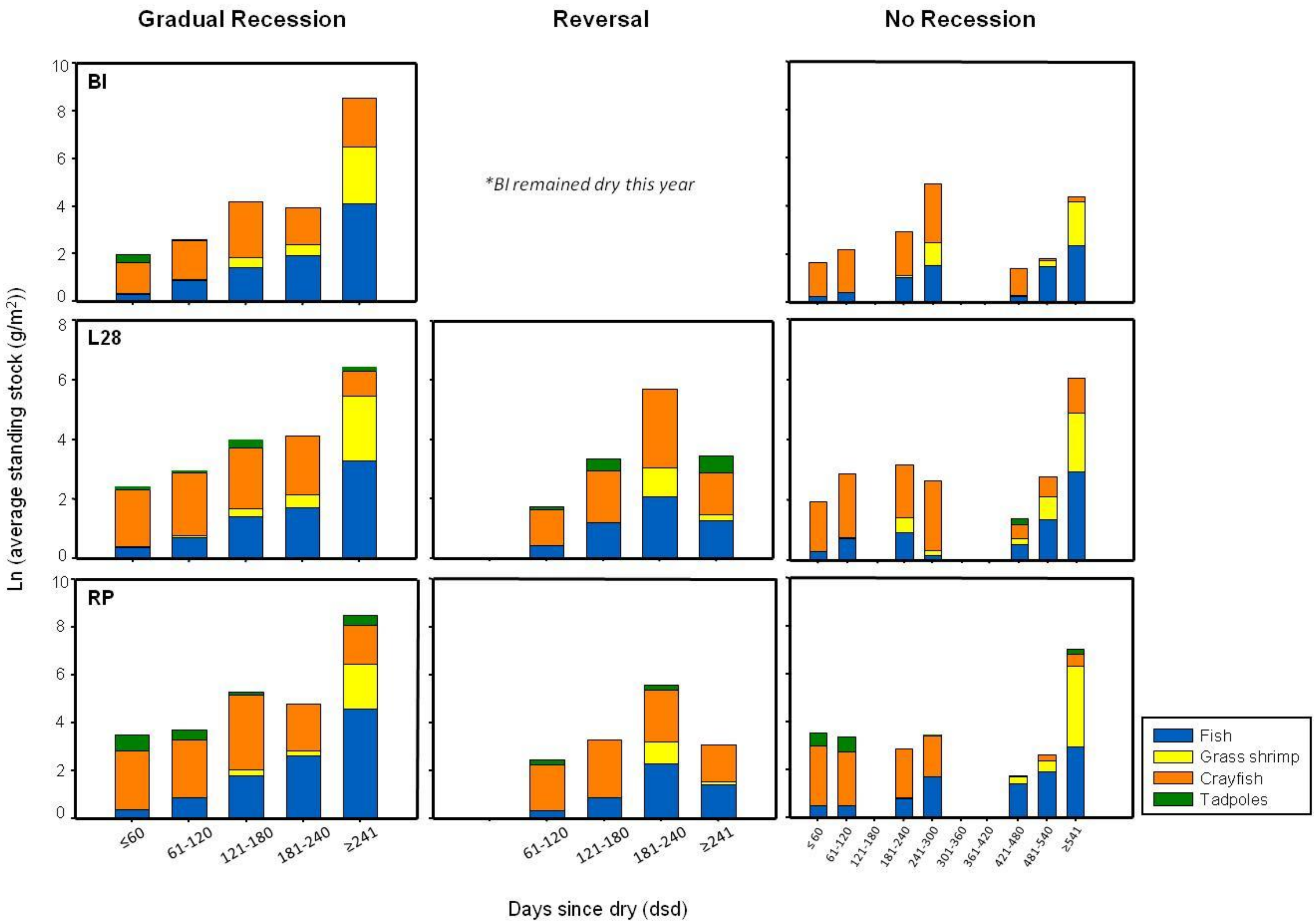

Figure 11. Contributions of key taxonomic groups (fish, grass shrimp, crayfish, tadpoles) to total aquatic fauna biomass at each site relative to duration of inundation during gradual recession, reversal and no recession years.

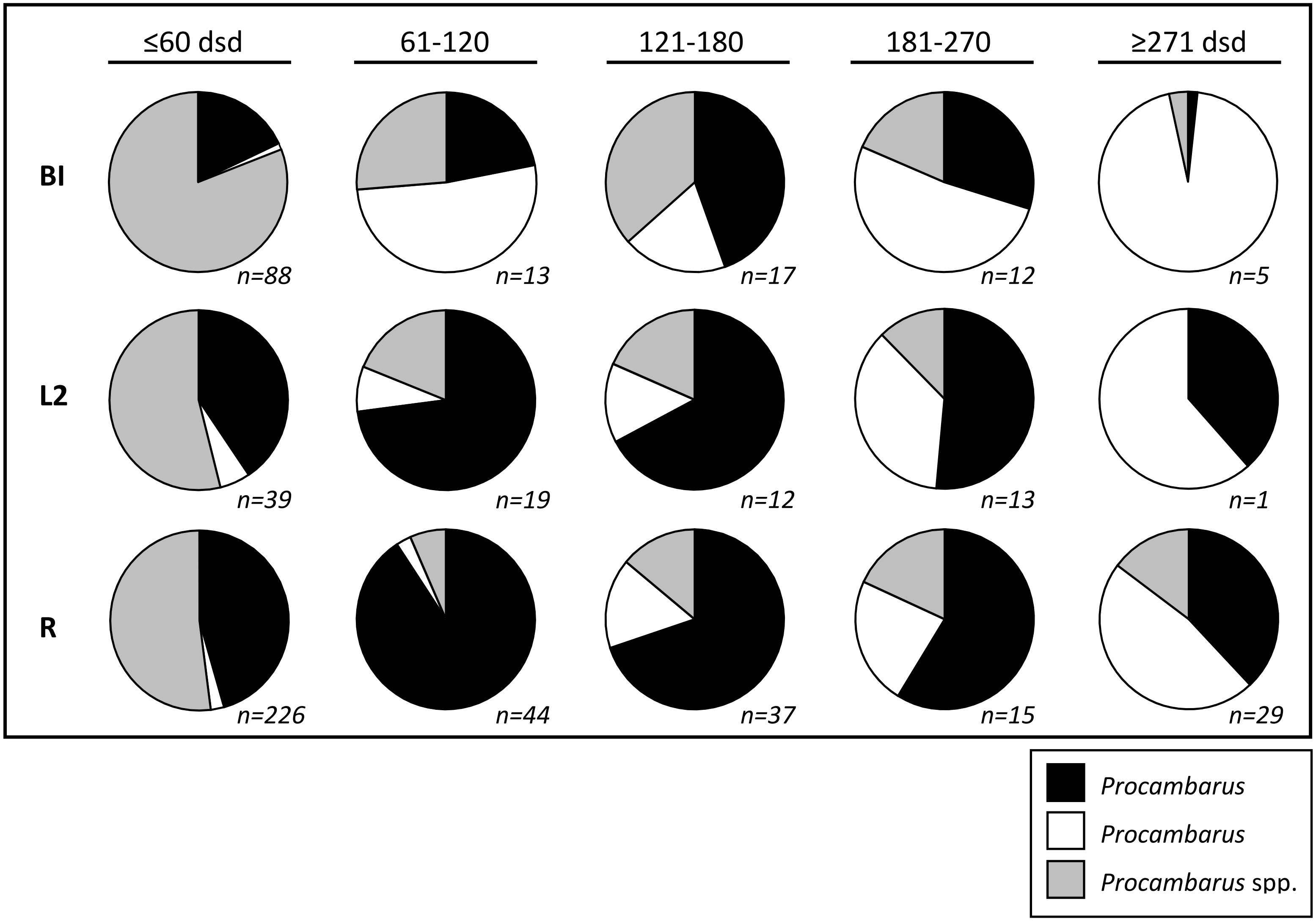

Figure 12. Relative contributions of Everglades crayfish (*Procambarus alleni*), slough crayfish (*Procambarus fallax*) and juvenile crayfish (*Procambarus* spp.) to the total number of crayfish collected throughout gradual dry-season recession years. N represents total number of individuals collected.

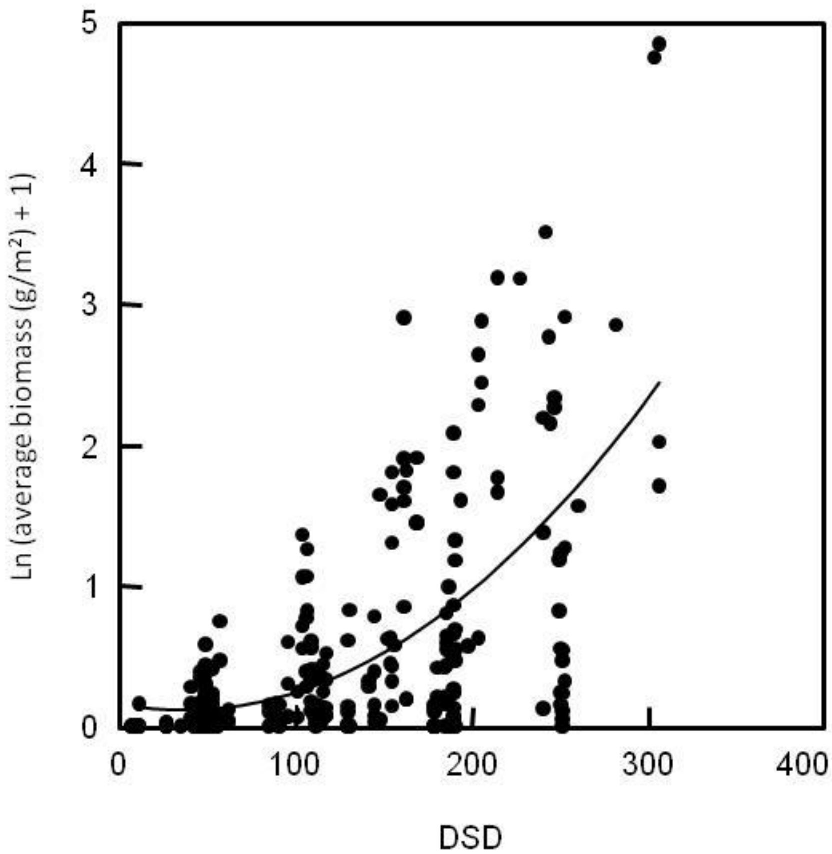

Figure 13. Variation in the average biomass (g wet weight/m^2) of eastern mosquitofish (*Gambusia holbrooki*) with hydroperiod (DSD=days since dry) at study sites in Big Cypress National Preserve during gradual dry-season recession years.

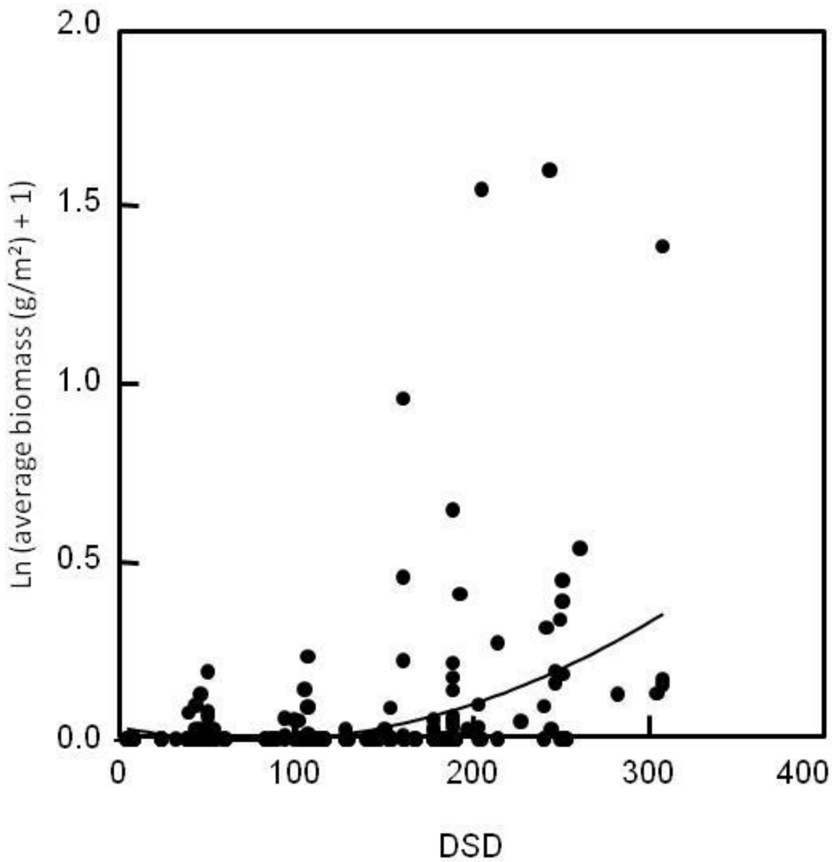

Figure 14. Variation in the average biomass (g wet weight/m^2) of least killifish(*Heterandria formosa*) with hydroperiod (DSD=days since dry) at study sites in Big Cypress National Preserve during gradual dry-season recession years.

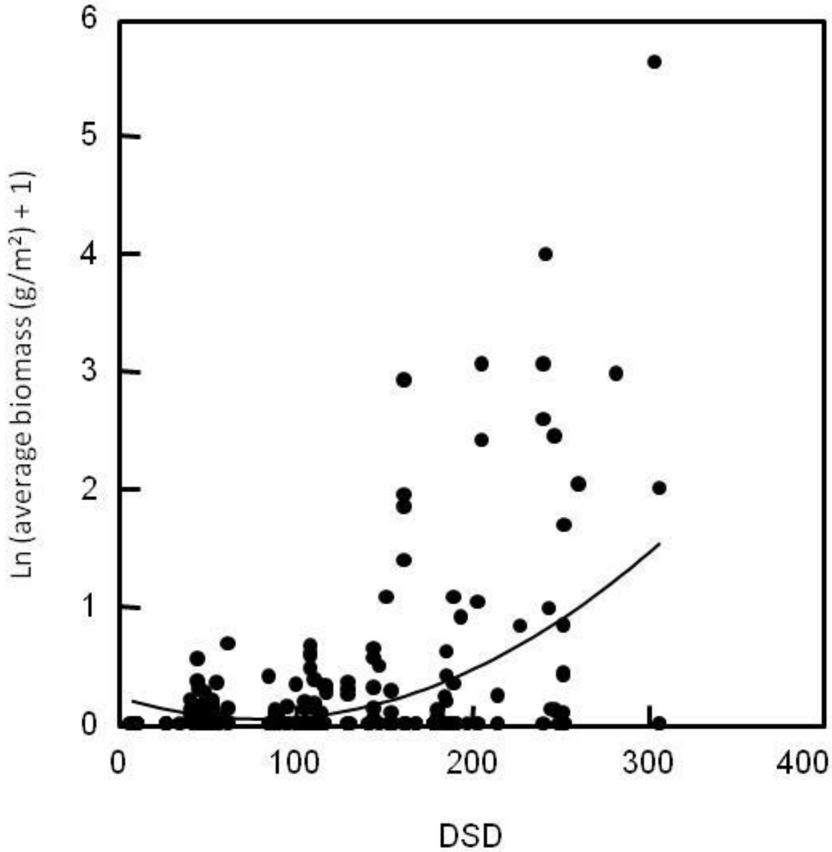

Figure 15. Variation in the average biomass (g wet weight/m^2) of flagfish (*Jordanella floridae*) with hydroperiod (DSD=days since dry) at study sites in Big Cypress National Preserve during gradual dry-season recession years.

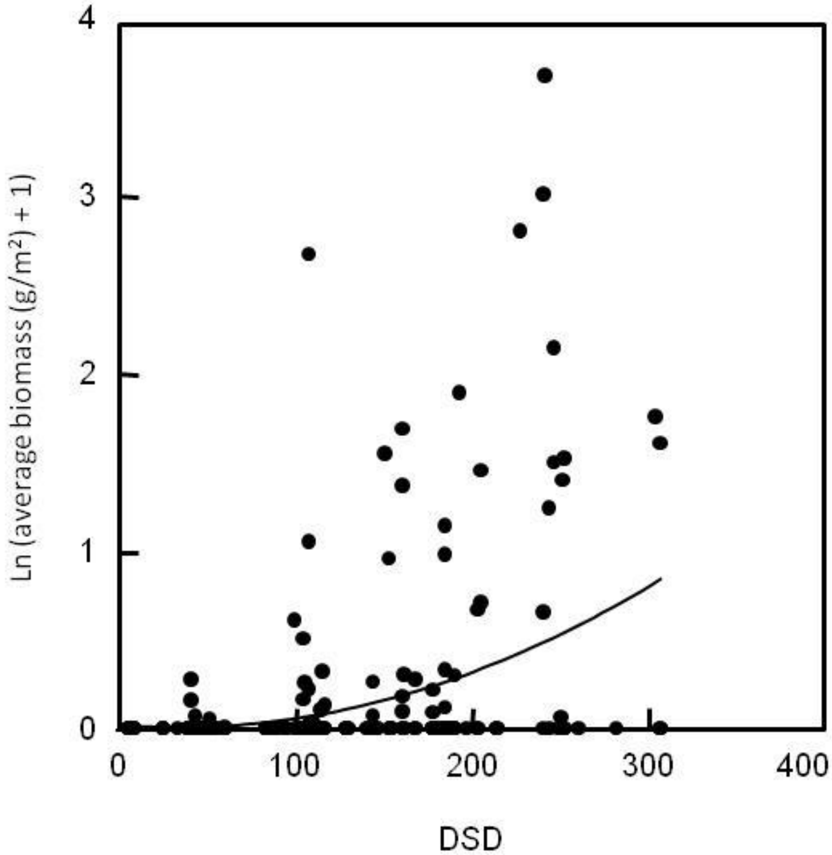

Figure 16. Variation in the average biomass (g wet weight/m^2) of warmouth (*Lepomis gulosus*) with hydroperiod (DSD=days since dry) at study sites in Big Cypress National Preserve during gradual dry-season recession years.

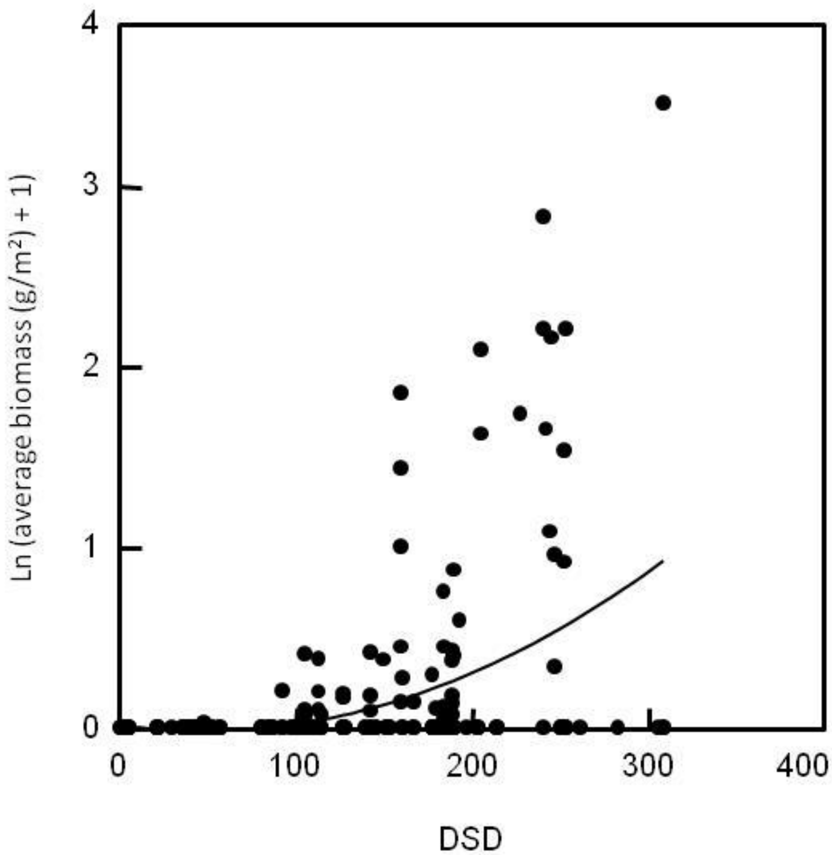

Figure 17. Variation in the average biomass (g wet weight/m^2) of dollar sunfish (*Lepomis marginatus*) with hydroperiod (DSD=days since dry) at study sites in Big Cypress National Preserve during gradual dry-season recession years.

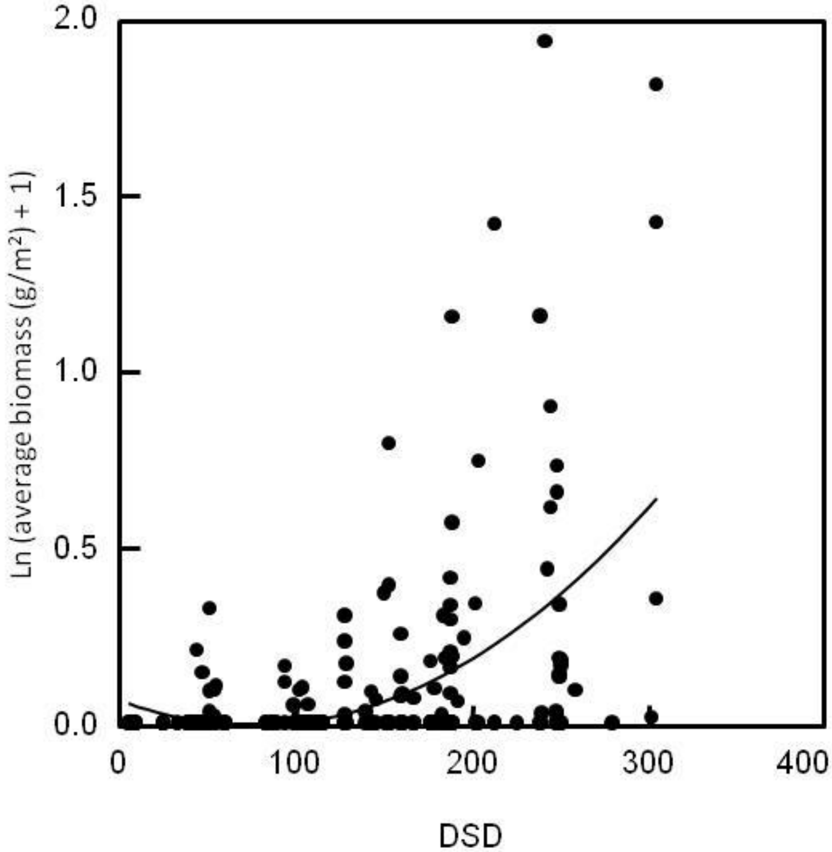

Figure 18. Variation in the average biomass (g wet weight/m^2) of bluefin killifish (*Lucania goodei*) with hydroperiod (DSD=days since dry) at study sites in Big Cypress National Preserve during gradual dry-season recession years.

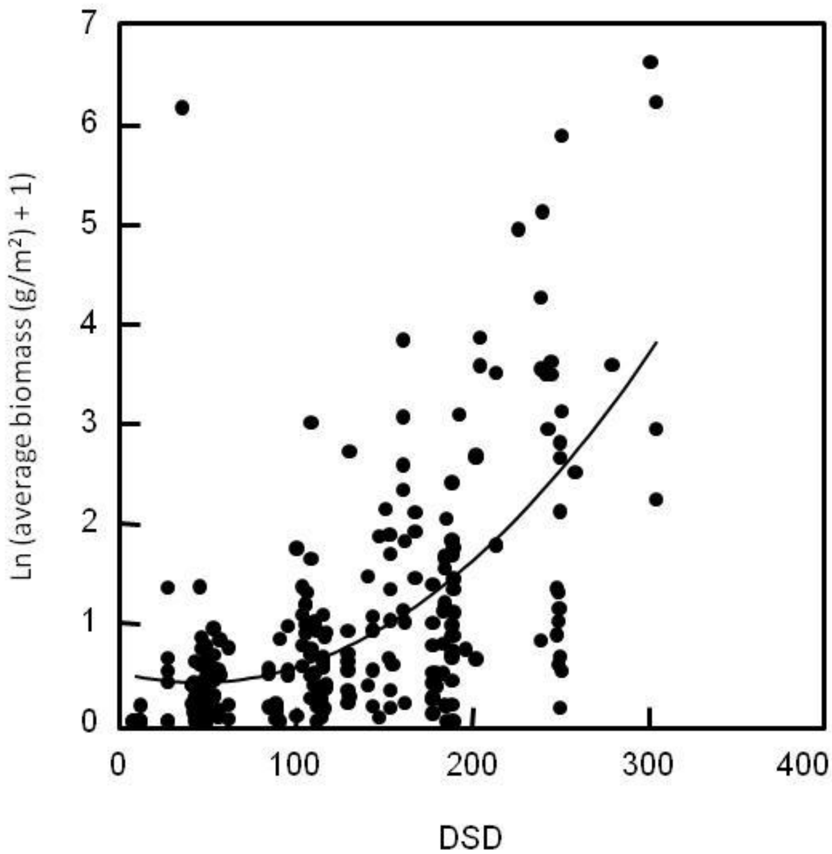

Figure 19. Variation in the average biomass (g wet weight/m^2) of total fish with hydroperiod (DSD=days since dry) at study sites in Big Cypress National Preserve during gradual dry-season recession years.

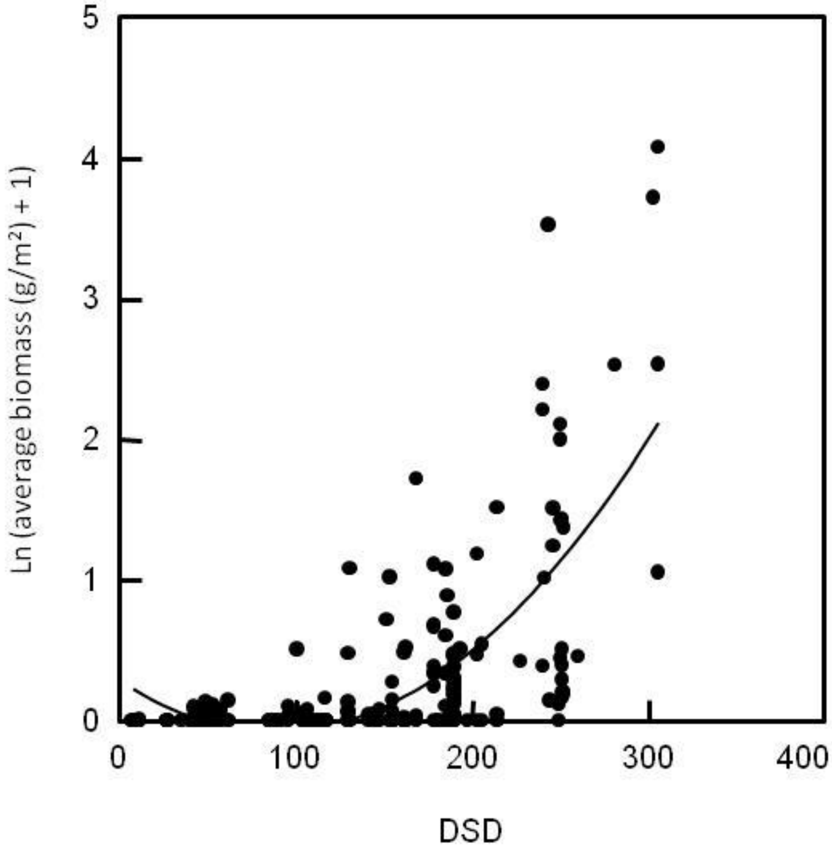

Figure 20. Variation in the average biomass (g wet weight/m^2) of grass shrimp (*Palaemonetes paludosus*) with hydroperiod (DSD=days since dry) at study sites in Big Cypress National Preserve during gradual dry-season recession years.

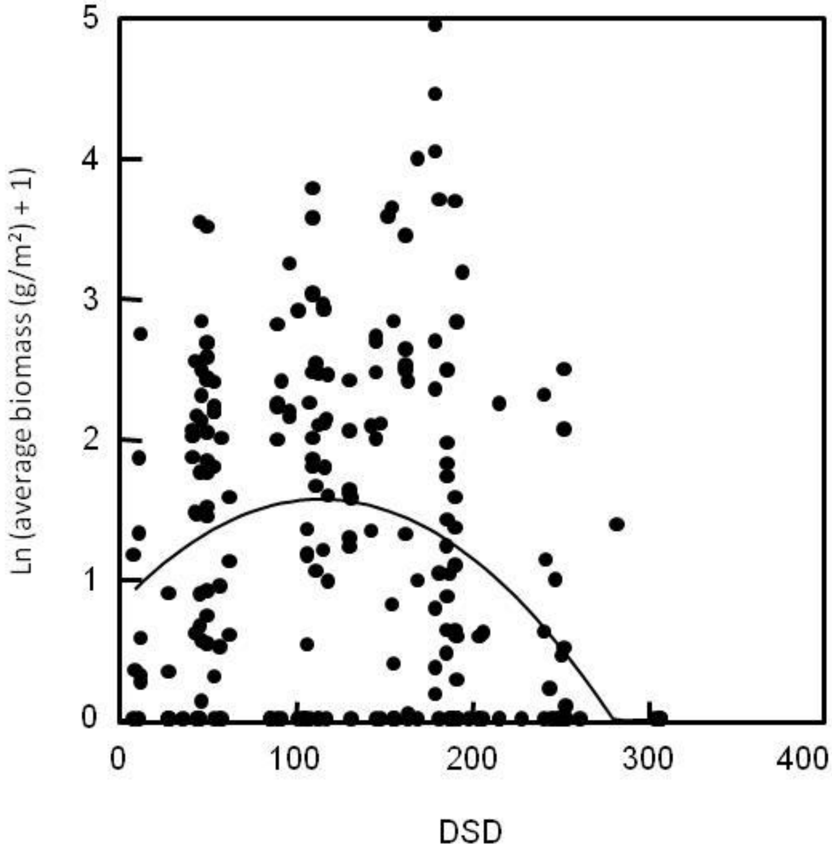

Figure 21. Variation in the average biomass (g wet weight/m^2) of Everglades crayfish (*Procambarus alleni*) with hydroperiod (DSD=days since dry) at study sites in Big Cypress National Preserve during gradual dry-season recession years.

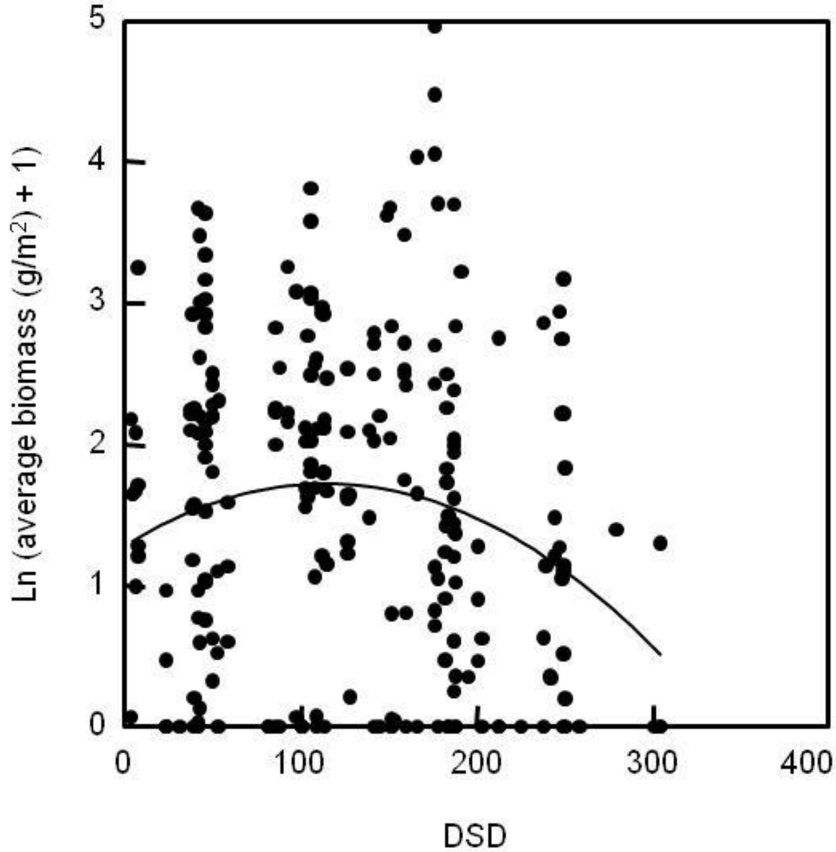

Figure 22. Variation in the average biomass (g wet weight/m^2) of crayfish (*Procambarus* spp.) with hydroperiod (DSD=days since dry) at study sites in Big Cypress National Preserve during gradual dry-season recession years.

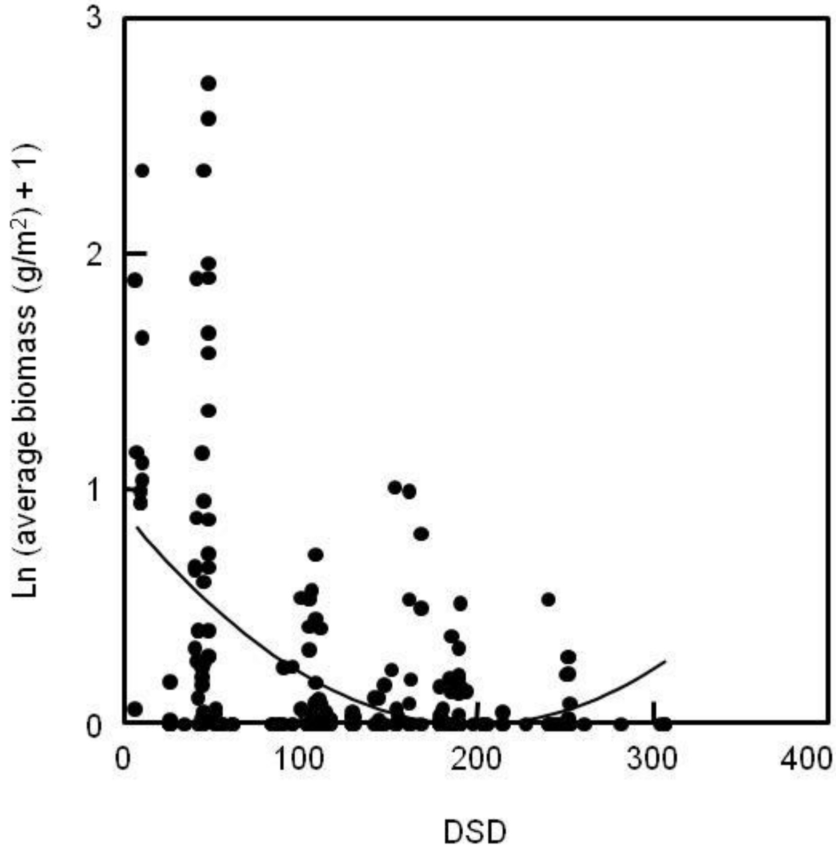

Figure 23. Variation in the average biomass (g wet weight/m^2) of juvenile crayfish (*Procambarus* spp.) with hydroperiod (DSD=days since dry) at study sites in Big Cypress National Preserve during gradual dry-season recession years.

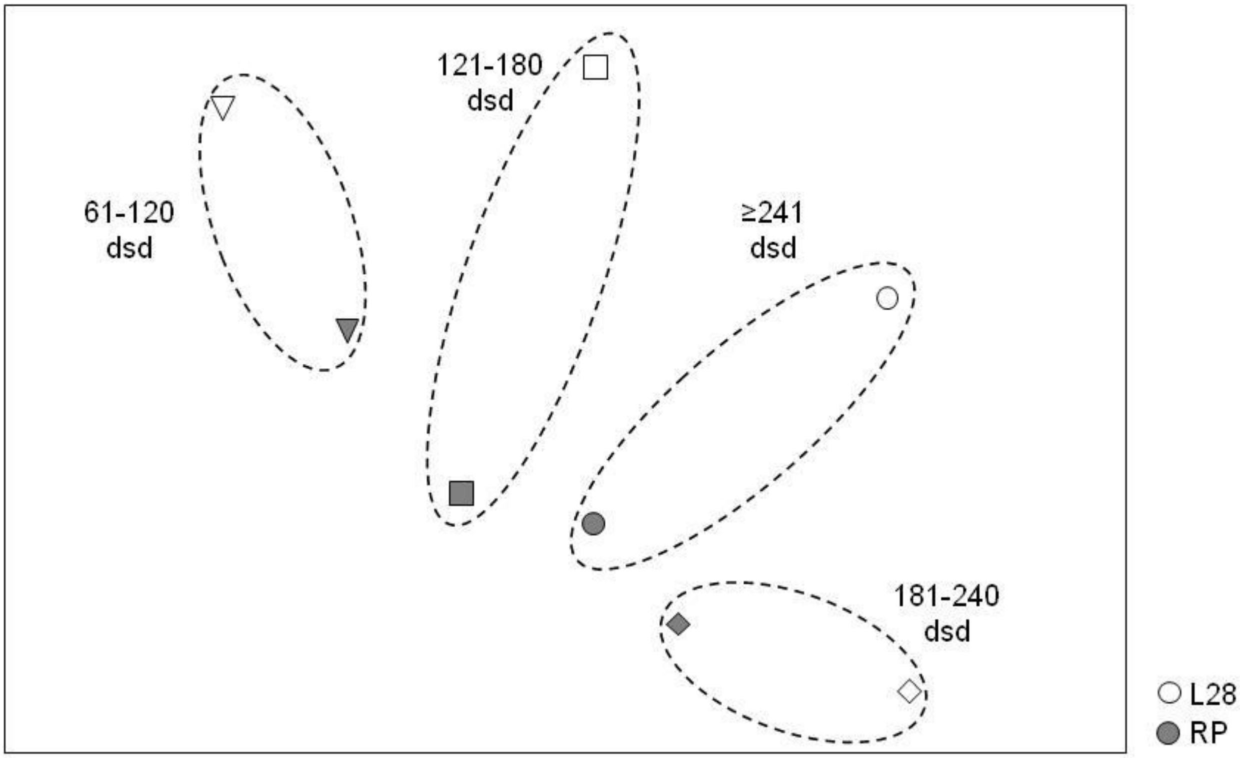

Figure 24. Non-metric multidimensional scaling plot of aquatic fauna communities throughout the hydrologic year in the study year with a significant dry-season reversal (stress=0.08).

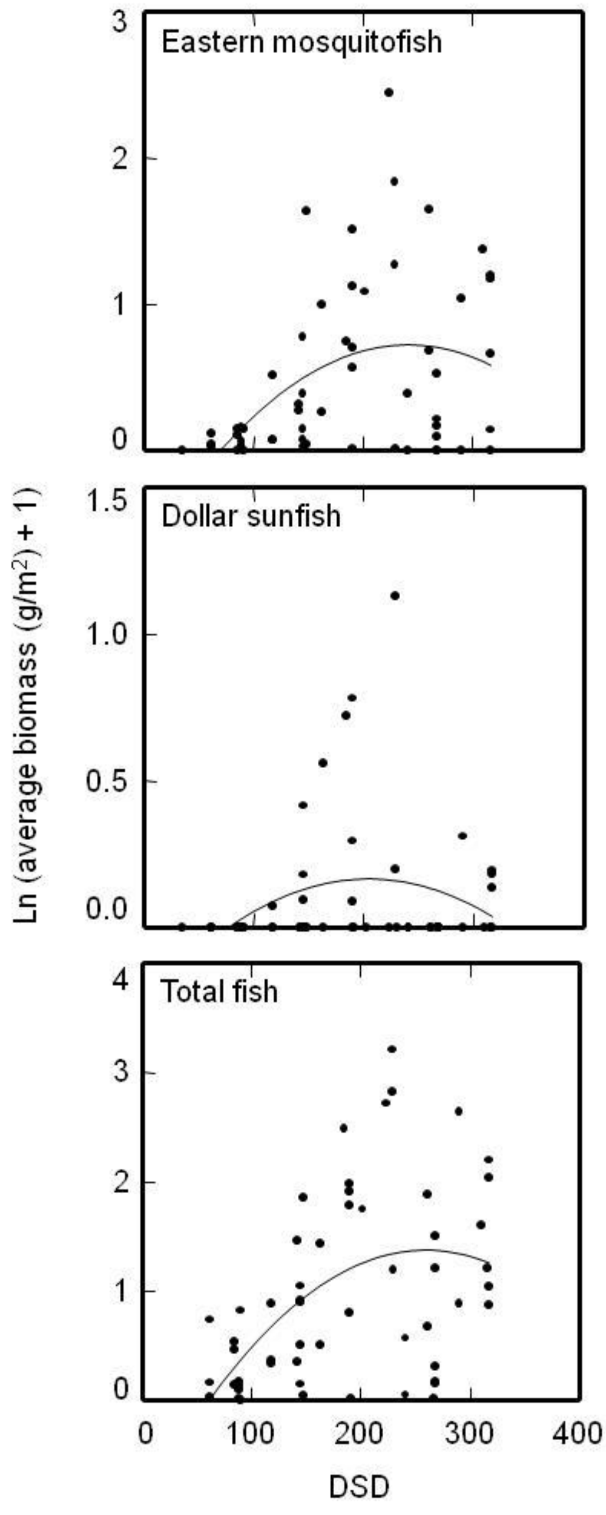

Figure 25. Variation in the average biomass (g wet weight/m^2) of eastern mosquitofish (*Gambusia holbrooki*), dollar sunfish (*Lepomis marginatus*) and total fish with hydroperiod (DSD=days since dry) at study sites in Big Cypress National Preserve during the dry season reversal year.

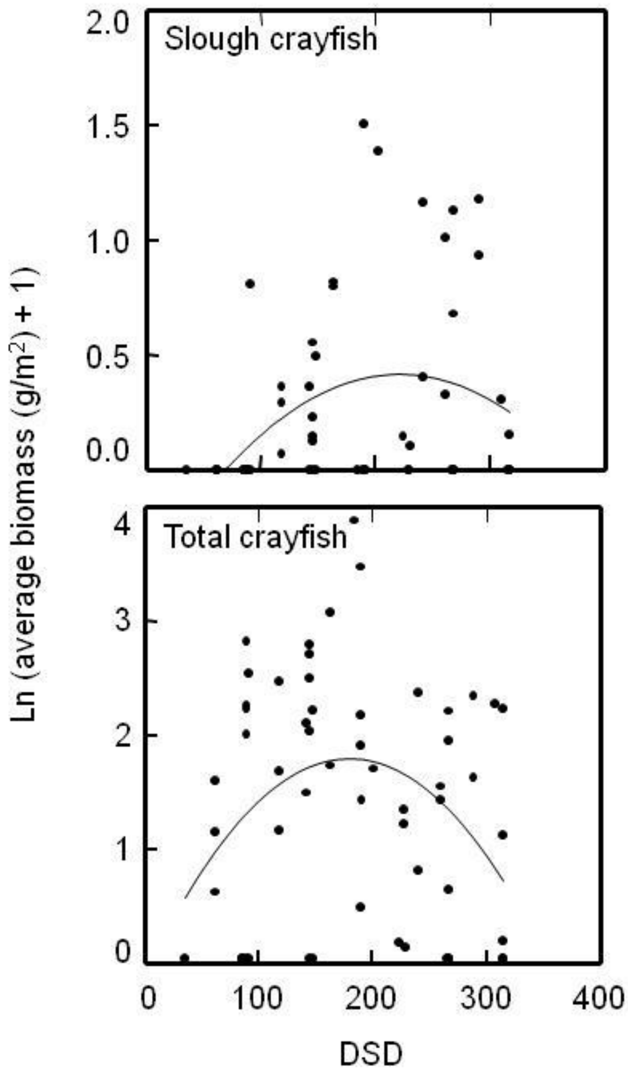

Figure 26. Variation in the average biomass (g wet weight/m^2) of slough crayfish (*Procambarus fallax*) and total crayfish with hydroperiod (DSD=days since dry) at study sites in Big Cypress National Preserve during a the dry-season reversal year.

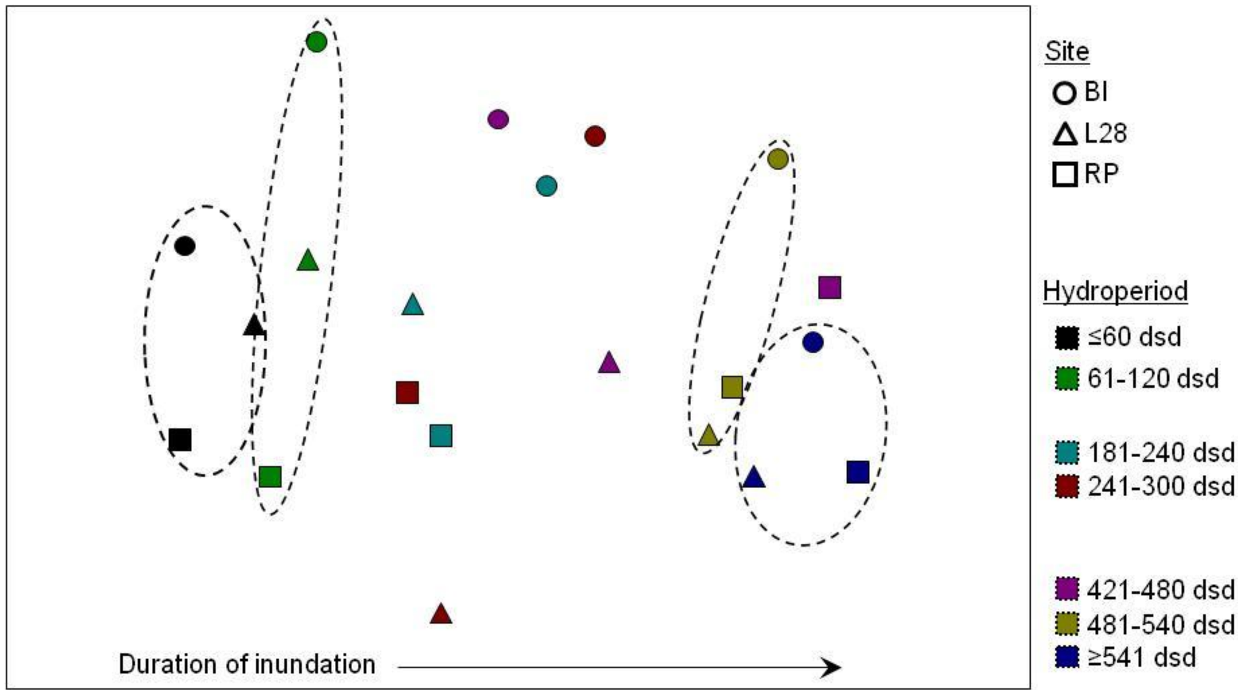

Figure 27. Non-metric multidimensional scaling plot of aquatic fauna communities throughout the hydrologic year with no dry-season recession and the subsequent year (stress=0.14).

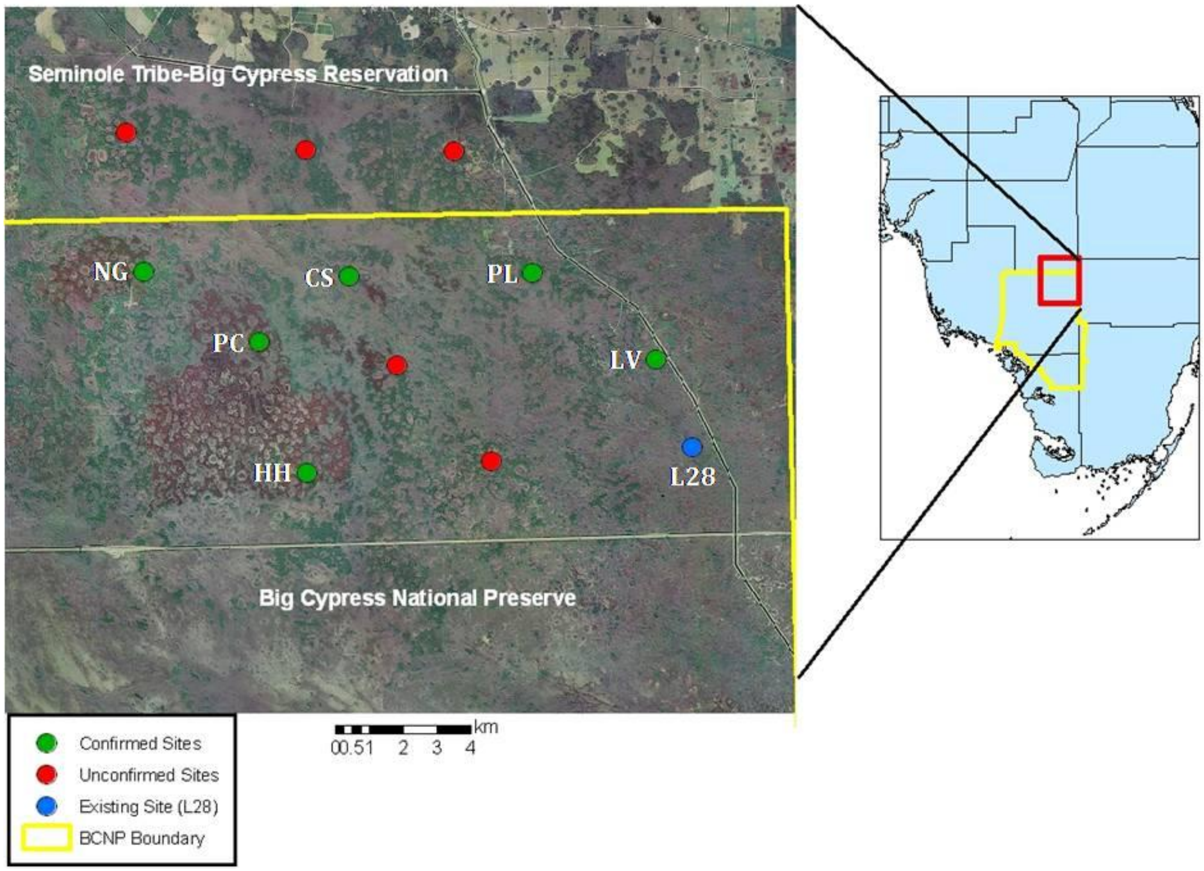

Figure 28. Proposed network of sampling sites in northeastern Big Cypress National Preserve (BCNP Addition Lands). The sentinel site (L28) included in this report would be joined by 8 additional sites in BCNP and 3 sites on the Seminole Tribe of Florida Big Cypress Reservation.

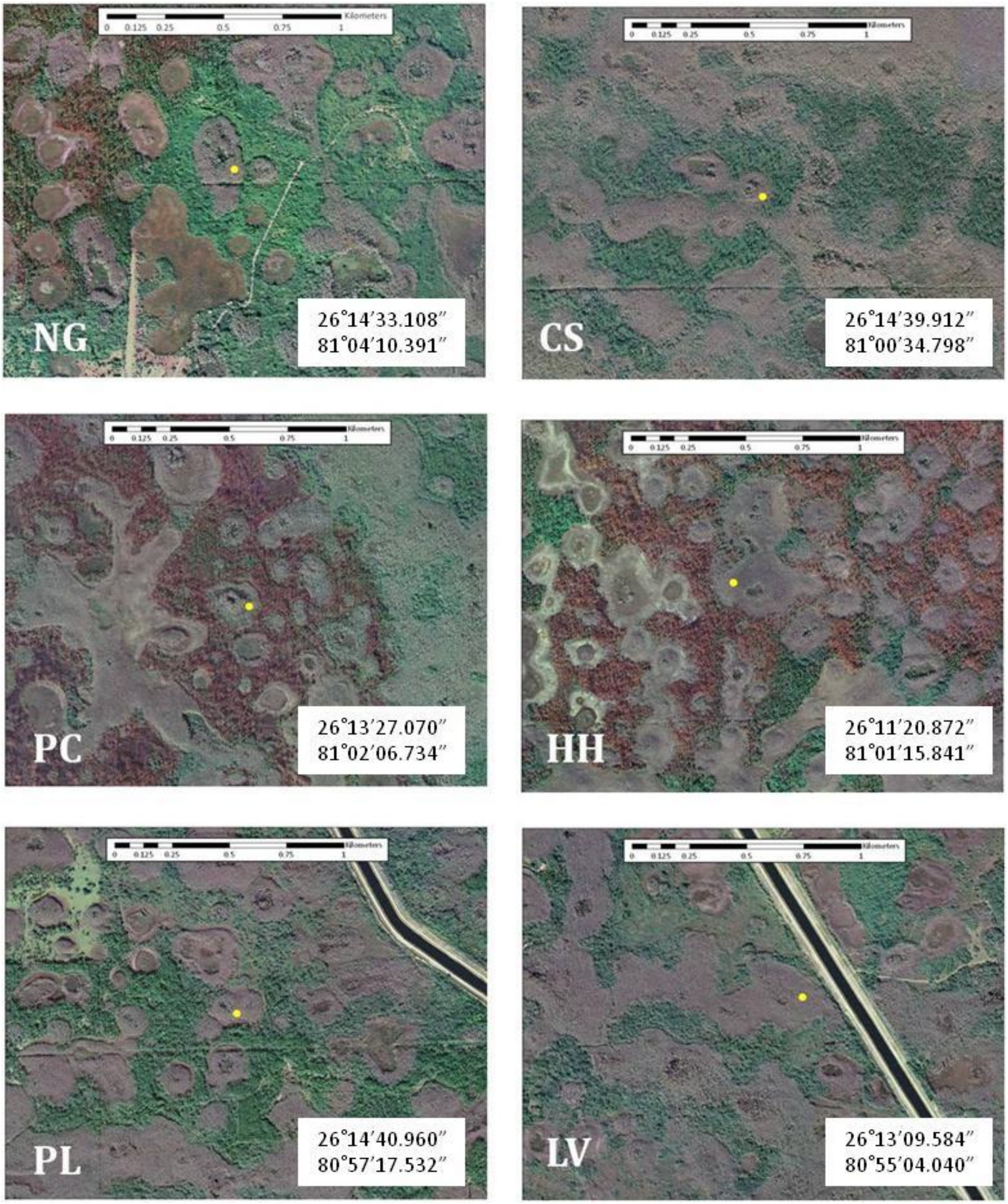

Figure 29. Aerial images of the six new sites established in northeastern BCNP: Nobles Grade (NG), Panther Camp (PC), Hunter's Hideaway (HH), Cowbell Strand (CS), Pipeline (PL) and Looneyville (LV).

Local-scale habitat use by small-bodied fishes in a cypress-dominated forested wetland

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Introduction

The importance of habitat structure for fishes has been well documented. In aquatic systems, structurally-complex habitats often support greater abundance of fishes (Jordan et al. 1996a; Horan et al. 2000; Almany 2003; Dibble and Pelicice 2010) and macroinvertebrates (e.g., Crowder and Cooper 1982; Jordan et al. 1996b; Ferreiro et al. 2011) than habitats with little or no structure. Habitat complexity can also indirectly influence both bottom-up and top-down trophic interactions. Complex habitats often have a greater diversity or abundance of food resources (e.g., Crowder and Cooper 1982; Harmon et al. 1986; Rozas and Odum 1988). Increased habitat complexity has also been demonstrated to impact predator-prey interactions by increasing prey refuge use (e.g., Holbrook and Schmitt 1988; Jordan et al. 1996a; Chick and McIvor 1997; Warfe and Barmuta 2004) and decreasing predator effectiveness (e.g., Nelson and Bonsdorff 1990; Jordan et al. 1996b; Lindholm et al. 1999).

Freshwater forested wetlands make up nearly half of wetlands in the contiguous United States (Dahl 2006). These wetlands often provide a high degree of habitat complexity with an abundance of treefall and a broad range of understory vegetation, dependent upon the degree of light attenuation, nutrient status and hydroperiod (Mitsch and Gosselink 2000). Primary productivity is highest in forested wetlands with pulsing hydroperiod (e.g., Mitsch and Ewel 1979; Brown 1981; Conner et al. 1981; Mitsch et al. 1991). The ecology of aquatic secondary producers (fish, macroinvertebrates, amphibians) in these wetlands, however, is poorly described relative to that of non-forested wetlands (Wharton et al. 1981; Hoover and Killgore 1998; Duever 2005).

The Greater Everglades ecosystem (southern Florida, USA) contains thousands of hectares of freshwater wetlands. In this rainfall-driven, seasonally-pulsed system, annual declines in surface water concentrate small-bodied fishes in depressions, ponds and sloughs. The distribution, abundance and availability of these prey species is particularly important to foraging wading birds (especially tactile-feeders) whose nest success depends on high-density concentrations of fish and crustacean prey (Bancroft et al. 1994; Frederick and Spalding 1994; Gawlik 2002). While the influence of hydrology on small freshwater fish distribution and movement has been well-documented in graminoid marshes (Loftus and Eklund 1994; DeAngelis et al. 2005; Ruetz et al. 2005; DeAngelis et al. 2010) and mangrove creeks (Green et al. 2006; Rehage and Loftus 2007), such studies in the cypress forest landscape of the western Everglades are lacking (Duever 2005).

In this paper, we describe the local (dome-scale) habitat use of small-bodied fishes in the cypress forest landscape of the western Everglades. Better understanding the distribution of small fish in this ecosystem will provide insight into their life-history behaviors, predator-prey interactions and habitat preference, as well as their availability for foraging wading birds.

Methods

Study Design

Sampling was conducted at 8 ~1-ha sites in Big Cypress National Preserve (Florida, USA) (Figure 1). These sites were established as part of a long-term monitoring effort and represented a range of forested wetland habitats, including cypress (domes and strands) and mixed swamp forest (Duever et al. 1986). Sites were generally located in the most consistently inundated forested habitats in the areas of the Preserve, and local topography resulted in each site spanning a full range of hydroperiods (from continuously- to occasionally-inundated).

Sampling effort at each site was stratified into sampling zones: long-hydroperiod forest (LHF; generally inundated >10mo/yr), intermediate-hydroperiod forest (IHF; generally inundated ~8 mo/yr), and wet prairie (WP; generally inundated <6 mo/yr) (Figure 2). Canopy in LHF was comprised mainly of pond apple (*Annona glabra*), bald cypress (*Taxodium distichum*) and/or pop ash (*Fraxinus caroliniana*). Canopy in IHF was comprised mainly of bald cypress, but also consisted of red maple (*Acer rubrum*), and/or cabbage palm (*Sabal palmetto*). The only large woody vegetation in WP was bald cypress (height < 3 m; very small specimens were occasionally included in throw trap samples). All sites contained LHF and IHF zones, but only 3 sites contained WP.

Sampling Methods

Sampling was conducted annually from 2005 to 2010 during the peak of South Florida's wet season (October-December). While the intent of sampling during this time was to ensure a maximum extent of inundation, WP at one site was only inundated during sampling visits 2 of 6 study years (it was inundated earlier in the year and dried). All other habitat zones at each site were inundated at the time of sampling.

Triplicate 1-m^2 throw trap (2-mm mesh) samples were haphazardly collected from each sampling zone within each site using standard methods (Jordan et al. 1997). Prior to fauna sampling, vegetation in each trap was identified to species (when possible) and quantified. Emergent stems were enumerated and summed, while percent cover was estimated for each submerged and floating taxon. Aquatic fauna were cleared from traps using a bar seine that spanned the width of the trap, passing it though until no fauna were collected for three consecutive passes. Any remaining fauna were collected using two sweep nets (1-mm mesh and 5-mm mesh), alternating turns until each had five consecutive empty sweeps. Fauna were preserved in the field; in the laboratory all fishes were identified to species and enumerated and wet weight and standard length (SL) were recorded. While all aquatic fauna were collected and processed, only fish data are presented.

Data Analyses

Because eastern mosquitofish (*Gambusia holbrooki*) are so abundant and ubiquitous in this system (Kushlan and Lodge 1974), community analyses excluded them. We created a habitat structure metric that incorporates both stem counts of emergent taxa and percent coverage of submerged and floating taxa. Stem counts per trap (1 m^2) were transformed into a percent coverage estimate by dividing the total number of stems per trap by 200 (200 stems/m^2 was considered 100% coverage, as the highest stem density observed was 172 stems/m^2). This estimated stem coverage was added to the percent coverage of floating and submerged vegetation for an estimate of total percent coverage of vegetation (%stem coverage + %floating/submerged coverage = %total vegetation coverage). This habitat structure variable was transformed (arcsin(√%habitat structure)), prior to analyses.

[insert details on data analysis]

Results

Spatial Variation in Habitat Structure

Multivariate analysis indicated the structure of macrophyte communities varied among sampling zones (P=0.001, Global R=0.204). IHF and LHF were similar (pairwise P=0.246), characterized by Chapman's arrowhead (*Sagittaria graminea* var. *chapmanii*), lemon bacopa (*Bacopa caroliniana*), and creeping primrose willow (*Ludwigia repens*) (cumulative similarity=96.19% and 99.04%, respectively). WP macrophyte communities, however, varied from both IHF and LHF (pairwise: P=0.001, Global R=0.345 and P=0.001, Global R=0.438, respectively), characterized by beaksedge (*Rhynchospora* spp.), spikerush (*Eleocharis* spp.), bald cypress (*Taxodium distichum*) and water toothleaf (*Stillingia aquatica*) (cumulative similarity=91.04%).

ANOVA indicated variation in macrophyte %cover among sampling zones ($F_{2,99}=12.0$, $P<0.001$, $R^2=0.195$). Average %cover in WP and IHF was similar (Tukey P=0.920; $\bar{X}_{WP}=57.1\pm5.6$, $\bar{X}_{IHF}=61.3\pm7.2$). Macrophyte %cover in IHF and WP was 2.6X and 2.4X higher than LHF ($\bar{X}_{LHF}=23.5\pm5.4$), respectively (Tukey P<0.001, P=0.002).

Spatial Variation in Fish Community

A total of 2,458 fish (24 species) were collected (Table 1). One large (8.78 cm SL) spotted sunfish (*Lepomis punctatus*) was omitted from the dataset, as the sampling method has been shown to be most effective for fish <8 cm SL (Kushlan 1981; Jacobsen and Kushlan 1987; Chick et al. 1992; Jordan et al. 1997; Rozas and Minello 1997).

Community structure of small fishes did not vary among sampling zones (P=0.351, Global R=0.012). In addition to eastern mosquitofish, communities were characterized primarily by least killifish (*Heterandria formosa*) and bluefin killifish (*Lucania goodei*) (cumulative similarities ≥66%). Total density of small fishes varied among zones (Table 2), reflecting density in IHF was 2.3X that of LHF (Tukey P=0.002)(Figure 3). Density of 2 of 5 common species varied among zones (Table 2). Eastern mosquitofish density was 2.5X higher in IHF than in LHF (Tukey P=0.005); pairwise comparisons of bluespotted sunfish (*Enneacanthus gloriosus*) density showed no apparent inter-zone variation (all Tukey P>0.05). Total fish density excluding mosquitofish also varied among zones ($F_{2,99}$=3.62, P=0.030, R^2=0.068), as density in IHF was 1.5X that of LHF (Tukey P=0.023).

Community structure of medium fishes varied among sampling zones (P=0.001, Global R=0.112; all pairwise P<0.05). In addition to eastern mosquitofish, WP was characterized by bluefin killifish, flagfish (*Jordanella floridae*), and golden topminnow (*Fundulus chrysotus*) (cumulative similarity=97.4%), IHF was characterized by flagfish, bluefin killifish, dollar sunfish (*Lepomis marginatus*), and bluespotted sunfish (cumulative similarity=88.3%), and LHF was characterized by brook silverside (*Labidesthes sicculus*), bluefin killifish, dollar sunfish, and flagfish (cumulative similarity=91.4%). Total density of medium fishes varied among sampling zones (Table 2), reflecting 2.4X higher density in IHF than WP (Tukey P<0.001) or LHF (Tukey P=0.001)(Figure 3). Density of 3 of 10 common species varied among zones (Table 2). Densities of bluespotted sunfish and mosquitofish were higher (xx and xx, respectively) in IHF than WP of LHF (all Tukey P<0.05).

[NEED TO ADD CORRELATIONS: FISH v HAB STR, FISH v WATER DEPTH]

Discussion

**Need to bring in timing of Big Cypress drying (earlier than Greater Everglades) and how the timing of fish availability is important in this system to feed wading birds preparing to nest --- emphasize importance of Wood Stork*

These small bodied fishes are the main forage items for predators in the Big Cypress region (Crozier and Gawlik 2003; DeAngelis et al. 2005), including nesting wading birds and the endangered Wood Stork. Historically BCR was home to one of the largest nesting colonies of wood storks in the nation (Duever 2005), and it is hypothesized that a diminishing of the fish forage base is the main reason for nesting wading bird decline in this region (Ogden and Davis 1994). One of the main goals of Everglades restoration is to return the wetland fish stocks to what they were before development. In order to accomplish this it will be imperative to know what types of habitats support large numbers of wetland fishes.

Since %coverage of macrophyte communities were higher in IHF than in LHF it is not surprising that higher abundances of both small and medium fishes were found in IHF (2.3X, 2.4X respectively), due to the positive correlation between % cover of macrophytes and fish abundance (small P= 0.004, medium P= 0.001), and because habitats with higher structural complexity have been shown to support greater abundance of fishes (Jordan et al. 1996a; Horan et al. 2000; Almany 2003; Dibble and Pelicice 2010). However, %coverare of macrophytes did not differ between IHF and WP, while medium fishes showed a significantly higher abundance (2.4X) in IHF than WP and small fishes did not. Even though the total % cover is similar between the IHF and WP, the macrophyte community of WP consists mainly of emergent stems. These stems provide habitat structure lower in complexity than the macrophytes (lemon bacopa and creeping primrose willow) of the IHF, which could provide greater refuge from predation (e.g., Holbrook and Schmitt 1988; Jordan et al. 1996a; Chick and McIvor 1997; Warfe and Barmuta 2004) or better food resources (e.g., Crowder and Cooper 1982; Harmon et al. 1986; Rozas and Odum 1988) for the medium fishes. One explanation for the insignificant difference between the WP and the IHF for small fishes is the small fish class could be utilizing the WP more in order to avoid the medium fishes and lower predation pressure. It has been found that average sized adult mosquitofish (SL = 22mm) can consume juveniles up to 9mm in length (Taylor et al. 2001).

Knowing what types of habitats support large abundances of wetland fishes will be of the utmost importance when restoring or creating wetlands. Mitigation banking is one of the fastest growing, most lucrative sectors in the green economy resulting in 2.9 billion dollars of expenditures annually (BenDor and Riggsbee 2011) and a main driving force behind wetland restoration and creation. In this process developers operating in ecologically significant areas such as wetlands, try to offset their impact on the environment by purchasing restoration credits, which are in turn used to restore or create wetlands in other areas (BenDor et al. 2009). In 2005 there were an estimated 400 operating and 200 planned mitigation banks in the United States (BenDor and Riggsbee 2011). With the amount of effort and money being pumped into wetland restoration and creation it is imperative to create wetlands which provide complex habitat and resulting large abundances of marsh fishes, since they constitute the main forage base for top predators in this ecosystem (Crozier and Gawlik 2003; DeAngelis et al. 2005).

Woody debris is not quantified in this study

Implications for creation of wetlands—important of different habitat types for protection/restoration

References

Almany GR (2003) Differential effects of habitat complexity, predators and competitors on abundance of juvenile and adult coral reef fishes. Oecologia 141: 105-113

Bancroft GT, Strong AM, Sawicki RJ, Hoffman W, Jewell SD (1994) Relationships among wading bird foraging patterns, colony locations, and hydrology in the Everglades. In: Davis SM, Ogden JC (eds.) Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Boca Raton, FL, pp 615-657

BenDor T, Sholtes J, Doyle MW (2009) Landscape characteristics of a stream and wetland mitigation banking program. Ecological Applications 19: 2078-2092

BenDor TK, Riggsbee JA (2011) A survey of entrepreneurial risk in U.S. wetland and stream compensatory mitigation markets. Environmental Science and Policy 14: 301-314

Brown S (1981) A comparison of the structure, primary productivity, and transpiration of cypress ecosystems in Florida. Ecological Monographs 51: 403-427

Chick JH, Jordan CF, Smith JP, McIvor CC (1992) A comparison of four enclosure traps and methods used to sample fishes in aquatic macrophytes. Journal of Freshwater Ecology 7: 353-361

Chick JH, McIvor CC (1997) Habitat selection by three littoral zone fishes: effects of predation pressure, plant density and macrophyte type. Ecology of Freshwater Fish 6: 27-35

Conner WH, Gosselink JG, Parrondo RT (1981) Comparison of the vegetation of three Louisiana swamp sites with different flooding regimes. American Journal of Botany 68: 320-331

Crowder LB, Cooper WE (1982) Habitat structural complexity and the interaction between bluegills and their prey. Ecology 63: 1802-1813

Crozier GE, Gawlik DE (2003) Wading bird nesting effort as an index to wetland ecosystem integrity. Waterbirds 26: 303-324

Dahl TE (2006) Status and trends of wetlands in the conterminous United States 1998 to 2004. U.S. Department of the Interior; Fish and Wildlife Service, Washington, D.C.

DeAngelis DL, Trexler JC, Cosner C, Obaza A, Jopp F (2010) Fish population dynamics in a seasonally varying wetland. Ecological Modelling 221: 1131-1137

DeAngelis DL, Trexler JC, Loftus WF (2005) Life history trade-offs and community dynamics of small fishes in a seasonally pulsed wetland. Canadian Journal of Fisheries and Aquatic Sciences 62: 781-790

Dibble ED, Pelicice FM (2010) Influence of aquatic plant-specific habitat on an assemblage of small neotropical floodplain fishes Ecology of Freshwater Fish 19: 381-389

Duever MJ (2005) Big Cypress Regional Ecosystem conceptual ecological model. Wetlands 25: 843-853

Duever MJ, Carlson JE, Meeder JF, Duever LC, Gunderson LH, Riopelle LA, Alexander TR, Myers RL, Spangler DP (1986) The Big Cypress National Preserve. National Audubon Society, New York, NY

Ferreiro N, Feijoo C, Giorgi A, Leggieri L (2011) Effects of macrophyte heterogeneity and food availability on structural parameters of the macroinvertebrate community in a Pampean stream. Hydrobiologia 664: 199-211

Frederick PC, Spalding MG (1994) Factors affecting reproductive success of wading birds (Ciconiformes) in the Everglades ecosystem. In: Davis SM, Ogden JC (eds.) Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Boca Raton, FL, pp 659-691

Gawlik DE (2002) The effects of prey availability on the numerical response of wading birds. Ecological Monographs 72: 329-346

Green DPJ, Trexler JC, Lorenz JJ, McIvor CC, Philippi T (2006) Spatial patterns of fish communities along two estuarine gradients in southern Florida. Hydrobiologia 569: 387-399

Harmon ME, Franklin JF, Swanson FJ, Sollins P, Gregory SV, Lattin JD, Anderson NH, Cline SP, Aumen NG, Sedell JR, Lienkaemper GW, Cromack K, Jr., Cummins KW (1986) Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research 15: 133-302

Holbrook SJ, Schmitt RJ (1988) The combined effects of predation risk and food reward on patch selection. Ecology 69: 125-134

Hoover JJ, Killgore KJ (1998) Fish communities. In: Messina MG, Conner WH (eds.) Southern Forested Wetlands: Ecology and Management. Lewis Publishers, Boca Raton, FL, pp 237-260

Horan DL, Kershner JL, Hawkins CP, Crowl TA (2000) Effects of habitat area and complexity on Colorado River cutthroat trout density in Uinta Mountain streams. Transactions of the American Fisheries Society 129: 1250-1263

Jacobsen T, Kushlan JA (1987) Sources of sampling bias in enclosure fish trapping: effects of estimates of density and diversity. Fisheries Research 5: 401-412

Jordan CF, Coyne S, Trexler JC (1997) Sampling fishes in heavily vegetated habitats: the effects of habitat structure on sampling characteristics of the 1-m^2 throw trap. Transactions of the American Fisheries Society 126: 1012-1020

Jordan F, Bartolini M, Nelson C, Patterson PE, Soulen HL (1996a) Risk of predation affects habitat selection by the pinfish *Lagodon rhomboides* (Linnaeus). Journal of Experimental Marine Biology and Ecology 208: 45-56

Jordan F, DeLeon CJ, McCreary AC (1996b) Predation, habitat complexity, and distribution of the crayfish Procambarus alleni within a wetland habitat mosaic. Wetlands 16: 452-457

Kushlan JA (1981) Sampling characteristics of enclosure fish traps. Transactions of the American Fisheries Society 110: 557-562

Kushlan JA, Lodge TE (1974) Ecological and distributional notes on the freshwater fish of southern Florida. Florida Scientist 37: 110-128

Lindholm JB, Auster PJ, Kaufman LS (1999) Habitat-mediated survivorship of juvenile (0-year) Atlantic cod Gadus morhua. Marine Ecology Progress Series 180: 247-255

Loftus WF, Eklund A-M (1994) Long-term dynamics of an Everglades small-fish assemblage. In: Davis SM, Ogden JC (eds.) Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Boca Raton, FL, pp 461-483

Mitsch WJ, Ewel KC (1979) Comparative biomass and growth of cypress in Florida wetlands. American Midland Naturalist 101: 417-426

Mitsch WJ, Gosselink JG (2000) Wetlands Third edition. John Wiley & Sons, Inc., New York

Mitsch WJ, Taylor JR, Benson KB (1991) Estimating primary productivity of forested wetland communities in different hydrologic landscapes. Landscape Ecology 5: 75-92

Nelson WG, Bonsdorff E (1990) Fish predation and habitat complexity: are complexity thresholds real? Journal of Experimental Marine Biology and Ecology 141: 183-194

Ogden JC, Davis SM (1994) A comparison of wading bird nesting colony dynamics (1931-1946 and 1974-1989) as an indication of ecosystem conditions in the southern Everglades. In: Ogden JC, Davis SM (eds.) Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Boca Raton, FL, pp 533-569

Rehage JS, Loftus WF (2007) Seasonal fish community variation in headwater mangrove creeks in the southwestern Everglades: an examination of their role as dry-down refuges. Bulletin of Marine Science 80: 625-645

Rozas LP, Minello TJ (1997) Estimating densities of small fishes and decapod crustaceans in shallow estuarine habitats: a review of sampling design with focus on gear selection. Estuaries 20: 199-213

Rozas LP, Odum WE (1988) Occupation of submerged aquatic vegetation by fishes: testing the roles of food and refuge. Oecologia 77: 101-106

Ruetz CR, III, Trexler JC, Jordan F, Loftus WF, Perry SA (2005) Population dynamics of wetland fishes: spatio-temporal patterns synchronized by hydrological disturbance? Journal of Animal Ecology 74: 322-332

Taylor RC, Trexler JC, Loftus WF (2001) Separating the effects of intra- and interspecific age-structured interactions in an experimental fish assemblage. Oecologia 127: 143-152

Warfe DM, Barmuta LA (2004) Habitat structural complexity mediates the foraging success of multiple predator species. Oecologia 141: 171-178

Wharton CH, Lambou VW, Newsom J, Winger PV, Gaddy LL, Mancke R (1981) The fauna of bottomland hardwoods in Southeastern United States. In: Clark JR, Benforado J (eds.) Wetlands of Bottomland Hardwood Forests. Elsevier Scientific Publishing Company, Washington, D.C., pp 87-160

Table 1. Number collected (n), percent incidence (%I), and relative abundance (%RA) of small (<2 cm SL) and medium (2-8 cm SL) fish captured in 1-m^2 throw trap samples.

	Small Fish			Medium Fish		
Species	n	%I	%RA	n	%I	%RA
Black acara	4	2.0	0.3	21	12.7	2.1
Bluefin killifish	1	1.0	0.1	118	32.4	11.6
Bluegill sunfish	190	35.3	13.2	2	2.0	0.2
Bluespotted sunfish	11	6.9	0.8	23	12.7	2.3
Brook silverside	9	8.8	0.6	73	15.7	7.2
Brown bullhead	0	0	0	1	1.0	0.1
Brown hoplo	0	0	0	4	2.9	0.4
Dollar sunfish	16	8.8	1.1	49	26.5	4.8
Eastern mosquitofish	663	80.4	46.1	488	76.5	47.8
Everglades pygmy sunfish	19	8.8	1.3	5	4.9	0.5
Flagfish	57	18.6	4.0	86	29.4	8.4
Golden shiner	0	0	0	3	2.0	0.3

Golden topminnow	4	3.9	0.3	15	11.8	1.5
Least killifish	393	36.3	27.3	4	3.9	0.4
Marsh killifish	0	0	0	8	3.9	0.8
Mayan cichlid	0	0	0	12	8.8	1.2
Pike killifish	0	0	0	1	1.0	0.1
Redear sunfish	1	1.0	0.1	2	2.0	0.2
Sailfin molly	5	4.9	0.3	17	4.9	1.7
Spotted sunfish	38	12.7	2.6	32	15.7	3.1
Spotted tilapia	2	2.0	0.1	3	2.9	0.3
Swamp darter	0	0	0	5	4.9	0.5
Unidentified fish	5	3.9	0.3	0	0	0
Unidentified sunfish	5	4.9	0.3	1	1.0	0.1
Warmouth	14	5.9	1.0	45	19.6	4.4
Yellow bullhead	0	0	0	2	2.0	0.2
Total fish	1437			1020		

Table 2. *Summary of variation by zone…*

	$F_{2,99}$	P	R^2
Small fish			
Bluespotted sunfish	3.036	0.053	0.058
Eastern mosquitofish	5.240	0.007	0.096
Total fish	5.926	0.004	0.195
Medium fish			
Bluespotted sunfish	6.551	0.002	0.117
Eastern mosquitofish	9.051	<0.001	0.155
Golden topminnow	4.253	0.017	0.079
Total fish	11.119	<0.001	0.183

Figure 1.

Figure 2.

Figure 3. Arithmetic means (±SE) of total density (no./m^2) of small (SL < 2 cm) and medium fish (SL 2-8cm) collected in long-hydroperiod forests (LHF; generally inundated >10mo/yr), intermediate-hydroperiod forests (IHF; generally inundated ~8 mo/yr), and wet prairies (WP; generally inundated <6 mo/yr).

Figure 4.

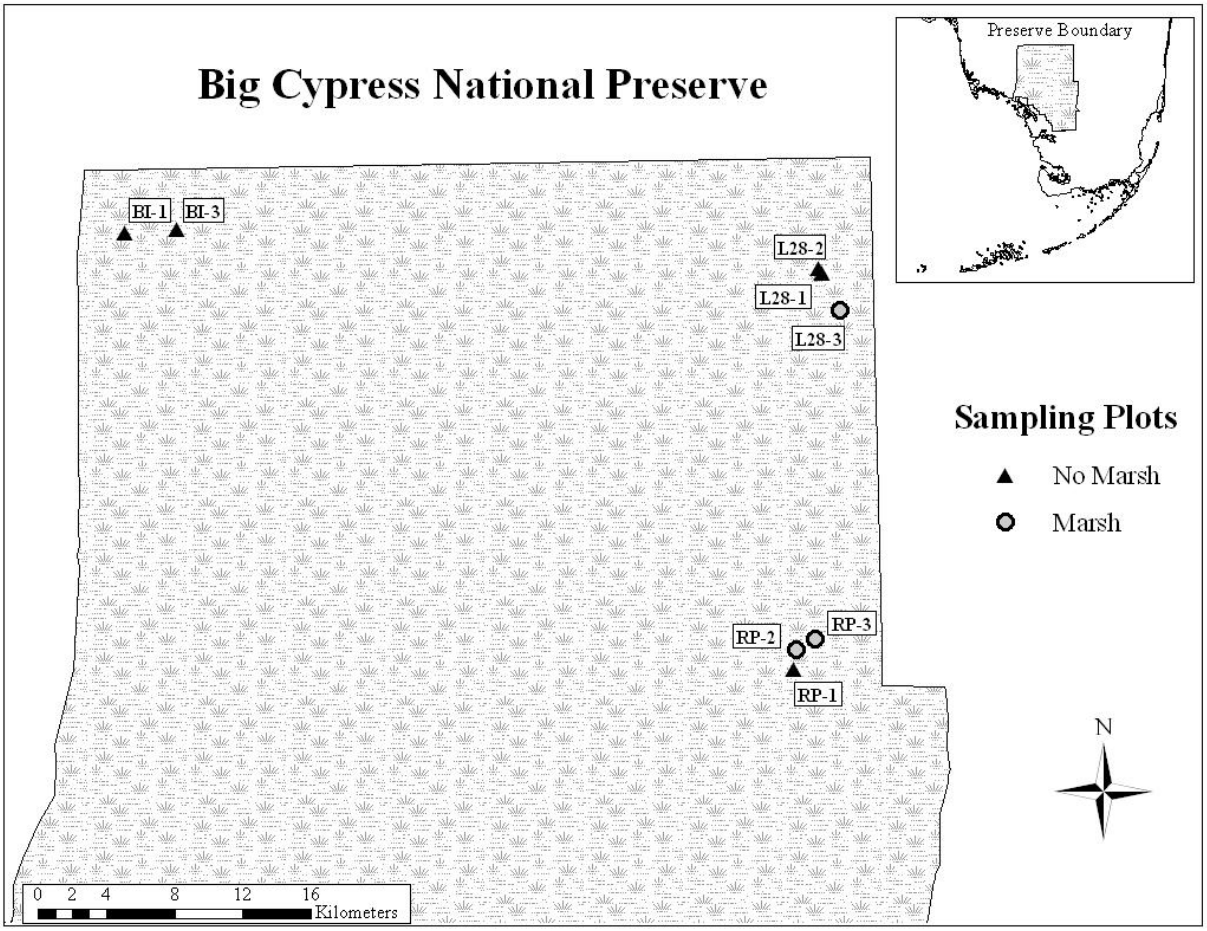
Big Cypress National Preserve
Preserve Boundary
BI-1
BI-3
L28-2
L28-1
L28-3
RP-2
RP-3
RP-1
Sampling Plots
No Marsh
Marsh
N
0 2 4 8 12 16
Kilometers

Figure 1.

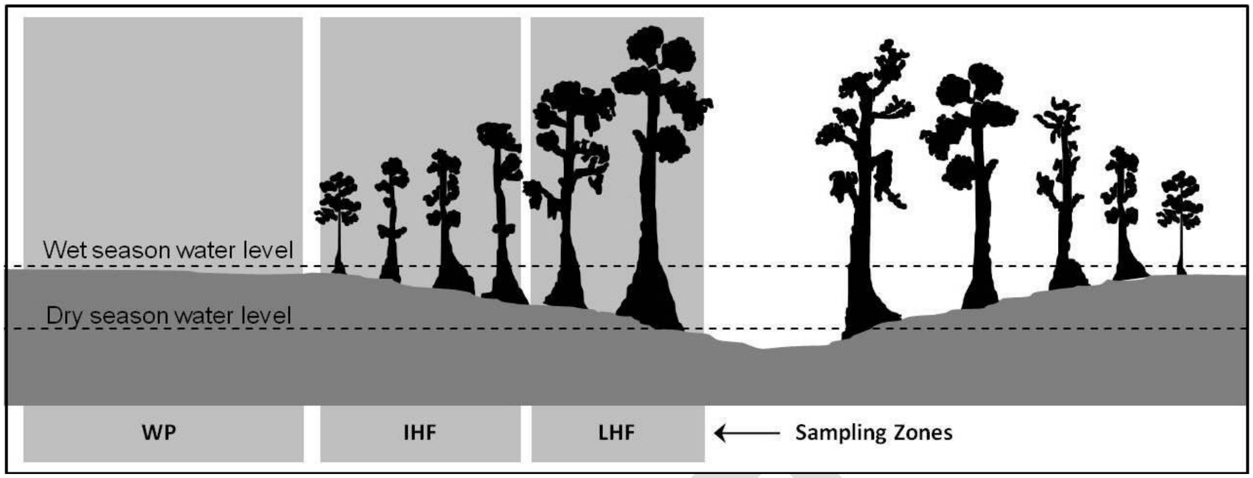
Wet season water level
Dry season water level
WP
IHF
LHF
Sampling Zones

Figure 2.

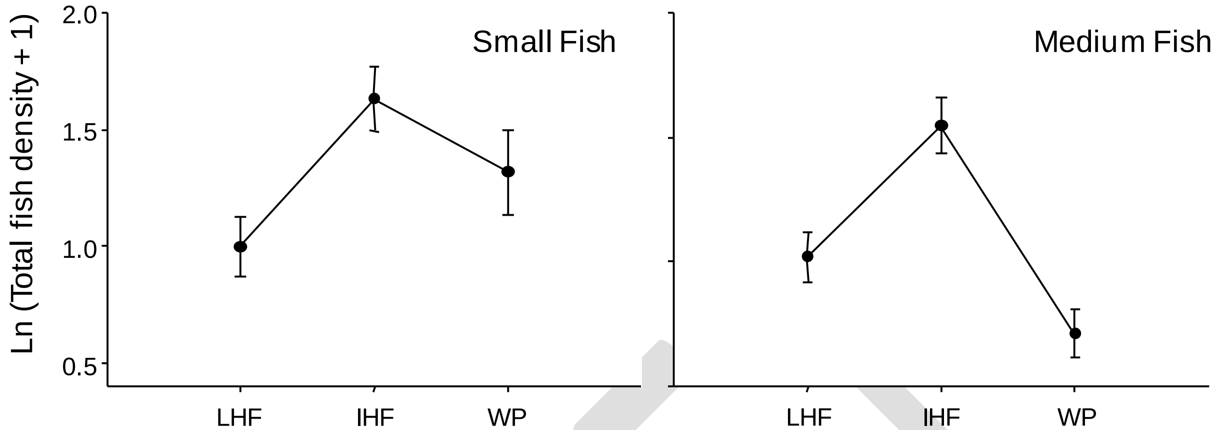
Ln (Total fish density + 1)
2.0
1.5
1.0
0.5
Small Fish
Medium Fish
LHF
IHF
WP
LHF
IHF
WP

Figure 3.

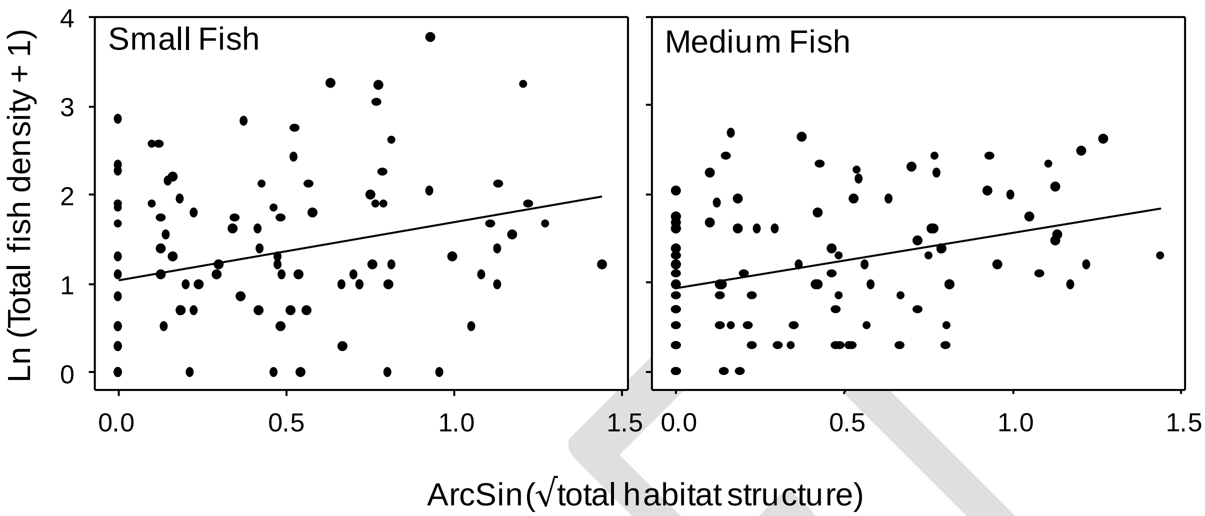
Small Fish
Medium Fish
Ln (Total fish density + 1)
ArcSin(√total habitat structure)
0
1
2
3
4
0.0
0.5
1.0
1.5

Figure 4.