Adaptive management and monitoring for restoration and faunal recolonization of Tallapoosa River shoal habitats.

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Executive Summary:

The widespread fragmentation and alteration of riverine habitat by dams require management options that both address restoration and conservation of native aquatic biota and fisheries and increase knowledge of the relations between faunal processes and flow variability. Since 2005, flow management changes from R.L. Harris Dam on the Tallapoosa River, Alabama, have been implemented as part of an adaptive management project to determine optimal flows for multiple competing management objectives. The main objective of the current project was to evaluate the effects of these management flows on the recovery of shoal-dwelling species of greatest conservation need (GCN) and the persistence of functional shoal habitats in the Tallapoosa River.

Faunal sampling was conducted in spring (May-June) and fall (September-November) 2005-2009 using prepositioned area electrofishers (PAEs). Specific microhabitat variables (depth, velocity, percent vegetation, and substrata composition) were measured for each PAE sample. Index of biotic integrity (IBI) was calculated for spring and summer samples in each year for each site. Crayfish catch data were examined for differences in catch per effort, size distribution, and species composition for differences between regulated and unregulated sites using non-parametric K-S tests and paired t-tests.

Estimates of detection, occupancy, extinction, and colonization were calculated for fourteen selected fish species; estimates of detection and occupancy were calculated for all collected crayfish species. These estimates were calculated using maximum likelihood methods and modeled as a function of measured covariates using the logit link function. Competing models of species dynamics were compared using Akaike’s information criterion (AIC).

To examine reproductive condition, a random subsample of fish from each shoal in each year were examined for presence of viable reproductive organs. Percent mature females was determined for each of nine species as an indicator of reproductive condition. To assess hatch date of Centrarchid sport fish, young-of-year (YOY) redbreast sunfish, spotted bass, and redeye bass were collected

approximately 30, 60, and 90-days after the onset of spawning in 2005 and 2007, and daily ages and hatch dates were estimated from extracted otoliths. Hydrologic data from USGS gage stations were examined against hatch frequencies to determine optimal flow conditions for spawning and subsequent recruitment.

Overall, IBI values were lower among regulated sites; however, IBIs varied widely among sites, within and among river reaches, between seasons, and among years. Nine of the fourteen species examined for species occupancy dynamics had parameters that varied between regulated and unregulated sites. Two of the six GCN fish species, both darters, were apparently unaffected by the impact of Harris Dam; lipstick darter appeared to have a slight positive response to regulation. Occupancy estimates of the remaining three GCN species suggested that these species are either in decline or absent altogether in the regulated reach below Harris Dam. For all crayfish species, detection was a function of habitat variables; vegetation and velocity affected detection positively, while depth had a negative effect on detection.

Proportion of mature female fish varied among years and sites. No mature largescale stoneroller *Campostoma oligolepis* females were observed at any sites or years. Mature female Tallapoosa shiners *Cyprinella gibbsi* and bullhead minnows *Pimephales vigilax* were observed in the unregulated reaches only. There were no significant differences in total length of YOY Centrarchids found among sites. Hatch dates of YOYs were not correlated to prolonged stable flow periods in 2005, but were correlated in 2007, when the majority of hatches occurred during or up to 3 days after periods of stable, low flows. Stable flow periods may provide for greater availability of suitable spawning and juvenile habitat which allows for recruitment to a stage and size where fish can withstand daily fluctuating discharges.

In general, our results indicated that the Tallapoosa River fish and crayfish assemblage varies considerably, not only between the regulated and unregulated river, but also within the unregulated reaches, both between seasons and among years. These results suggest that there is a natural level of variability that should be expected, and even perhaps managed for. Maximizing conservation potential in free-flowing sections of rivers of Alabama will require, at minimum, clear evidence of effects of regulated flow regimes on river biota. An adaptive management approach holds substantial promise for improving management of regulated rivers by allowing managers and scientists to address the uncertainty in predicting and measuring faunal response to flow alterations.

Introduction

Faunal assemblages native to larger rivers now mostly persist in river fragments that are variously affected by upstream dams and reservoirs. This fragmentation and alteration of riverine habitat demand management options that explicitly address restoration and conservation of native aquatic biota and fisheries in flow-altered river reaches downstream from dams. Management options have usually been standard one-time negotiated flow plans without flexibility for change. These management scenarios have hampered our ability to define relations between faunal processes (e.g., recruitment) and flow variability. Freeman et al. 2001 reported that juvenile fish abundance was related to persistence of shallow-fast habitat (i.e., shoal habitat); however, persistence was limited by peaking hydropower operations in the middle Tallapoosa River.

Irwin and Freeman (2002) proposed that an adaptive approach could be used to manage riverine fish faunas in the southeast U.S., and elsewhere. As of Spring 2005, flow management changes were implemented at Harris Dam (http://www.RiverManagement.org), consistent with the approach outlined in Irwin and Freeman (2002). This SWG project addresses restoration and management of a strongly flow-regulated reach of the Tallapoosa River as an experimental system for determining effects of management on shoal dwelling GCN species and their critical habitats.

The Tallapoosa River: a case study for shoal habitat restoration

The proposed regulated study reach, beginning at Harris Dam and terminating 78 km downstream in the headwaters of Martin Reservoir, represents one of the longest and highest quality segments of Piedmont river habitat remaining in the Mobile River drainage (Lydeard and Mayden 1995; Mettee et al. 1996; Neves et al. 1997). Because of the flow management project with the Alabama Power Company, the Tallapoosa River presents an opportunity to determine flow conditions for functional shoal habitat. Findings should be directly transferable to other similarly fragmented, flow-managed rivers that harbor shoal-dwelling GCN species, such as the Coosa and Tennessee Rivers.

Critical Habitat

Disruption of riverine flows from impoundment and river regulation are primary reasons for high levels of imperilment of fishes and mollusks in Alabama. The primary cause of major extinction events for mollusks (mussels and gastropods) was the impoundment of shoal and riffle habitat (Neves et al. 1997). Similarly, 53% of fishes inhabiting medium sized rivers and creeks in the southeast U. S. are in jeopardy (Etnier 1997). This is likely because fish communities in these habitats are dependent on shoals and riffles for at least part of their life history (Etnier 1997). A brief survey of mollusks and fishes listed in Mirarchi et al. 2004, Volume Two indicated that 64% of imperiled species (extirpated, P1 and P2[1]) are potentially shoal dependent. Therefore, restoration

[1] This estimate was derived by including fishes, mussels and snails that had lotic, riffle, moderate or fast current, or shoal habitat listed in their descriptions in Mirarchi et al. (2004). I did not include species from below the fall line (except Gulf sturgeon and shoal bass), other obvious non-shoal dependent fauna (e.g., spring or cave dwellers), or species that occur habitats adjacent to shoal. This hopefully provided a relatively conservative estimate.

of *functional* shoal habitat in unimpounded riverine reaches may provide critical habitat for species of concern, as well as all other species inhabiting the river.

Occupancy of crayfishes may have important implications for stream communities because when present, they typically represent the largest standing biomass of macroinvertebrates (Huryn and Wallace 1987, Rabeni et al. 1995). Crayfishes are considered keystone species because they may directly or indirectly influence populations at multiple trophic levels (Mormot 1995). Multiple species of predatory fishes, birds, and mammals feed on crayfish and trophic complexity is increased by the opportunistic feeding of crayfishes on macrophytes, algae, detritus, macroinvertebrates, amphibians, and fish (Mormot 1995). Crayfishes perform important ecosystem functions such as processing macrophytes and leaf litter into fine particulate organic matter which may be used by other stream organisms (Huryn and Wallace 1987, Whitledge and Rabeni 1997) and influencing sediment movement (Helms and Creed 2005). Therefore, crayfishes are important members shaping lotic communities through influencing ecosystem level processes of energy flow and nutrient transfer. The multi faceted role crayfish serve in system dynamics should make them high priority when setting objectives for stream management (Brewer et al. 2009). The greatest diversity of crayfish in the world is in the southeastern United States where more than 300 species of approximately 540 species worldwide are extant. Estimates of between 1/3 and 1/2 of crayfish species are at risk of serious decline or even extinction (Taylor 2002). Narrow geographical ranges and limited distributions increase species susceptibility to risk of extinction and lack of information on habitat requirements and effects of habitat alteration hamper conservation efforts (Taylor 2002, Jones and Bergey 2007).

The native fish assemblage in the Piedmont section of the Tallapoosa includes at least 57 species (Mettee et al. 1996), including at least five species endemic to the Tallapoosa system. The invertebrate fauna is less well known; however, the finelined pocketbook (*Hamiota altilis*, listed as Threatened under the Endangered Species Act), the Alabama spike (P1) and the delicate spike (P1) are known to occur in the drainage (Irwin et al. 1998). Two priority level two species (P2) of crayfish occur in the piedmont region of the Tallapoosa River Basin; *Cambarus englishi*, the endemic Tallapoosa crayfish and its close relative the *Cambarus halli*, the slackwater crayfish. In addition, records of the common, widespread (P5) white- tubercled crayfish, *Procambarus spiculifer,* have been documented in the region (Ratcliffe and Devries 2002). Until recently, no life history studies existed on either of the two Cambarid species other than original species accounts. This diversity presents a challenge for management of flow regulated systems. Additional information on occupancy rates of these species will benefit conservation efforts.

Species of Concern

Recent inventory (Mirarchi 2004; AL-CWCS 2005) indicates there are at least 13 species with conservation issues in the Piedmont reaches of the Tallapoosa River in Alabama (Table 1.). Eight of these were identified in the CWCS and the others are considered species at risk by the U.S. Fish and Wildlife service.

The lipstick darter *Etheostoma chuckwachatte*, has been listed by the State of Alabama as a species of “high conservation concern” and is on the Non-Game

Species Regulation list (Alabama Regulation 220.-2-.92; P2; Mirarchi 2004). *E. chuckwachatte* occurs exclusively in the Tallapoosa River system above the Fall Line (Boschung and Mayden 2004) typically in shallow water habitats with moderate-to-high velocities and coarse bed sediment (Freeman et al. 1997). The establishment of Harris Dam has fragmented and isolated populations of *E. chuckwachatte* and altered the habitat associated with the river's natural flow regime (Bowen et al. 1998). However, comparison of species abundance between populations below and above Harris reservoir indicates that *E. chuckwachatte* may be persisting in at least one segment (22 km downstream from the dam) of the flow-regulated reach below Harris Dam (Freeman et al. 2001), and in unregulated tributaries that flow into the regulated reach (Freeman et al. 2004). Studies are needed to estimate species occurrence in the Tallapoosa River below the hydropower dam, as well as to monitor potential long-term effects of the altered flow regime on this listed species.

Four additional fish species have been identified as species of "moderate conservation concern": stippled studfish *Fundulus bifax*, "Tallapoosa sculpin" *Cottus tallapoosa* Gulf-strain striped bass *Morone saxatilis*, and "muscadine darter" *Percina smithvanizi* (i.e. watch list; Mirarchi 2004). The muscadine darter is closely related to the Warrior bridled darter *Percina* sp. cf. *macrocephala*, which is a species of "highest conservation concern" (P1) that is endemic to the Black Warrior River system (Boschung and Mayden 2004). Monitoring the response of the muscadine darter in the Tallapoosa system will potentially provide valuable information for the conservation of its sister species in the Black Warrior Basin.

Fundulus bifax is primarily limited to medium-sized streams in the Tallapoosa drainage (save for a small population in a tributary of the Coosa River close to its confluence with the Tallapoosa), and are generally uncommon (Freeman et al. 2004; Irwin et al. 2004). The U.S. Fish and Wildlife Service and Alabama State DCNR are closely watching the status of this species and data are lacking relative to life history traits that may contribute to the species apparent decline. The Tallapoosa sculpin is locally abundant in tributaries of the Tallapoosa River (Irwin and Peyton 1997) but rare in the mainstem (Irwin and Freeman, unpublished data). The species is on the watch list because of lack of data (Mirarchi 2004).

Objectives:

The main objective of this project was to evaluate the effects of experimental flows on recovery of shoal dwelling fauna and persistence of shoal habitat in the Tallapoosa River. Secondary objectives were to 1) develop, implement and evaluate strategies for monitoring fauna in large river systems and 2) incorporate knowledge into future management and conservation programs of river systems. There are currently two main changes to flows from R. L. Harris Dam; increase in base flow and provision of spawning windows. Species of concern were targeted; however, community wide assessment was also conducted. Two unregulated river reaches (Hillabee Creek and the upper Tallapoosa River) were monitored to assess how measured state variables fluctuated independent of regulated flows (i.e., under "natural" conditions).

Specifically we:

1. Calculated an index of biotic integrity (IBI; Bowen et al. 1996) for each site.

2. Compared fish and crayfish occupancy and detection probabilities and extinction and colonization rates (fishes only) among regulated and unregulated reaches. We also modeled these population state variables as a function of various co-variates (e.g., distance from dam, habitat parameters).

3. Quantifed the proportion of the species in the fish assemblage that were in reproductive condition and consequently likely to take advantage of spawning windows if they were provided.

4. Estimated hatch date for two nesting Centrarchids to determine relations between hydrology and successful recruitment.

Methods:

Sampling was conducted in Spring (May-June) and Fall (September-November 2005-2009. We randomly selected five shoals per sampling reach (Figure 1). We used prepositioned area electrofishers (PAEs; Bowen et al. 1998; Freeman et al. 2001) to sample fishes and crayfishes from shoals. In 2005-2007 we collected 20 PAE samples per shoal and in 2008-2009 we collected 10 PAE samples per shoal. The reduced effort was warranted based on analysis of previous year's data and allowed us to spend more effort on field identification (see below).

Specific microhabitat variables were measured for each PAE sample: depth (cm), velocity (cm/s), percent vegetation (% areal coverage), and substrata composition were recorded. Depth and velocity were measured using a Marsh-McBirney flow staff and meter. Percent vegetation and substrate composition were quantified by visual estimation. Substrate particle designation were defined as silt (<0.1 mm), sand (0.1-1mm), gravel (0.1-6 cm), cobble (6-12cm), boulder (>12 cm), and bedrock and were recorded in the order of dominance.

Fishes.-In 2005-2007 fishes were euthanized in MS-222, preserved in 10% formalin and returned to the laboratory for processing. Larger specimens were field identified and released after total length (mm) data were recorded. Fishes were transferred to 70% ethanol for long-term storage. In the laboratory, specimens were identified to species and total length (mm) data were recorded. In 2008 and 2009, specimens were field identified, measured and released; small individuals that could not be identified without a microscope were preserved and returned to the laboratory as described above.

Crayfishes.-Similar to fishes, specimens from 2005-2007 were preserved and brought back to the laboratory for processing; 2008-2009 samples were field identified, measured and released. Carapace length (CL = tip of rostrum to post-median margin of carapace) was measured to the nearest 1.0 mm using calipers. Specimens were sexed and identified to species when possible; individuals smaller than 14 mm in carapace length were not identified to species and were classified as juveniles. Because of erroneous field identification during the summer of 2008, no specimens were identified to species, and were simply classified as either adult or juvenile based on aforementioned carapace length criteria.

IBI

IBI values were calculated for spring and summer samples in each year for each site (25 total sites) using methods from Bowen et al. (1996); metrics are listed in Table 2.

Fish species detection, occupancy, and extinction/colonization analysis

For fourteen selected fish species estimates of detection, occupancy, extinction, and colonization were calculated using maximum likelihood methods and modeled as a function of measured covariates using the logit link function following the methods of MacKenzie et al. 2006. Covariates modeled with detection and occupancy were based on the *a priori* hypotheses that detection varies by specific habitat sampled within a PAE and occupancy differs between regulated and unregulated segments of the Tallapoosa, or along a linear gradient downstream from Harris Dam; extinction and colonization were modeled according to the hypothesis that these values will vary between regulated and unregulated sites (Table 3). Competing models were compared using Akaike's information criterion (AIC; Burnham and Anderson 2002). To limit the number of models in each model set, detection, occupancy, and extinction/colonization were first modeled separately, and covariates were selected for the final model set based on strength of evidence among competing models and careful review of raw data. The final model set was examined for weak covariates (i.e., those covariates that did not add substantial model support to the best model; Burnham and Anderson 2002: 131, 173); models with covariates that did not add substantial model support were eliminated. All analyses were conducted using Program Presence v. 2.2 (Hines 2006). Reported parameter estimates were those of the best model for each species. Because laboratory identifications were not complete for 2009 at the time of writing this report, analysis was conducted on 2005-2008 data.

Crayfish population parameters, detection, and occupancy analysis

Crayfish catch data were examined for differences in catch per effort, size distribution, and species composition for differences between regulated and unregulated sites, using non parametric K-S tests and paired t-tests.

The same methods used for fishes were used for model building, selection, and analysis of crayfish data. However, different covariates were used (Table 4). and extinction/colonization values were not calculated for crayfish. Additionally, separate detection models for each year and species were analyzed to investigate species specific habitat preferences. For habitat analysis substrate was transformed in to a numerical index using substratum values modified from Gore and Bryant 1990 (bedrock and silt=0, sand=,1 gravel=2, cobble=3, small woody debris=3.5 bedrock shelf=4, large woody debris= 4.5 boulder=5,). The index was hypothesized to reflect the value of the substrate to provide refuge or alter velocity.

Reproductive condition analysis

Fish collected during 2005-2007 were examined for presence of viable reproductive organs. Ten percent of samples collected from each shoal for each year were randomly subsampled (i.e., two samples of 20 total samples per shoal) for reproductive assessment. In order to ensure a representative sample, a minimum of ten fish were assessed from each subsample. If this criterion was not met another randomly selected sample was examined until the numerical criterion was met. However, if ten fish were examined before the subsample was completed the remaining fish in the subsample were also assessed.

Fish were identified to species, total length (mm) was recorded, and prepared for dissection. All dissections were made ventrally from the urogenital opening posteroanterior through the pectoral girdle with small dissection scissors. With the dorsal side down each lateral section was stretched and pinned to a dissection pan with insect pins to expose internal organs. A dissecting microscope was used to determine presence/absence of reproductive organs, followed by reproductive stage (visible, mature, reabsorbing, or spent) and sex. Specimens ≤35 mm were excluded because damage from dissection made it difficult to visually assess reproductive organs.

Percent mature females was determined for each species as an indicator of reproductive condition. Reproductive condition was assessed for nine species where sample sizes were adequate. Therefore, species with <75 total individuals were excluded from assessment; however stippled studfish *Fundulus bifax* and Tallapoosa sculpin *Cottus tallapoosae* were considered because of their GCN status (Mirarchi 2004).

Hatch date assessment

Young-of-year (YOY) redbreast sunfish *Lepomis auritus*, spotted bass *Micropterus punctulatus,* and redeye bass *M. coosae* were collected using prepositioned area electrofishers (PAE) and backpack electrofishing units at approximately 30, 60, and 90–days after the onset of spawning in spring/summer 2005 and 2007. These times were determined by water temperatures reaching 22º C and the presence of nests in the river. Fish were collected from regulated (Tallapoosa River at Wadley, Alabama) and unregulated (Tallapoosa River at Heflin, Alabama, and Hillabee Creek, Alexander City, Alabama) reaches of the Tallapoosa River. Fish were euthanized in MS-222 (140 mg/l), placed on ice and returned to the laboratory. Total length (TL, mm) and weight (g) data were recorded and sagittal otoliths removed. Daily ages and hatch dates were estimated following the techniques of Taubert and Coble (1977), Santucci and Wahl (2003), and Roberts et al. (2004); for black basses Miller and Storck (1982), Isely and Noble (1987a,b) and Jones and Brothers (1987).

Hydrologic data from USGS gage stations was examined against hatch frequencies to determine optimal flow conditions for spawning and subsequent recruitment (www.usgs.gov; 1) USGS 02412000, Upper Tallapoosa River near Heflin, Alabama, 3) USGS 02414500, Middle Tallapoosa River at Wadley, Alabama, and 3) USGS 02415000, Hillabee Creek near Hackneyville, Alabama.

Results:

IBI

Calculated IBI values are presented in graphical form in Figure 2. Values varied widely among sites, within and among river reaches, between seasons, and among years (Figure 2). Overall, regulated sites had lower IBI values, but these values were not always consistent (Figure 2).

Fish Species Detection, Occupancy, Extinction and Colonization

Forty-nine species of fish from nine families were captured and identified during the course of the study. The species encountered most often across the entire basin was *Percina palmaris* (2061 encounters; Table 5); this species also had the most proportional encounters (adjusted by number of samples) in the regulated river between Harris Dam and Wadley (Table 5). In the Upper Tallapoosa, *Etheostoma stigmaeum* was proportionally encountered most often; at Horseshoe Bend and Hillabee Creek, *Cyprinella callistia* was the most-encountered species (Table 5).

Estimated values of occupancy, extinction, and colonization are given in Table x. Nine of the selected species had estimates that varied between regulated and unregulated sites ("'Dam' Effect;" Table 6). Two of these species (*Cyprinella callistia* and *Moxostoma duquensnei*) had parameter estimates that showed a linear gradient in relation to distance downstream from Harris Dam, and two species (*M. poecilurum* and *Cottus tallapoosae*) had parameter estimates that differed among sites (Table 6). *Cyprinella callistia* had an estimated occupancy of near 0 in 2005 at the site below the dam (2.5 km); this value increased as distance from the dam increased, such that occupancy was estimated as 1 at Horseshoe Bend (61.0-91.1 km; Table 6). In 2005, estimates of occupancy for *M. duquesnei* were 0 for all regulated sites until the furthest site at 91.1 km below the dam, which had an occupancy estimate of 1 (Table 6). Occupancy estimates of *M. poecilurum* were greatest in 2005 at the shoals below Harris Dam (Dam to Malone), followed by the Upper Tallapoosa and Hillabee Creek; occupancy was estimated as 0 at the shoals from Malone to Horseshoe Bend (11.9-91.1 km; Table 6). *Cottus tallapoosae* occupancy estimates were greatest in the Upper Tallapoosa, followed by the shoals between Malone and Wadley (11.9-19.0 km) in 2005, Horseshoe Bend, the shoals between Harris Dam and Malone (2.5-9.7 km), and Hillabee Creek (Table 6).

Extinction and colonization rates varied among species. *Cyprinella callistia*, *Etheostoma tallapoosae*, and *Percina palmaris* had extinction estimates of 0. The species with the greatest extinction rate was *Cottus tallapoosae*, which had an estimated extinction of 1 in the regulated reach; extinction was also estimated as greater than colonization in the unregulated reaches, such that occupancy rates were estimated as lower in subsequent years across all sites in the basin (Table 6). *Micropterus punctulatus, Etheostoma chuckwachatte,* and *Percina smithvanizi* all had estimated colonization rates of 1; *Cyprinella gibbsi* had an estimated colonization of 1 in the unregulated reaches only. The species with the lowest colonization rates were *M. poecilurum* (across all sites) and *Fundulus* *bifax* (in the regulated reach), which each had an estimated colonization rate of 0 (Table 6).

Crayfish catch-per effort, size and community composition

Over the five year sampling period a total of 1650 crayfish were sampled using PAE's (Table 7); a total of three species were identified: *P. spiculifer*, *C. englishi*, and *C. halli*. No significant differences (p=0.071) were found between CPEs at regulated and unregulated sites, however when juveniles were excluded CPEs were slightly different (p=0.04; Figure 9-10). Significant seasonl differences (p=0.001): summer mean CPE = 0.69 (CI: 0.52 -0.87) was higher than fall mean 0.37 (CI: 0.27-0.48). However, no significant seasonal differences (p=0.66) in CPE were observed when juveniles were excluded Figures 9-10).

When all *P. spiculifer* data were pooled (n=572), differences (p>0.0001) were observed in carapace length between regulated (32.1mm; CI: 31.1mm-33.1mm) and unregulated (27.6 mm; CI: 26.5mm-29.1mm) sites. However, crayfish species length data had significant differences in carapace lengths between seasons and among years. When regulated and unregulated differences were compared by season and year, significant differences were only found in the summer of 2007 (p<.0001) for all three species. Mean body size of *Procambarus spiculifer* decreased significantly (*p*<0.001) between 2006-2007 at unregulated sites; whereas, mean body size did not significantly differ between years at regulated sites (*p* = 0.93). This particular comparison was not made between other species due to low sample sizes.

Percentage of *P. spiculifer* catch was significantly higher (p=0.02) in the unregulated sites (65%; CI: 55%-75%), versus regulated (51%; CI: 44%-59%) when all years were pooled (Figure 11). Percent composition of *C. englishi* differed overall (p=0.0002) between regulated and unregulated sites (Figure 11). When analyzed by year, percent composition of *C. englishi* differed between regulated and unregulated sites in 2006 (p=0.0004) and 2007 (p=0.004). Percentage of *C. halli* was found to differ between regulated and unregulated sites (p=0.044) for all years pooled (Figure 11). Juvenile composition did not differ between regulated and unregulated sites when all data were pooled (p=0.09); however, when analyzed by year 2006 (p=.001) and 2009 (p=0.003) were significantly different in percent juvenile than other years.

Crayfish Species Detection and Occupancy Results

For all species, detection was a function of habitat variables. Table 7 indicates the top detection model for each model set. Vegetation and velocity had a positive effect while depth consistently had a negative influence on detection for all species; although the strength of each variables the effect differed among species. The relationship and importance of substrate varied among species detection models. Selected examples in Figures 13-15 demonstrate the relation of species detection to specific habitat parameters. Estimated values of detection probabilities (Figure 12) and occupancy (Table 8) are reported by year and species.

Reproductive condition analysis

A total of 28 species representing six families: Catostomidae, Cyprinidae, Cottidae, Fundulidae, Ictaluridae, and Percidae were examined (Table 10). In spring of 2005 no samples were collected in the unregulated stretches. No mature female Tallapoosa shiners *Cyprinella gibbsi* or bullhead minnows *Pimephales vigilax* were examined at the regulated stretches (Table 11). Mature female largescale stonerollers *Campostoma oligolepis* were never observed for any year or site. Tallapoosa shiner mature female percentages were highest in 2006. Alabama shiner *Cyprinella callistia* mature female percentage was low (<15%) for both regulated stretches in 2005, but was highest in 2006 at Dam to Malone stretch. At unregulated stretches *Cyprinella callistia* percentages were highest in 2006 and lowest in 2007.

All female *Etheostoma* species observed from the most highly regulated stretch (Dam to Malone) were mature in 2005 (Table 11). Percent of bronze darter *Percina palmaris* mature females was similar between the two regulated stretches for 2005, but the highest percentage was at Hillabee Creek in 2006. A high percentage of mature female lipstick darters *Etheostoma chuckwachatte* was observed at the Malone to Wadley stretch in 2005 and 2006. Mature female Tallapoosa darters *Etheostoma tallapoosae* were not examined from the unregulated stretches but those from the regulated stretches consisted of 50% or greater mature females across years. Mature female speckled darters *Etheostoma stigmaeum* percentages were highest at the regulated stretches for 2005 and 2006. Muscadine darters *Percina smithvanizi* were examined from both regulated and unregulated stretch and 50% were mature females in the unregulated mainstem stretches in 2006 and >50% in the regulated stretches for all years.

One individual female Tallapoosa sculpin *Cottus* sp. cf. *C. bairdii* was observed in fall 2005 at the Dam to Malone site in reabsorb stage. A total of nine stippled studfish *Fundulus bifax* specimens were examined from Hillabee across years. One visible male was examined in fall 2005, one visible male in spring 2006 and three absent in fall 2006. Three were examined in fall 2007 consisting of one mature female, one reabsorb stage female and one visible male. Fall assessment was limited as very few fish were of reproductive condition. In 2005, 50% (n=2) of female bronze darters were mature at the Malone to Wadley stretch and 1.4% (n=71) at Hillabee Creek stretch in 2006. A few Alabama shiner females were mature (4.8%, n=21) in the unregulated mainstem in fall 2006. Mature female bullhead minnows (3.3%, n=61) and stippled studfish (33.3%, n=3) were observed at the Hillabee Creek stretch and 12.5% (n=16) muscadine darters at the Dam to Malone stretch in fall 2007.

Hatch date assessment

A total of 194 age-0 redbreast sunfish (n=115), spotted bass (n=52), and redeye bass (n=27) were aged using sagittal otoliths. Average age of collected redbreast sunfish was 54 d (range= 25 to 135 d) and average total length 53 mm (range = 22-119 m; Figure 16). Average age of collected spotted bass was 68 d (range = 35-126 d) and average total length was 66 mm (range = 37-104 mm; Figure 17). Average age of collected redeye bass was 74 d (range = 51-119 d) and average total length was 68 mm (range = 40-94 mm; Figure 18). Analyses of total length versus site and age for all species resulted in no significant between site differences (regulated vs. unregulated) in the slope of total length versus age and site type (i.e., site effect; Table 12). Between site comparisons of total length for all species were not significant, indicating only small differences in total length between regulated and unregulated sites.

Hatch date estimates for all species in 2005 did not support the hypothesis that successful recruitment depends on prolonged stable flow periods. The majority of hatches fell on dates with releases from Harris Dam and no clear pattern was observed between hatch frequency and hydrology , presumably due to low sample size and more water in the river due to increased rainfall (Figures 19-21). Hatch date estimates for all species in 2007 did support the hypothesis that successful recruitment depends on prolonged stable flow periods. The majority of hatches occurred on or 2-3 days after periods of stable, low flows (Figure 22-24). Sixty-nine percent (n=60) of redbreast sunfish hatched when discharge was less than 600 cfs, 57% of spotted bass hatched at discharges less than 400 cfs, and 70% of redeye bass hatched at discharges less than 500 cfs.

Discussion

Maximizing conservation potential in free-flowing sections of rivers of Alabama will require, at minimum, clear evidence for effects of the present and of alternative regulated flow regimes on river biota. Despite potential obstacles (Irwin and Freeman 2002), an adaptive management approach holds substantial promise for improving management of regulated rivers by allowing managers and scientists to address the uncertainty in predicting and measuring how river fauna will respond to flow-regime alterations.

Fish Occupancy and IBI

Of the six GCN species in the Tallapoosa River, two, both darters, were apparently unaffected by the impact of Harris Dam, as indicated by the high occupancy and colonization estimates across all sampled sites: Tallapoosa darter and muscadine darter. The other GCN darter species, lipstick darter, was the only species of all fourteen species evaluated to exemplify a slight positive response to the effects of regulation, as indicated by the greater occupancy estimates and extinction estimate of 0 within the regulated reach. The response of these species is likely related to habitat requirements; darters are benthic species that are largely associated with shallow habitat. Since 2005, flows in the regulated portion of the Tallapoosa River have been maintained above historic regulated minimum flows (equal to flows in the upper Tallapoosa as recorded at the USGS gage at Heflin; Kennedy et al. 2006). It is possible that this flow management has provided for the persistence of this type of habitat, allowing these benthic organisms to thrive.

Occupancy estimates of the remaining three GCN species, Tallapoosa shiner, Tallapoosa sculpin, and stippled studfish, suggested that these species are either in decline (for the former two species) or absent altogether (for the latter species) in the regulated reach below Harris Dam. Although Tallapoosa shiner is typically detected more often in shallow water, it is not a benthic species; rather, it is usually found within the water column. Thus one possibility for its lower occupancy in the regulated river is the periodic high flows corresponding to generation events at Harris Dam. Whereas benthic organisms can maintain their position because of lower velocities due to frictional drag, species that occupy the water column are likely carried downstream along with high-velocity flows. It is possible, therefore, that these high flow events prevent this species from occupying much of the regulated river.

Unlike the Tallapoosa shiner, Tallapoosa sculpin and stippled studfish have very low detection probabilities even in the unregulated reaches; consequently reasons for their absence in the regulated river are unclear. It is possible that these species do not prefer large river habitat at all; in this case, an evaluation of local tributaries might provide information toward a better understanding of occupancy dynamics of these species.

Also of potential concern were the sucker species; both black redhorse *Moxostoma duquesnei* and blacktail redhorse *M. poecilurum* appear to be in decline in the regulated reach. Reasons for this decline are unclear, but could possibly be related to availability of habitat; both species depend on shoal habitat for spawning and for juvenile refuge (Boschung and Mayden 2004). Details of these requirements, such as timing and duration of preferred flows, are unknown; these are two species that could potentially benefit from specific targeted flows, or "spawning windows."

The three remaining species that were evaluated, channel catfish *Ictalurus puncatus*, redeye bass *Micropterus coosae*, and bronze darter *P. palmaris*, had estimated parameters that suggested both increases (e.g., redeye bass) and declines (e.g., channel catfish) in occupancy probabilities. However, these population changes were unrelated to river regulation. In general, our results indicated that the Tallapoosa River species assemblage varies considerably, not only between the regulated and unregulated river, but also within the unregulated reaches, both between seasons and among years. Because of the high number of species in the basin and the corresponding interactions among these species, it should follow that this assemblage undergoes dynamic changes in community structure as species respond differently to environmental (e.g., rainfall) and anthropological (e.g., hydrological regulation) changes that in turn variably affect flow dynamics, availability of food and cover, predator and prey abundances, and a possibly infinite number of other variables. Such dynamism is present naturally, even without the added complexity of anthropological stressors, such as flow regulation from hydropower facilities. Though the upper Tallapoosa River and Hillabee Creek are not void of human impact (e.g., from cattle grazing, forest clear-cutting, or urban development), the variation in IBI values within these sites and the estimates of species extinction and colonization that were constant among all sites in the Tallapoosa provide evidence that there is a natural level of variability that should be expected, and even perhaps managed for.

The IBI is an inherently rigid measure of ecological condition. That is, it assumes that high numbers (or high percentages of individuals) of certain species are always preferable. We contest that this "more is always better" assumption should be challenged in systems with high diversity and therefore dynamic community structure. The IBI also assumes that higher densities are always preferable. Because of the drought of 2007-2008, we have shown that this should not be an accepted assumption. In the summer of 2007, low flows forced all the fish in shoal “A” of Hillabee Creek into a very small area, resulting in densities so high that the calculated IBI values surpassed the maximum value (100; see Figure 2).

The metric values calculated at Hillabee shoal “A” also provide an example of the importance of incorporating detection probabilities into measures of fish community composition. The detection probability (e.g., the probability of capturing a species in one sample) of certain species increases as depth decreases; this is particularly the case for those species that have an affinity for low depths (e.g., Figure 6 and 7). Therefore, whether or not these species are detected on a particular occasion is highly dependent on the environmental conditions. Failing to incorporate detection into measures such as IBI can thus lead to erroneous assessments of ecosystem condition, as was the case (albeit a dramatic example) at Hillabee shoal “A.”

Because of the complexity of large river systems like the Tallapoosa River, it may not be appropriate to use rigid measures of ecosystem condition such as the IBI; rather an “informed species richness” value may be preferred. A measure of species richness alone does not have the information necessary to make a responsible judgment of ecosystem condition. However, because the components of species richness, that is the individual species within the community and their specific habitat and life history requirements, are the usual targets for management decisions, they should also be the targets for assessment of ecosystem condition. An “informed species richness” value would be based on a predicted species pool for a particular management area, and would incorporate species-specific detection probabilities based on habitat measures as well as season of capture. Figure 24 provides an illustration of how such a measure might be assessed. In this example, a predicted species pool was established, detection probabilities were estimated as functions of habitat, and these relations were incorporated into a network to predict the number of species and the probabilities of individual species collected in each sample. Added together for each sample collected, these probabilities would provide a total estimate of species number and composition at a particular site.

Crayfishes

Crayfishes are known to be difficult to sample (Rabeni 1997) and as a result had low detection probabilities. *P. spiculifer* had the highest detection probabilities of all species and *C. halli* had the lowest. One of the greatest factors affecting detection may be the underlying population size or density at a shoal at any given time (Royle and Nichols 2003). Although we modeled detection to the best of our ability with habitat covariates, we suspect there are still underlying heterogeneities that were not measured or were not identified.

Models of detection indicated vegetation and depth were the most important variables affecting detection of *P. spiculifer* in 2005-2007; whereas, in 2008 velocity and substrate were the most important variables. This may be a result of seasonal habitat responses related to changes in flow and vegetation cover in fall months. Detection probability or capture success of *C. englishi* was influenced in all years by vegetation and substrate and in most years by velocity. *C. englishi* demonstrated a positive relationship to substrate with higher detection over larger substrates such as boulders. Catch rates were consistently high for *C.*

englishi in the 8-12 km regulated section at sites which had a great proportion of larger substrates, however, very low catch rates were observed in Hillabee creek which had similar substrate features. Detection of *C. halli* was consistently and strongly influenced by depth. There was only one site where *C. halli* was never found in the presence of *C. englishi* therefore there is no evidence to support that either of the *Cambarus* species competitively exclude one another. However, differences in carapace lengths reflected in our data suggested that *C. englishi* were on average larger than *C. halli.* Additionally, the strong influence of depth on capture success of *C. halli* and of substrate type for *C. englishi* may be evidence of habitat partitioning among these closely related species. Dennard et al. (2009) reported similar results in that *C. halli* was smaller and shifted habitat use to exploit shallow riffles at sites when sympatric with *C englishi*. Juvenile crayfish detection was consistently influenced by vegetation and substrate; demonstrating a negative relationship to substrate and indicating their preference for gravel substrates because the interstitial refuge provided prevents predation from fish and larger crayfish (Stein and Mangson 1976; Flinders and Magoulick 2007; Ollsson and Nystrom 2009). Additionally, juvenile crayfish did not recruit to sampling gear until reaching 5mm; strong evidence in our catch data from all years of suggests this occurs around the second week of June at both regulated and unregulated sites.

Modeling indicated no effect of regulation on occupancy estimates for most species across most years with the exception of *C. englishi* in 2006 and 2007 and juvenile models in 2006. *P. spiculifer, C. halli,* and juvenile occupancy estimates all showed a decrease across years. A low occupancy estimate for *P. spiculifer* in 2008 could have been the result of missing species-specific data for the summer samples. However, the summer of 2008 sampling had the lowest observed CPE (0.28) among all years suggesting detection and underlying population density may have been low. In 2005 occupancy was estimated $\Psi = 1$ for both *C. halli* and juvenile models which may be a result of the models having extremely low detection probabilities which can result in an overestimate of Ψ (Mackenzie et. al. 2006). Therefore higher detection rates may produce more precise estimates of occupancy. Top occupancy models for 2006 and 2007 suggested that occupancy rates of *C. englishi* were potentially influenced by river regulation. In 2007 the distance model estimated a negative trend in occupancy following distance from the dam such that occupancy estimates were highest at the site closest to the dam. This is not consistent with the hypothesis of a downstream recovery gradient. Catch rates were consistently highest in the reach 8-12 km below the dam and catch rates were relatively lower in the unregulated sites and at Horseshoe Bend. Reasons for this pattern are unknown and are likely due to some unmodeled heterogeneity. In 2006, top juvenile models suggested a difference between regulated and unregulated sites both based on regulation group. However in 2006, 271 juveniles were collected on June 13th-14th at the Upper Tallapoosa sites compared to 3 individuals captured in the summer of 2006 at regulated sites when all sites were sampled before June 1st.

Reproductive Condition and Hatch Date Assessment

Freeman et al. (2001) reported that juvenile numbers of common species in the Upper Tallapoosa River and in the regulated section below Harris Dam varied yearly and by individual species and were correlated with persistence of shallow-water habitats. Our initial assessment of reproductive condition of fishes in the study reaches suggests that for most species, some individuals of each species that we assessed were reproductively viable in spring and early summer were not viable in the fall. On average the regulated reaches had higher percentages of mature females of similar species compared to the unregulated reaches. Tallapoosa shiner and bullhead minnow *Pimephales vigilax* were not usually detected in the regulated reaches. Accounts of tubercle formation on Tallapoosa shiners indicate that they may spawn in late-June or July (Boschung and Mayden 2004). River regulation may impact recruitment of Tallapoosa shiners and bullhead minnows below the dam. In addition to investigation of the importance of tributaries for maintaining these species, specific habitats for spawning should be quantified and evaluation of the effects of spawning windows on their populations should be made.

Campostoma oligolepis spawn in early spring (Boschung and Mayden 2004) before samples were collected; therefore mature females were not observed in our samples. Spring spawning windows may benefit stonerollers. Higher percentages of mature female *Cyprinella callistia* in the regulated reaches could be due to higher frequency of pulsed flows in 2006, but in 2007 lower numbers of mature females could be a direct effect of extreme drought conditions. Mature female *Etheostoma chuckwachatte* were abundant in 2005 and 2006 but because collection ceased after 2006, drought effects on their reproductive condition is not known. *Etheostoma tallapoosae* appears to be in reproductive condition in the regulated reaches and in general seem to be persisting well below the dam. *Fundulus bifax* and *Cottus tallapoosae* were not observed often in the reproductive assessment but efforts to better assess their populations should be considered as they are GCN species. In addition, length data will be assessed in the future to determine juvenile recruitment in relation to river regulation and rainfall patterns.

This assessment of reproductive condition was a first attempt to understand individual species ability to spawn in the system. Future assessments could include histological assessment of reproductive condition to better define reproductive condition. Better sampling to assess temporal trends in spawning periodicity would assist in determination of potential effectiveness of spawning windows.

Irwin and Freeman (2002) and Irwin et al. (1997) hypothesized that lower water temperatures from pulsing hypolimnetic releases likely delays spawning periods, impedes hatching success, and decreases rates of larval development. Periods of stable flow are expected to provide spawning fish and their nests refugia from higher, less stable flows and could increase juvenile bass hatching success (Irwin et al. 1997). Differences in hatch dates between years for redeye bass, redbreast sunfish and spotted bass were evident in our study. Increased rainfall resulted in higher discharges from Harris Dam and subsequently fewer numbers of hatches across all species in 2005. Comparisons of hatch frequencies to flow regimes in 2007 showed successful recruits hatched during periods of stable, low flows; stable flows (and higher temperatures) may have cued spawning and hatching earlier than in 2005. Stable flow periods may provide for greater availability of suitable spawning and juvenile habitat which allows for recruitment to a stage and size where fish can withstand daily fluctuating discharges.

The project will continue through 2010 and information regarding changes in habitat with changes in flow regime will be incorporated into our results. In addition we intend to investigate potential for reintroduction of the finelined pocketbook (*Hamiota altilis*; Roe and Hartfied 2005), another P2 species in the system and is listed as Threatened under the Endangered Species Act (Federal Register, 58 FR 14339, March 17, 1993). Though once distributed throughout the Mobile River basin, *H. altilis* is now limited to several isolated populations in the Cahaba, Coosa, and Tallapoosa River basins, primarily above the Fall Line (USFWS 2003; Mirarchi 2004). In the Tallapoosa River basin, populations are restricted to the upper reaches of the basin, above Harris reservoir (Freeman et al. 2004) and Sandy Creek, Chambers County (recent record; M. Gangloff). *H. altilis*, reported to inhabit sand, gravel, and cobble substrates in moderate currents, as well as depositional areas along stream margins (Parmalee and Bogan 1998; USFWS 2000; Mirarchi 2004), is threatened by fragmented populations and habitat degradation (Mirarchi 2004; Freeman et al. 2004). Due to its method of glochidia transmittal, *H. altilis* may also be threatened by potential reductions in populations of host fish (Mirarchi 2004; Freeman et al. 2004). *H. altilis* transmits glochidia via a superconglutinate “lure” that mimics a small swimming fish (Haag and Warren 1999), prompting attacks from host species such as basses *Micropterus* spp. or green sunfish *Lepomis cyanellus* (Haag et al. 1999; USFWS 2000). Conservation strategies for this species should include a thorough inventory in the regulated reach, identification of potential habitat for reintroduction, and specific flow conditions needed for reproduction.

Three additional mussel species, the Alabama spike (P1), the delicate spike (P1) and the Alabama creekmussel (P2) may be distributed in the reach. Alabama spike and delicate spike are known to occur in tributaries in the upper watershed, but may occur in tributaries or the mainstem in the regulated reach. Again, identification of functional habitat for introduction may be critical for these species.

Large rivers are inherently difficult to sample, yet we believe that our monitoring approach is robust for detecting effects of management. Results from this study will be applicable to many other regulated rivers in Alabama and other southeast states. For example, an adaptive management approach has been suggested for restoration of the Bypass Reach of the Coosa River. Monitoring approaches developed and evaluated on the Tallapoosa River will be transferable to other similar systems. Consequently, this project has provided a template for monitoring and adaptive management of aquatic systems.

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Table 1. Status of fauna in the Piedmont section of the Tallapoosa River basin, Alabama.

Taxa	Status	Reference
Alabama spike	P1	CWCS
Delicate spike	P1	CWCS
Finelined pocketbook	P2, Threatened	CWCS, ESA
Alabama creekmussel*	P2	CWCS
Cambarus englishi	P2	CWCS
Cambarus halli	P2	CWCS
*Cambarus cracens**	P2	CWCS
Lipstick darter	P2	CWCS
Tallapoosa darter	Watch list, “at-risk”	Mirarchi 2004, USFWS
Muscadine darter	Watch list, “at-risk”	Mirarchi 2004, USFWS
Stippled studfish	Watch list, “at-risk”	Mirarchi 2004, USFWS
Tallapoosa sculpin	Watch list, “at-risk”	Mirarchi 2004, USFWS
Striped bass	Watch list, “at-risk”	Mirarchi 2004, USFWS

*potential occurrence in basin

Table 2.-Metrics and expected values for the Tallapoosa River IBI (Bowen et al. 1996)

Category and metric	Expected Values
Species richness and composition	
Total *N* of fish species	49 or 44[a]
N of sucker species	4
N of darter species	6
N of sunfish species	10
Indicator species	
% of individuals as intolerant species	22
Evenness multiplied by 100	100
Trophic function	
% of individuals as insectivorous cyprinids	49
% of individuals as benthic fluvial specialists	85
Abundance	
Density (mean *N*/PAE sample)	24.8

[a]Bowen et al. 1996 estimated fewer expected species in the Upper Tallapoosa

Table 3.- List of covariates used to model detection, occupancy and extinction/colonization of fish species in the Tallapoosa River.

Detection	Occupancy	Extinction/Colonization
Depth	Regulated	Regulated
Velocity	Distance	
Vegetation	Group	
Season		
Year		
Wateryear		

Table 4.- List of covariates used to model detection and occupancy crayfish species in the Tallapoosa River.

Detection	Occupancy
Depth	Regulated
Velocity	Distance
Vegetation	
Substrate	

Table 5. -Species encountered in each of 5 sites in the Tallapoosa River basin. Values indicate the proportional number (based on number of samples) of times the species was encountered (not count data).

	Proportional Encounters					
	Upper Tallapoosa	**Dam to Malone**	**Malone to Wadley**	**Horseshoe Bend**	**Hillabee Creek**	**Total**
Ambloplites ariommus	10	10	5	31	71	85
Ameiurus natalis	4	1	0	0	8	9
Campostoma oligolepis	522	80	114	123	413	868
Cottus tallapoosae	101	2	3	3	4	75
Cyprinella callistia	610	118	326	821	781	1811
Cyprinella gibbsi	198	5	32	199	409	543
Cyprinella venusta	186	16	54	119	219	400
Dorosoma petenense	0	11	1	6	0	16
Etheostoma chuckwachatte	379	223	371	539	500	1469
Etheostoma stigmaeum	665	62	133	205	324	953
Etheostoma tallapoosae	173	123	111	273	128	584
Fundulus bifax	4	0	0	0	86	58
Fundulus olivaceus	64	1	1	0	18	55
Gambusia affinis	49	0	0	3	4	36
Hybopsis lineapunctata	0	0	2	3	2	5
Hypentelium etowanum	562	63	119	420	193	912
Ictalurus punctatus	108	11	49	107	66	234
Lepomis auritus	120	141	65	77	222	470
Lepomis cyanellus	7	6	2	0	3	14
Lepomis gulosus	0	0	0	0	5	4
Lepomis macrochirus	10	44	7	6	50	93
Lepomis megalotis	0	1	0	0	2	2
Lepomis microlophus	2	1	2	0	5	7
Luxilus chrysocephalus	64	10	11	9	102	134
Lythrurus bellus	3	0	0	0	2	3
Macrhybopsis aestivalis	0	0	0	0	113	73
Micropterus coosae	9	20	10	116	73	150
Micropterus punctulatus	51	19	13	113	67	173
Micropterus salmoides	2	4	1	0	5	10
Moxostoma duquesnei	119	1	0	31	82	149
Moxostoma erythrurum	5	0	0	0	2	4
Moxostoma poecilurum	33	5	0	0	23	42
Nocomis leptocephalus	0	0	0	0	14	9
Notemigonus crysoleucas	0	1	0	0	0	1
Notropis asperifrons	3	0	0	0	7	6
Notropis baileyi	3	4	21	3	2	30
Notropis stilbius	108	13	40	165	251	379
Notropis texanus	0	1	1	9	24	23
Notropis xaenocephalus	0	0	1	0	0	1
Noturus funebris	55	9	9	18	161	168
Noturus leptacanthus	249	3	7	61	111	279
Percina kathae	5	1	3	25	70	66
Percina palmaris	523	346	480	790	692	2061
Percina smithvanizi	411	130	261	423	431	1176

Table 5.-(continued). Species encountered in each of 5 sites in the Tallapoosa River basin. Values indicate the proportional number (based on number of samples) of times the species was encountered (not count data).

	Proportional Encounters					
	Upper Tallapoosa	Dam to Malone	Malone to Wadley	Horseshoe Bend	Hillabee Creek	Total
Phenacobius catostomus	243	1	0	15	151	264
Pimephales vigilax	455	0	0	31	222	456
Pomoxis nigromaculatus	0	0	0	3	0	2
Pylodictis olivaris	12	1	4	15	18	33
Semotilus atromaculatus	18	0	0	9	2	18

Table 6.-Estimated occupancy, extinction, and colonization values for selected fish species in the Tallapoosa River. Species are separated by those that showed a possible response to hydro regulation, and those that showed no differences between regulated and unregulated segments of the river.

	Site or Distance (km) below dam	Estimated Occupancy 2005	2006	2007	2008	Extinction	Colonization
		"Dam" Effect					
Cyprinella callistia	Upper Tallapoosa	1	1	1	1	0	0.2500
	2.5	0.0407	0.2805	0.4604	0.5953	0	0.2500
	8.3	0.6232	0.7174	0.7880	0.8410	0	0.2500
	9.2-9.7	0.8051	0.8538	0.8904	0.9178	0	0.2500
	11.9-12.0	0.9627	0.9720	0.9790	0.9842	0	0.2500
	16.0	0.9975	0.9981	0.9986	0.9990	0	0.2500
	16.8	0.9984	0.9988	0.9991	0.9993	0	0.2500
	19.0	0.9996	0.9997	0.9998	0.9998	0	0.2500
	Horseshoe Bend	1	1	1	1	0	0.2500
	Hillabee Creek	1	1	1	1	0	0.2500
Cyprinella gibbsi	Unregulated	0.7279	0.9736	0.9647	0.9650	0.0363	1
	Regulated	0.7279	0.5931	0.5359	0.5116	0.2914	0.2842
Moxostoma duquesnei	Upper Tallapoosa	1	1	1	1	0	0.0643
	2.5-61.0	0	0.0643	0.0951	0.1099	0.4561	0.0643
	91.1	1	0.5439	0.3251	0.2202	0.4561	0.0643
	Hillabee Creek	1	1	1	1	0	0.0643
Moxostoma poecilurum	Upper Tallapoosa	0.9066	0.9066	0.9066	0.9066	0	0
	2.5-9.7	0.9328	0.3404	0.1242	0.0453	0.6351	0
	11.9-91.1	0	0	0	0	0.6351	0
	Hillabee Creek	0.6251	0.6251	0.6251	0.6251	0	0
Fundulus bifax	Unregulated	0.3269	0.3561	0.3694	0.3755	0.3362	0.2067
	Regulated	0	0	0	0	0.3362	0
Cottus tallapoosae	Upper Tallapoosa	1	0.7264	0.5342	0.3994	0.2736	0.0244
	2.5-9.7	0.2749	0.0177	0.0239	0.0238	1	0.0244
	11.9-19.0	0.4868	0.0125	0.0240	0.0238	1	0.0244
	Horseshoe Bend	0.2848	0.0174	0.0239	0.0238	1	0.0244
	Hillabee Creek	0.2194	0.1784	0.1496	0.1294	0.2736	0.0244
Lepomis auritus	Unregulated	0.4370	0.7983	0.9277	0.9741	0	0.6417
	Regulated	0.4370	0.7366	0.8017	0.8158	0.1411	0.6417
Micropterus punctulatus	Unregulated	1	0.7986	0.8392	0.8310	0.2014	1
	Regulated	1	0.3228	0.7814	0.4709	0.6772	1
Etheostoma chuckwachatte	Unregulated	0.9203	0.9597	0.9580	0.9580	0.0438	1
	Regulated	0.9203	1	1	1	0	1
		No "Dam" Effect					
Ictalurus punctatus	All Sites	0.7326	0.6340	0.6272	0.6267	0.3477	0.5837
Micropterus coosae	All Sites	0.3117	0.5619	0.6750	0.7262	0.1268	0.4209
Etheostoma tallapoosae	All Sites	0.7281	0.9169	0.9746	0.9922	0	0.6945
Percina palmaris	All Sites	1	1	1	1	0	0.4422
Percina smithvanizi	All Sites	0.9219	0.9730	0.9715	0.9715	0.0293	1

Table 7.-Catch data summary for crayfishes from sites in the Tallapoosa River basin. Sites are arranged in a longitudinal fashion descending from Harris Dam. Sites below the solid black line are the unregulated sites (Hillabee Creek and the Upper Tallapoosa River.

Site	*Procambarus spiculifer*	*Cambarus englishi*	*Cambarus Halli*	Juvenile <14mm	Adult-2008 summer"	TOTAL
Malone A	24	3	1	12	3	43
Malone B	65	38	7	46	14	170
Malone C	53	32	7	8	5	105
Malone D	25	14	6	2	0	47
Malone E	39	38	6	5	3	91
Wadley A	26	13	0	9	3	51
Wadley B	14	20	5	15	1	55
Wadley C	25	60	19	39	1	144
Wadley D	18	25	5	8	0	56
Wadley E	27	4	4	2	0	37
Griffin A	5	4	7	4	0	20
Griffin B	6	1	6	1	0	14
Griffin C	9	8	3	2	0	22
Peters D	6	2	0	2	0	10
Peters E	5	2	0	2	0	9
Hillabee A	21	1	2	6	0	30
Hillabee B	2	0	0	3	0	5
Hillabee C	28	3	2	18	0	51
Hillabee D	29	2	2	11	1	45
Hillabee E	26	0	5	7	1	39
Upper A	19	3	8	86	0	116
Upper B	21	1	7	112	0	141
Upper C	41	10	27	97	0	175
Upper D	19	13	17	59	10	118
Upper E	19	2	5	26	4	56
Total	572	299	151	582	46	1650

Table 8.-Top models describing covariates affecting crayfish detection by species and year.

Species	Year	Top Detection Model
Procambarus spiculifer	2005	psi(.),p(depth + vegetation)
	2006	psi(.),p(depth + vegetation)
	2007	psi(.),p(depth + vegetation)
	2008	psi(.),p(velocity + substrate)
	2009	psi(.),p(velocity + vegetation + substrate)
Cambarus englishi	2005	psi(.),p(velocity + depth + vegetation + substrate)
	2006	psi(.),p(vegetation + substrate)
	2007	psi(.),p(velocity + vegetation + substrate)
	2008	psi(.),p(velocity + vegetation + substrate)
	2009	psi(.),p(vegetation + substrate)
Cambarus halli	2005	psi(.),p(depth + vegetation)
	2006	psi(.),p(velocity + depth + vegetation)
	2007	psi(.),p(velocity + depth + vegetation + substrate)
	2008	psi(.),p(depth + vegetation + substrate)
	2009	psi(.),p(depth + vegetation + substrate)
Juvenile	2005	psi(.),p(depth + vegetation)
	2006	psi(.),p(velocity + depth + vegetation + substrate)
	2007	psi(.),p(velocity + vegetation + substrate)
	2008	psi(.),p(vegetation)
	2009	psi(.),p(vegetation + substrate)

Table 9.-Occupancy estimates for crayfish species (and juveniles <14mm carapace length) for models incorporating regulated versus unregulated sites and distance from the dam. Top models are reported for each year.

Species	Site or Distance (km)	2005	2006	2007	2008	2009
Procambarus spiculifer		1	1	0.85	0.42	0.77
Cambarus englishi	unregulated	0.69	0.51	0.36	0.58	0.75
	regulated		1	.		
	2.5			0.82		
	8.3			0.81		
	9.7			0.81		
	11.9			0.80		
	12			0.80		
	16			0.79		
	16.8			0.79		
	19			0.79		
	Horseshoe Bend			0.58		
Cambarus halli		1	0.86	0.71	0.61	0.58
Juvenile	unregulated	1	0.92	0.92	0.71	0.71
	regulated		0.23			

Table 10.-List of fish species, yearly counts, and totals assessed for reproductive condition. Those species with <75 total individuals were excluded from further reproductive condition assessment with exception to *F. bifax* and *Cottus tallapooosae*

Species	2005 Count	2006 Count	2007 Count	**Total**
Cyprinella callistia	137	231	219	587
Percina palmaris	124	191	231	546
Campostoma oligolepis	31	170	88	289
Pimephales vigilax	36	159	98	293
*Percina smithvanizi**	46	52	97	195
Etheostoma stigmaeum	35	45	101	181
*Etheostoma chuckwachatte***	122	63	.	185
Cyprinella gibbsi	19	33	67	119
*Etheostoma tallapoosae**	16	30	37	83
Cyprinella venusta	12	46	12	70
Notropis stilbius	21	7	31	59
Hypentelium etowanum	11	8	7	26
Phenacobius catostomas	10	6	7	23
Luxilus chrysocephalus	4	3	2	9
Noturus funebris	2	13	3	18
Noturus leptacanthus	1	8	2	11
Macrhybopsis aestivalis	.	8	2	10
*Fundulus bifax**	1	4	3	8
Percina kathae	4	2	.	6
Fundulus olivaceus	.	.	7	7
Notropis baileyi	5	1	.	6
Moxostoma duquesnei	.	3	2	5
Notropis texanus	4	.	.	4
Hybopsis lineapunctata	.	.	.	0
*Cottus tallapoosae**	1	.	.	1
Moxostoma poecilurum	.	1		1
Nocomis leptocephalus	.		1	1
Notemigonus crysoleucas	.		1	1

*species watch list “at-risk”, **GCN P2 species

Table 11.-Percent mature females (%♀) and sample size (n) from spring samples of 2005-2007 from unregulated and two regulated stretches.

	Spring			
	Unregulated		**Regulated**	
	Hillabee	**Upper**	**Malone**	**Wadley**
Species	**%♀ (n)**	**%♀ (n)**	**%♀ (n)**	**%♀ (n)**
2005				
Campostoma oligolepis	dns	dns	.	0 (3)
Cyprinella callistia	dns	dns	14.3 (7)	10.8 (37)
Cyprinella gibbsi	dns	dns	.	0 (13)
Etheostoma chuckwachatte	dns	dns	100 (4)	81.4 (43)
Etheostoma stigmaeum	dns	dns	100 (2)	66.7 (3)
Etheostoma tallapoosae	dns	dns	100 (5)	0 (1)
Percina palmaris	dns	dns	50.0 (26)	52.5 (40)
Percina smithvanizi	dns	dns	0 (2)	60.0 (5)
Pimephales vigilax	dns	dns	.	.
2006				
Campostoma oligolepis	0 (31)	0 (79)	.	.
Cyprinella callistia	30.4 (23)	0 (2)	100 (1)	62.5 (8)
Cyprinella gibbsi	40.0 (10)	50.0 (2)	.	.
Etheostoma chuckwachatte	.	.	92.3 (13)	96.0 (50)
Etheostoma stigmaeum	0 (1)	33.3 (6)	.	75.0 (8)
Etheostoma tallapoosae	0 (2)	.	66.7 (3)	100 (1)
Percina palmaris	80.0 (5)	33.3 (3)	63.2 (38)	77.8 (18)
Percina smithvanizi	0 (3)	50.0 (4)	100 (2)	80.0 (5)
Pimephales vigilax	22.7 (44)	48.2 (85)	.	.
2007				
Campostoma oligolepis	0 (52)	0 (8)	0 (1)	0 (4)
Cyprinella callistia	3.8 (26)	0 (20)	0 (8)	18.2 (11)
Cyprinella gibbsi	30.0 (20)	25.0 (4)	0 (1)	.
Etheostoma stigmaeum	0 (1)	0 (29)	33.3 (3)	0 (7)
Etheostoma tallapoosae	.	0 (1)	50.0 (4)	0 (1)
Percina palmaris	0 (7)	0 (11)	53.1 (32)	13.8 (58)
Percina smithvanizi	0 (7)	0 (2)	80.0 (5)	60.0 (25)
Pimephales vigilax	50.0 (2)	26.1 (23)	.	.

*species watch list, “at-risk”, **GCN P2 species, dns = did not sample

Table 12. Results of ANOVA testing effects of age and site on total length of all species.

Species/Effect	F-value	Probability>F	Site Types
Redbreast sunfish			Regulated, Unregulated
Site x Age interaction	1.64	0.16	
Site effect	2.76	0.11	
Spotted bass			Regulated, Unregulated
Site x Age interaction	0.19	0.68	
Site effect	0.75	0.42	
Redeye bass			Regulated, Unregulated
Site x Age interaction	-	-	
Site effect	0.62	0.51	

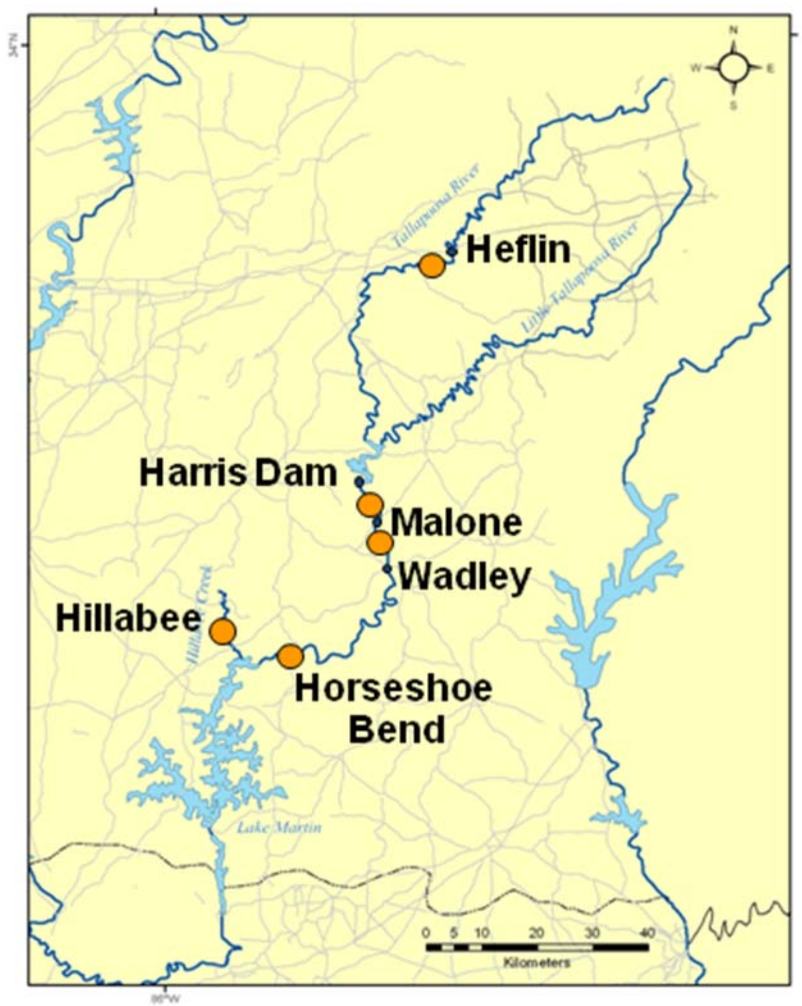


Figure 1.-Sampling reaches (orange dots) for faunal monitoring associated with adaptive management of the Tallapoosa River. Each reach (except Horseshoe Bend) has five randomly selected sampling shoals that are ordered linearly in a downstream fashion and labeled A-E. Horseshoe Bend has two randomly selected sampling shoals; however, increased effort was expended on those shoals.

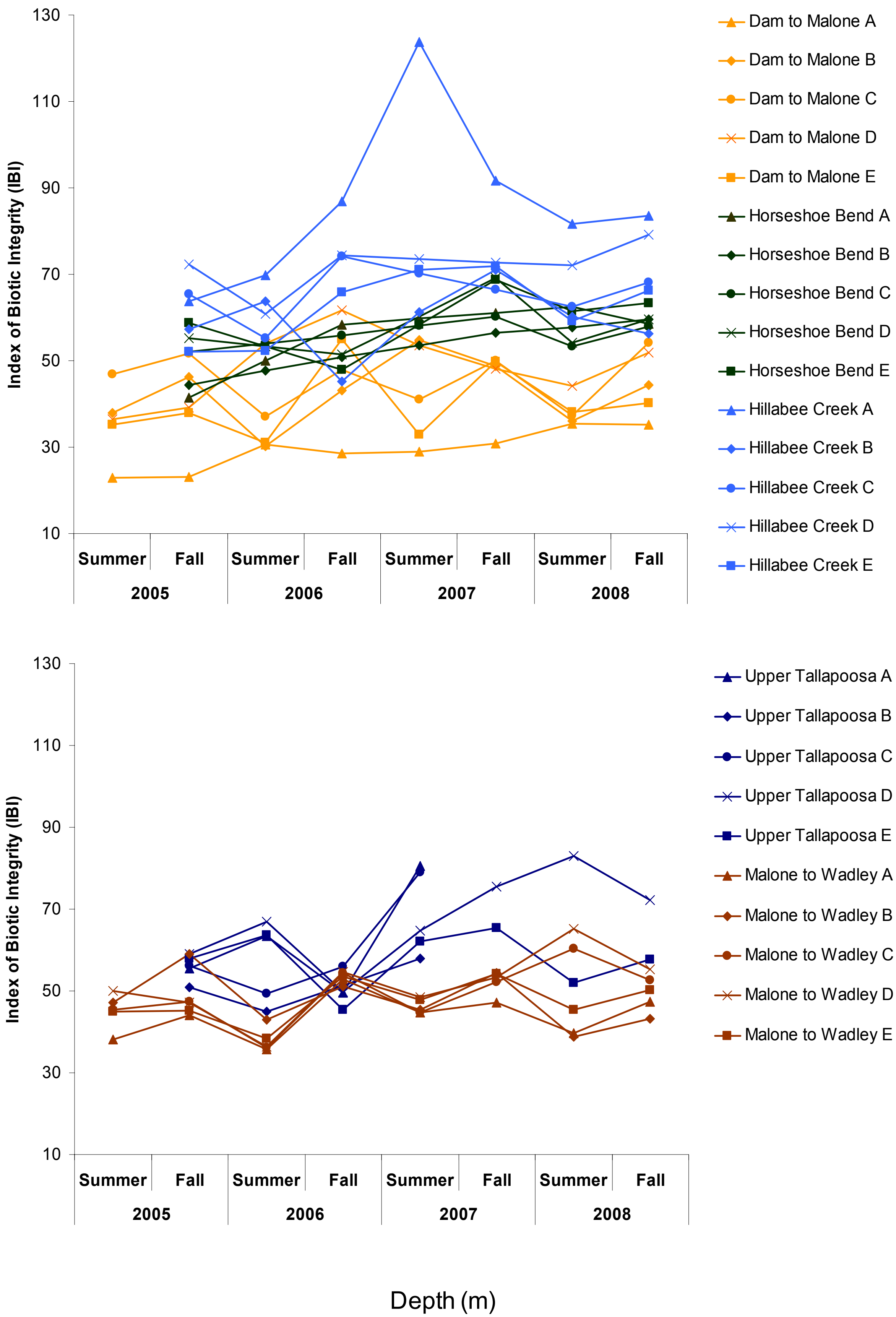


Figure 2. Calculated index of biotic integrity (IBI) values for the Tallapoosa River. Letters correspond to sampled shoals (see Figure 1), from upstream (A) to downstream (E).

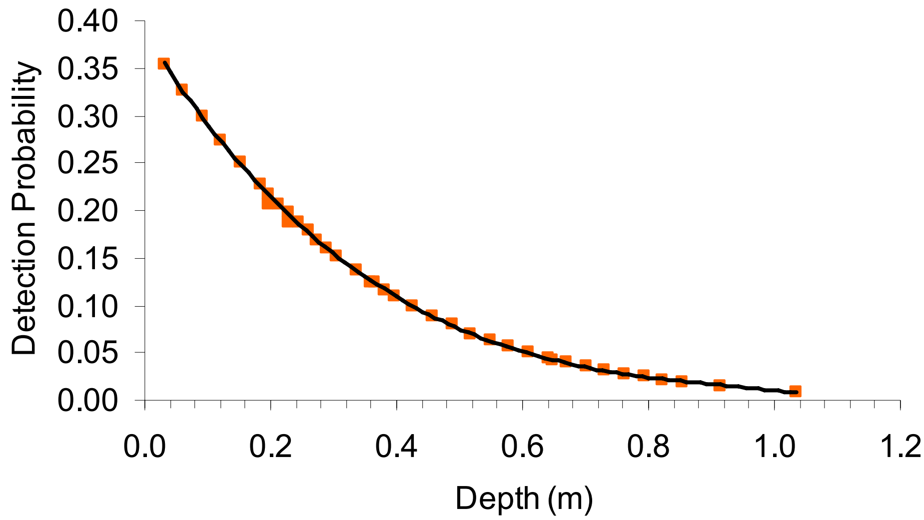


Figure 3.-Estimated detection probabilities of Tallapoosa shiner *Cyprinella gibbsi* plotted in relation to measured depth of each PAE sample.

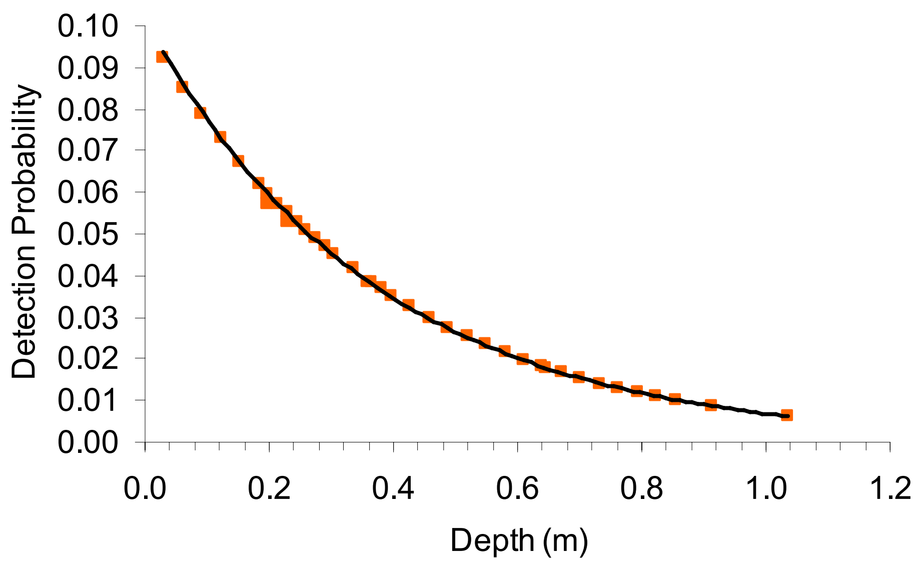


Figure 4.-Estimated detection probabilities of stippled studfish *Funduls bifax* plotted in relation to measured depth of each PAE sample.

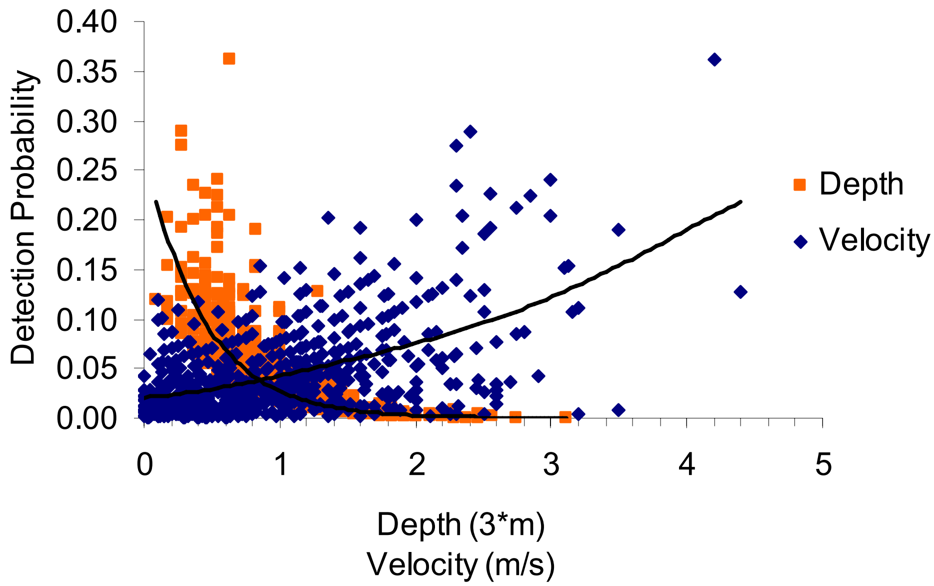


Figure 5. -Estimated detection probabilities of Tallapoosa sculpin *Cottus tallapoosae* plotted in relation to measured depth and velocity of each PAE sample.

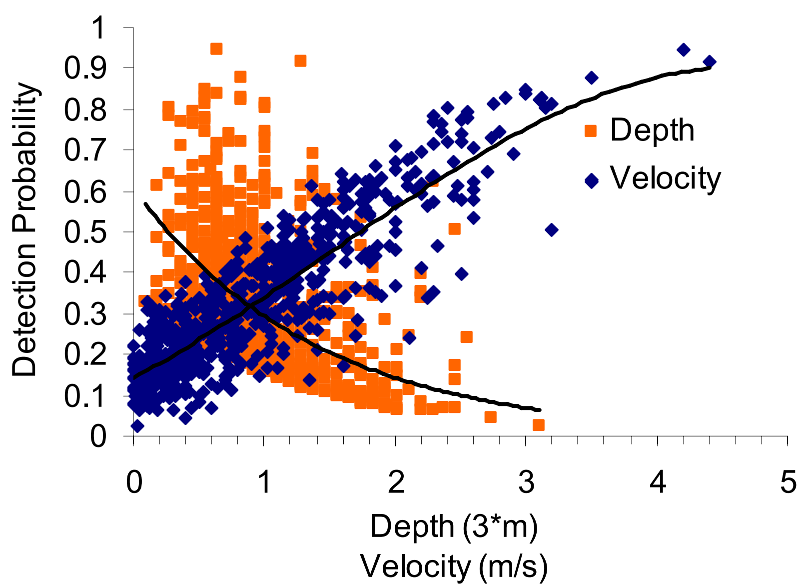


Figure 6.-Estimated detection probabilities of lipstick darter *Etheostoma chuckwachatte* plotted in relation to measured depth and velocity of each PAE sample.

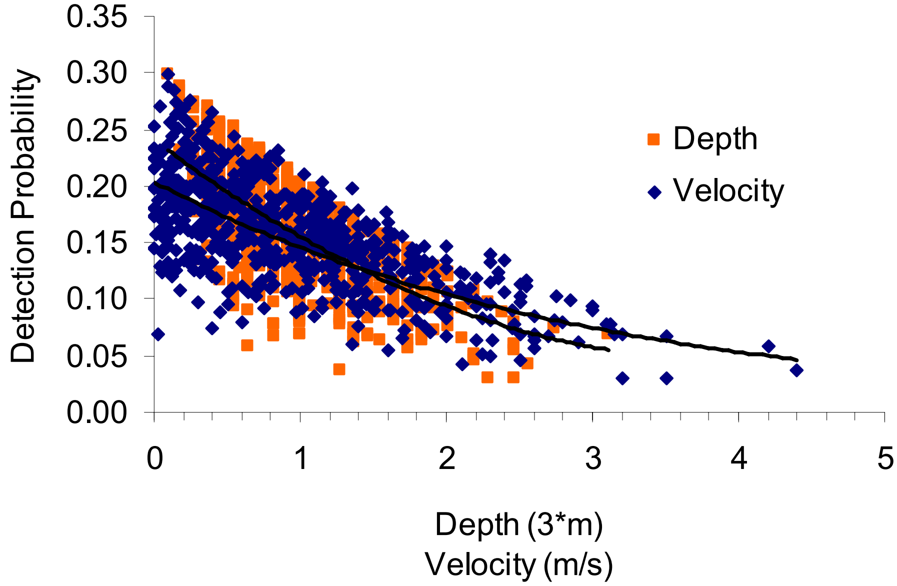


Figure 7.-Estimated detection probabilities of Tallapoosa darter *Etheostoma tallapoosae* plotted in relation to measured depth and velocity of each PAE sample.

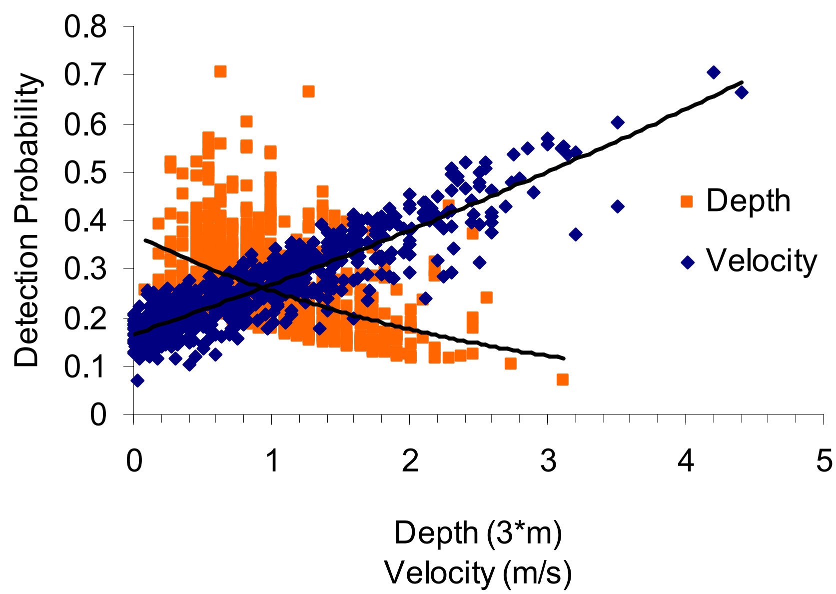
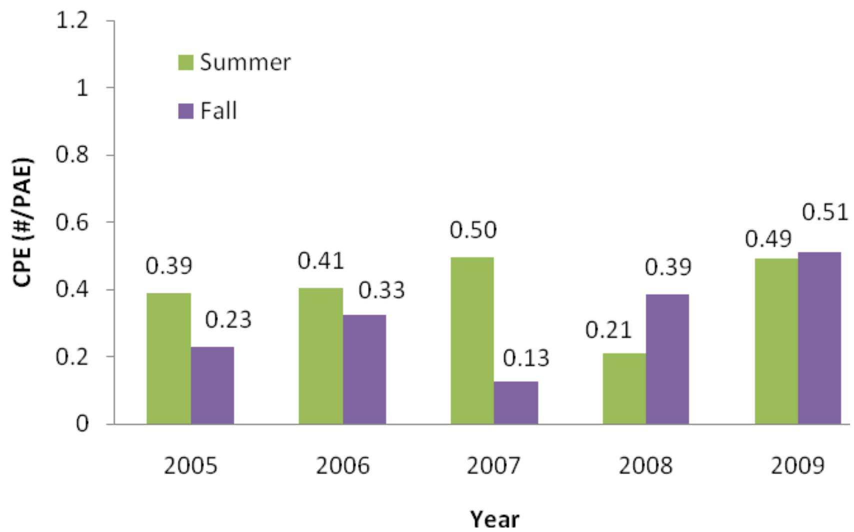


Figure 8.-Estimated detection probabilities of muscadine darter *Percina smithvanizi* plotted in relation to measured depth and velocity of each PAE sample



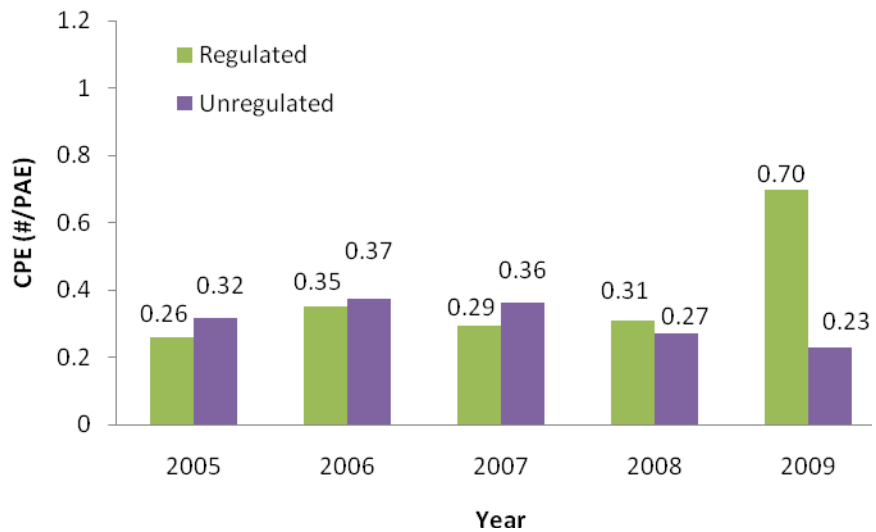


Figure 9.-Catch-per-unit-effort (#/PAE) of three species of crayfishes (*Procambarus spiculifer, Cambarus englishi and C. halli*) captured in the Tallapoosa River Basin. Only individuals with carapace length >14mm are represented.

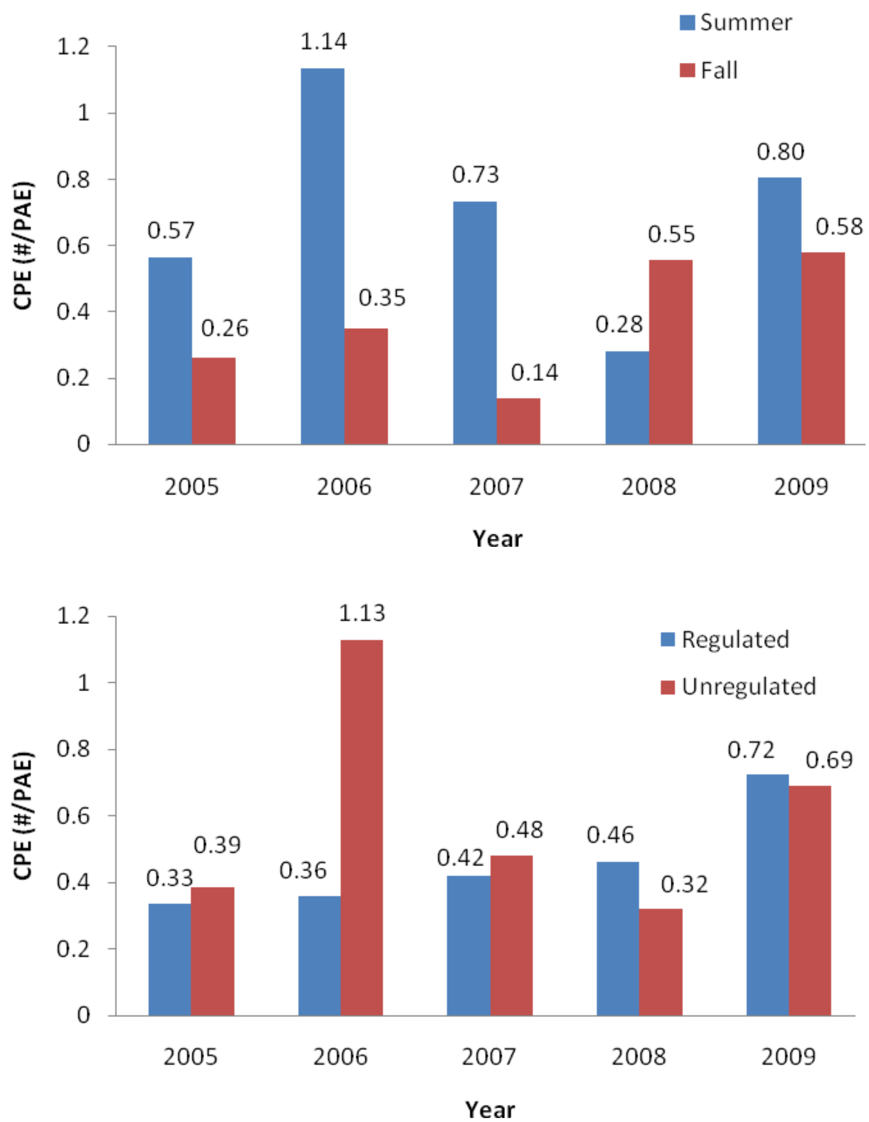
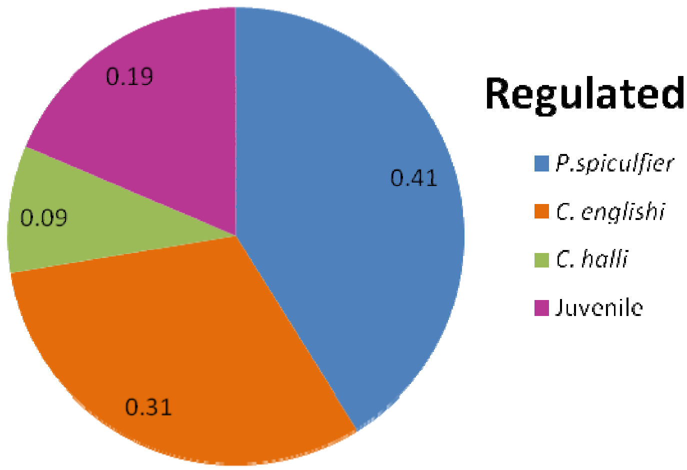


Figure 10.-Catch-per-unit-effort (#/PAE) of three species of crayfishes (*Procambarus spiculifer, Cambarus englishi and C. halli*) captured in the Tallapoosa River Basin. All sizes of individuals are represented in this graph.



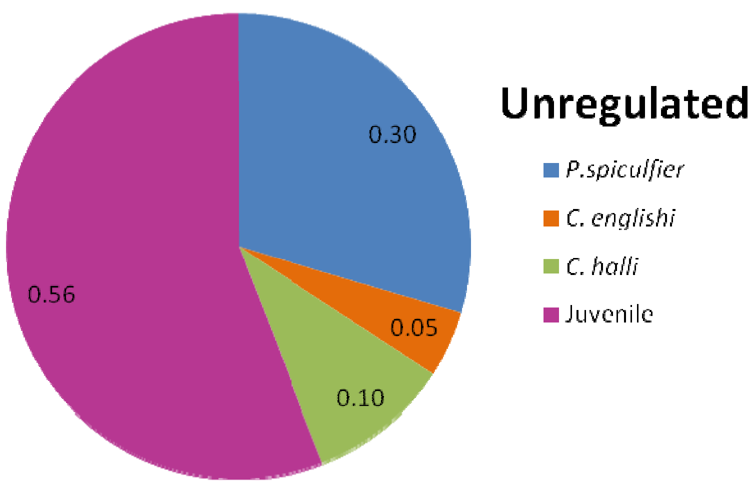


Figure 11.-Community compostition for three species of crayfishes (*Procambarus spiculifer, Cambarus englishi and C. halli*) and unidentified juveniles (<14mm carapace length) at regulated (top graph) and unregulated (bottom graph) sites in the Tallapoosa River Basin.

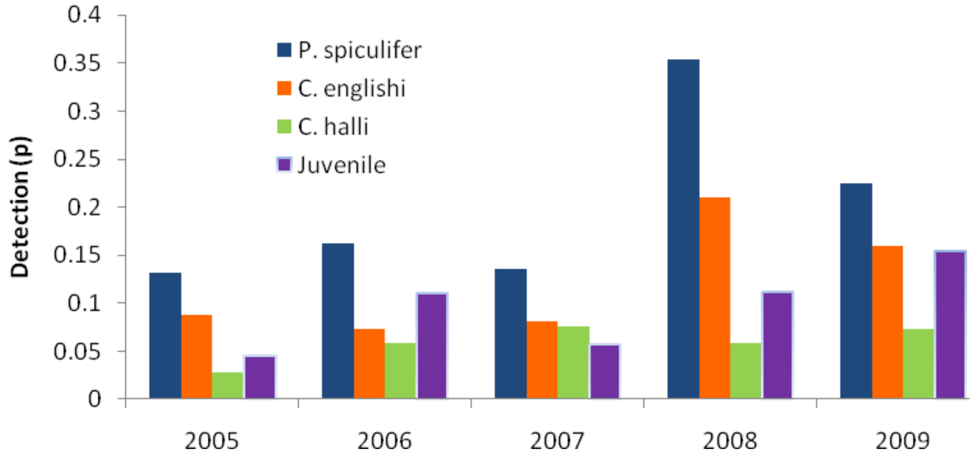


Figure 12.- Average detection probabilites (2005-2009) for three species of crayfishes (and juveniles <14mm carapace length) from sites in the Tallapoosa River Basin.

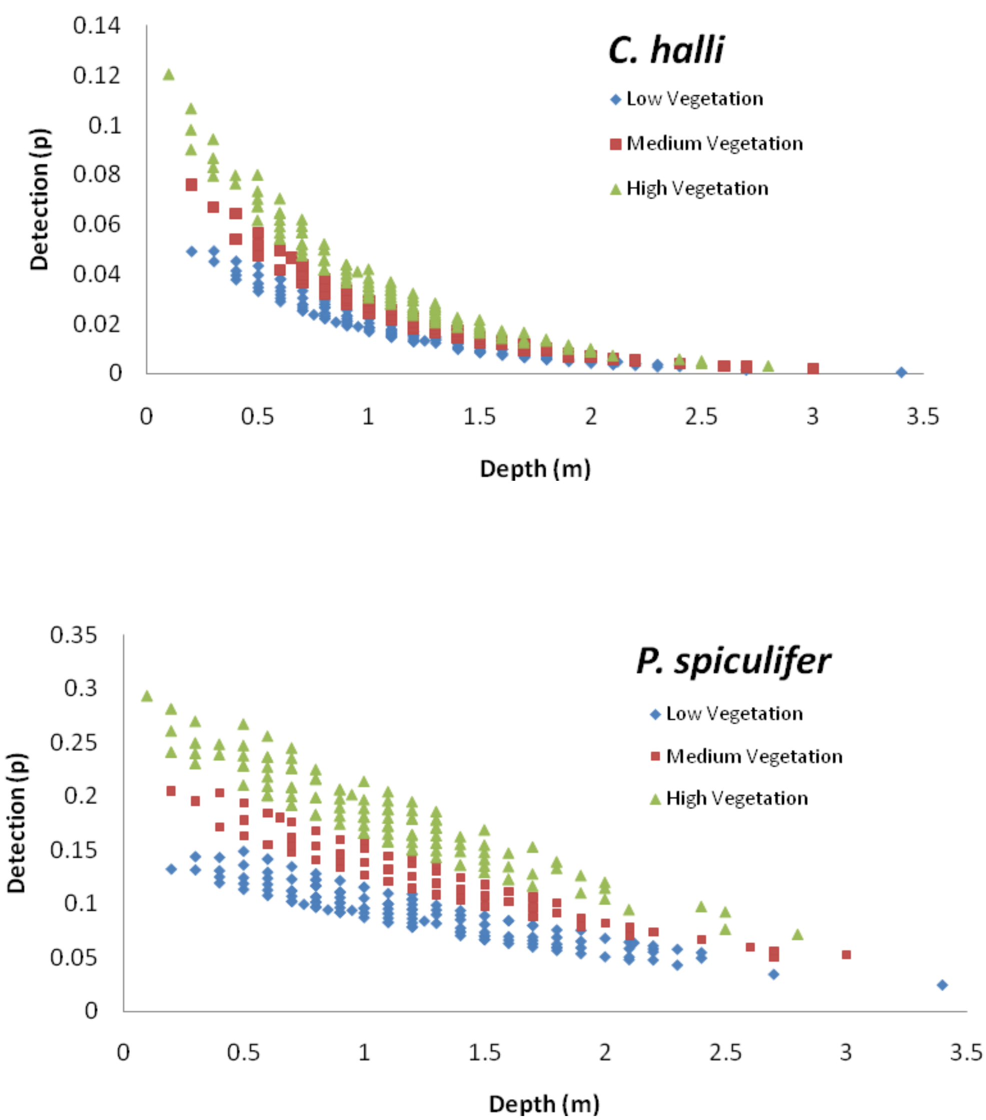
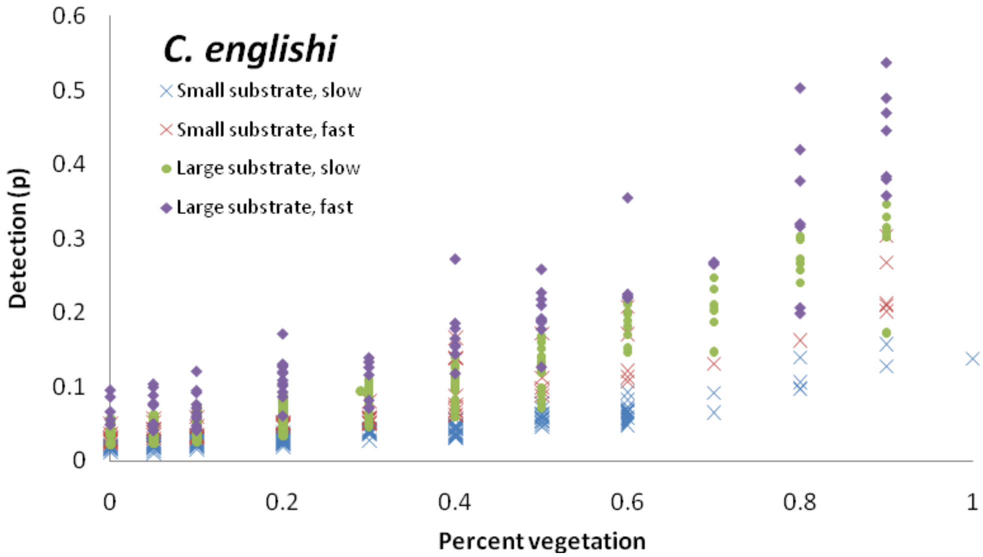


Figure 13.- Relation between detection of two species of crayfishes and depth for three levels of vegetation (low = 0-30%, medium = 31-60% , and high = 61-100% areal coverage) . Data are from 2005 detection models from regulated and unregulated sites in the Tallapoosa River Basin.



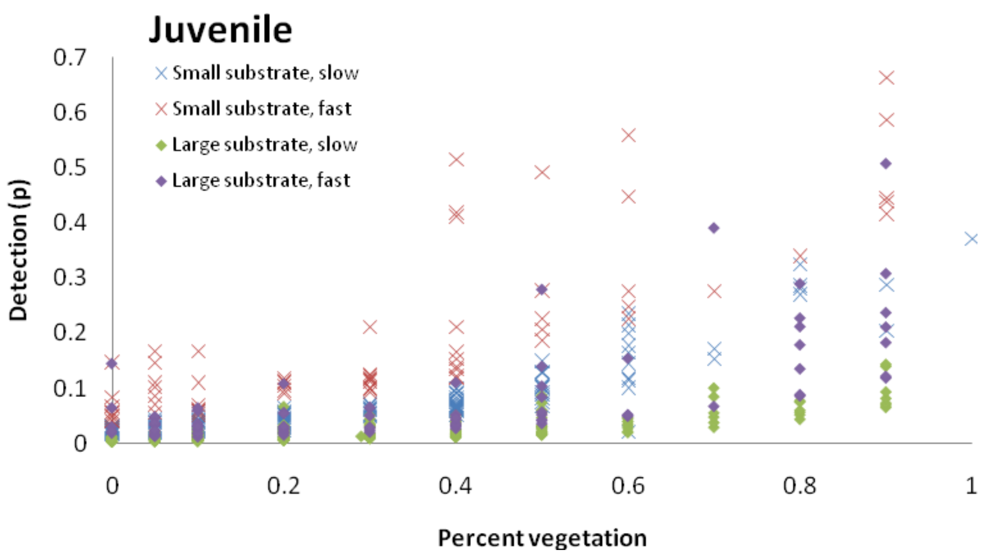


Figure 14.- Relation between detection and vegetation cover (%) combined with the influence of substrate size, slow (<20cm/s) and fast >0.21 cm/s) velocities for *C. englishi* (>14mm carapace length) and all crayfish individuals <14 mm carapace length. Small substrate includes sand and gravel; large substrate including small and large woody debris, cobbles, boulders, and bedrock shelf. Data are from 2007 detection models from regulated and unregulated sites in the Tallapoosa River Basin.

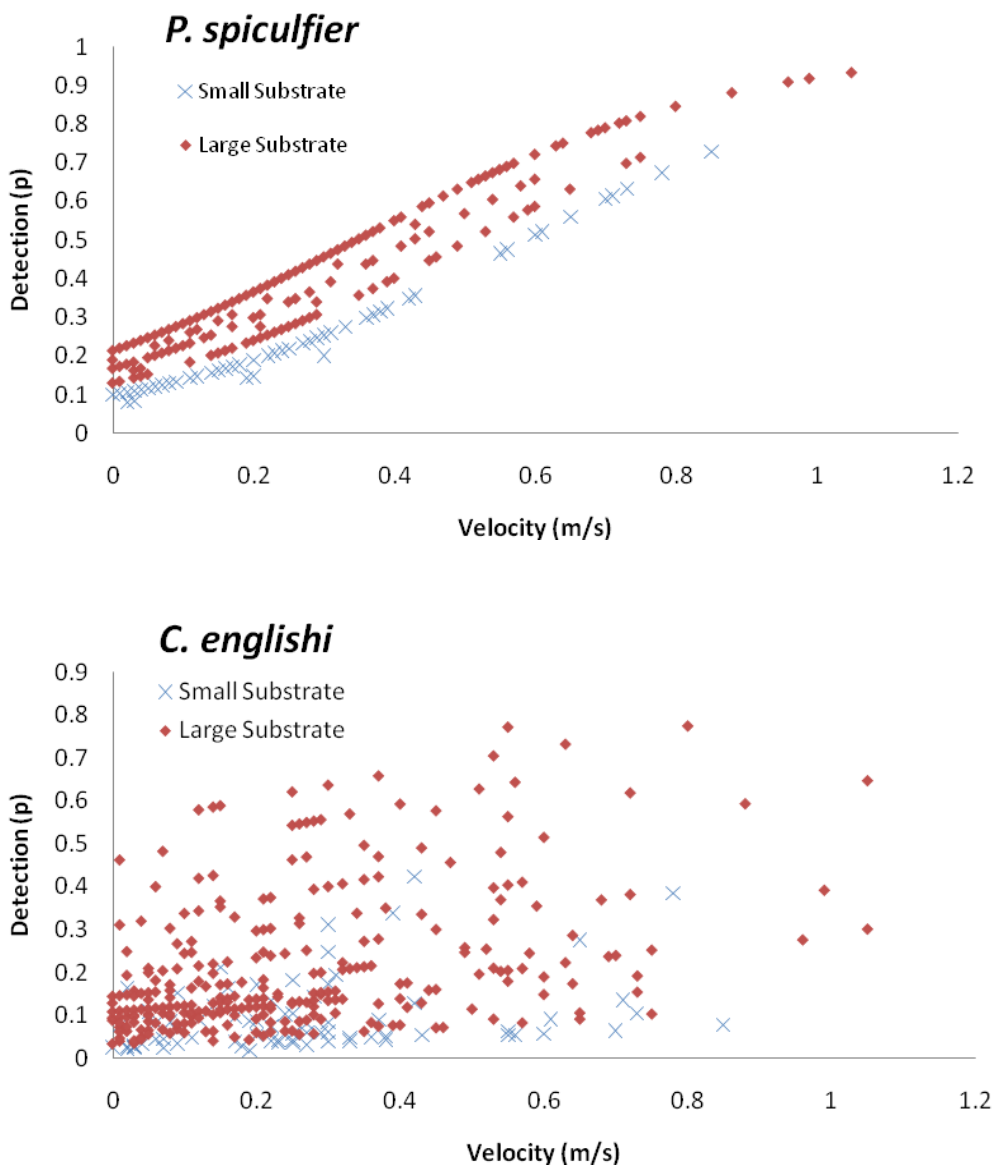


Figure 15.- Relationship between detection and velocity with the influence of substrate on 2 crayfish species . Small substrate includes sand and gravel; large substrate including small and large woody debris, cobbles, boulders, and bedrock shelf. Data are from 2008 detection models from regulated and unregulated sites in the Tallapoosa River Basin. Detection was also influenced by vegetation (not represented) for data plotted in the bottom graph.

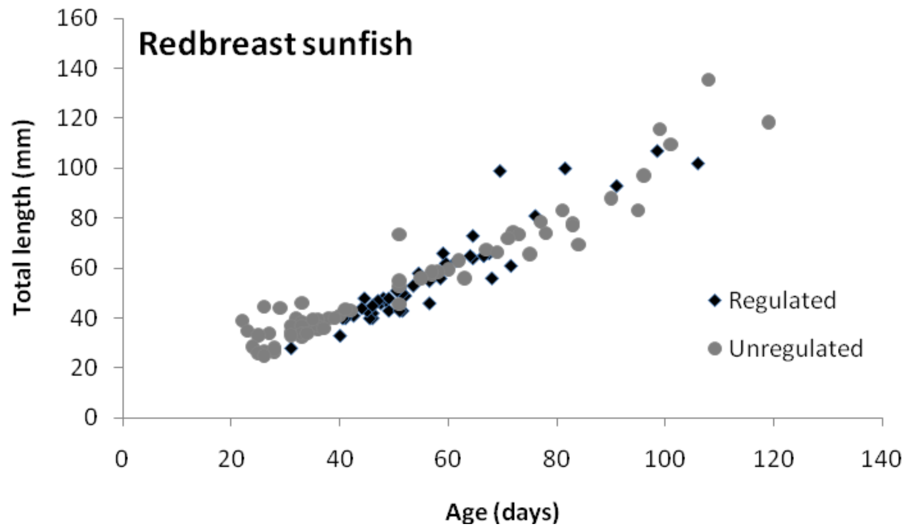


Figure 16.-Plots of total length versus age for redbreast sunfish at regulated and unregulated sites.

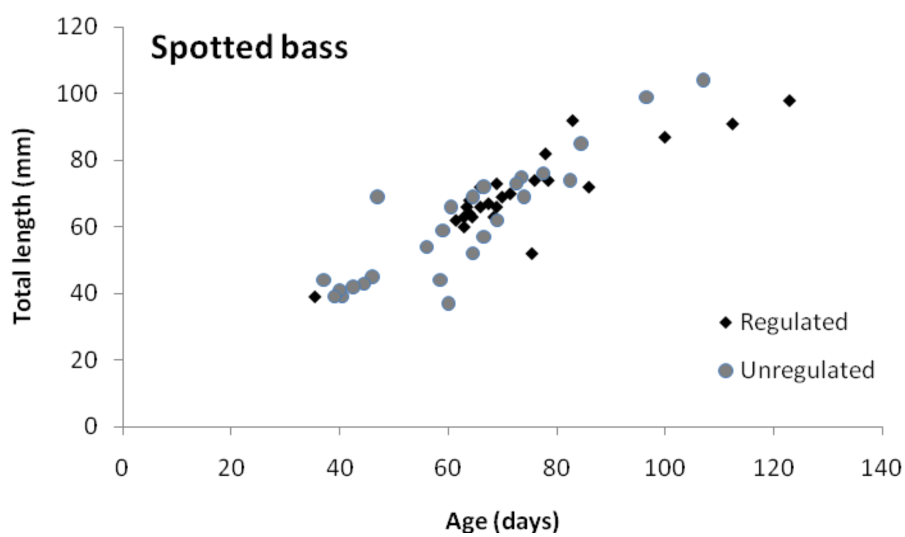


Figure 17.-Plots of total length versus age for spotted bass at regulated and unregulated sites.

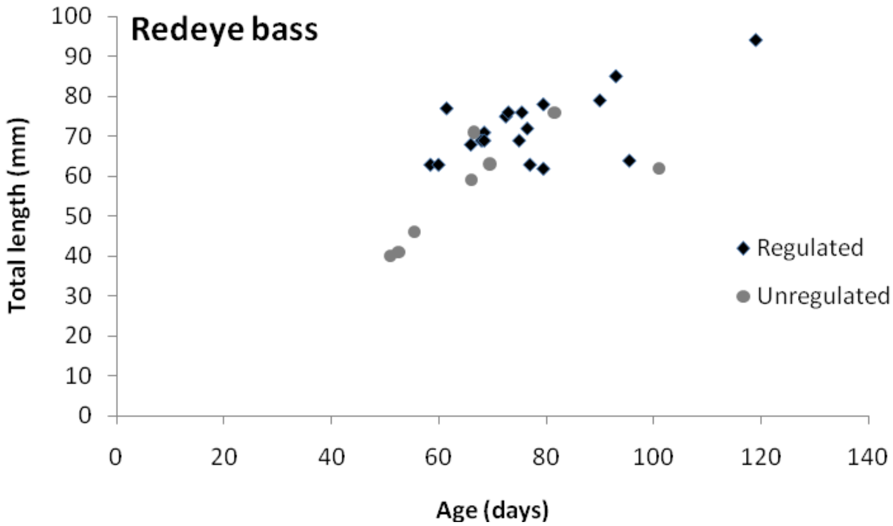


Figure 18.-Plots of total length versus age for redeye bass at regulated and unregulated sites.

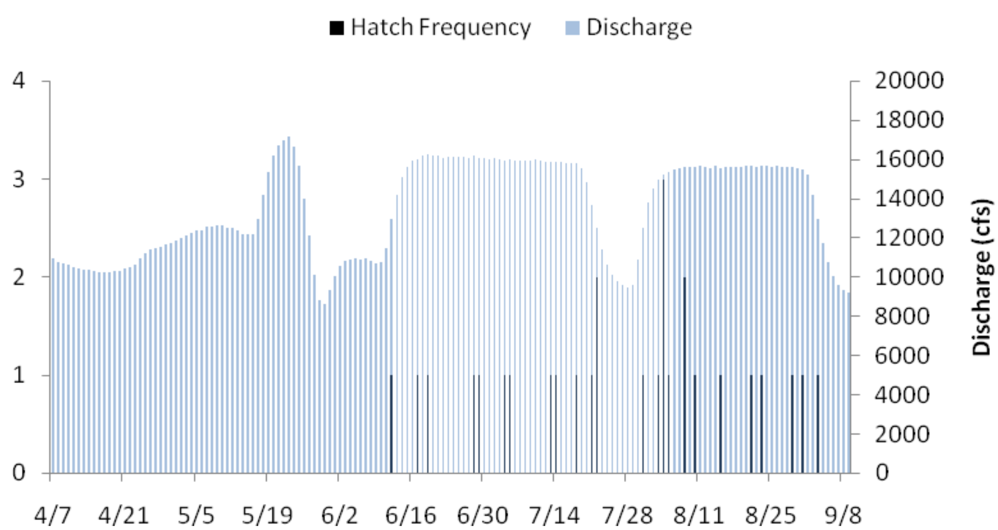


Figure 19.-Hourly flows and redbreast sunfish hatches from 7 April – 9 September 2005.

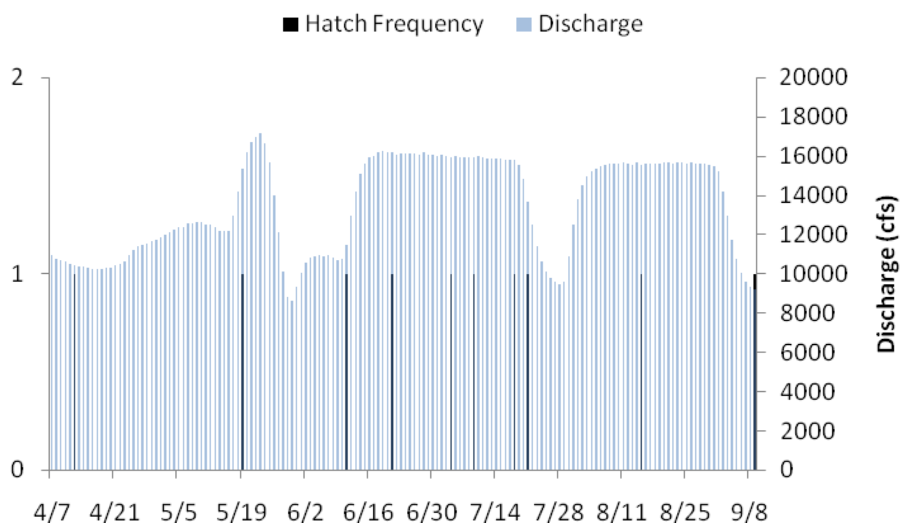


Figure 20.-Hourly flows and spotted bass hatches from 7 April – 9 September 2005.

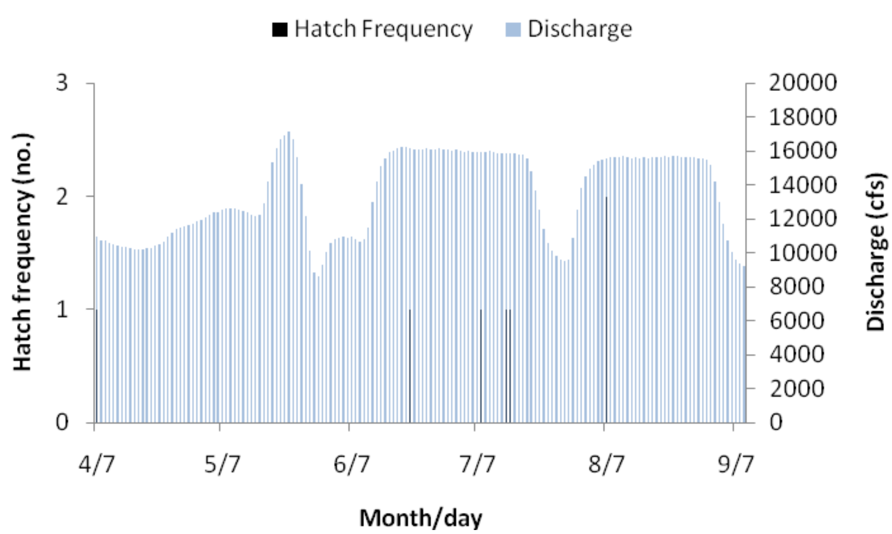


Figure 21.-Hourly flows and redeye bass hatches from 7 April – 9 September 2005.

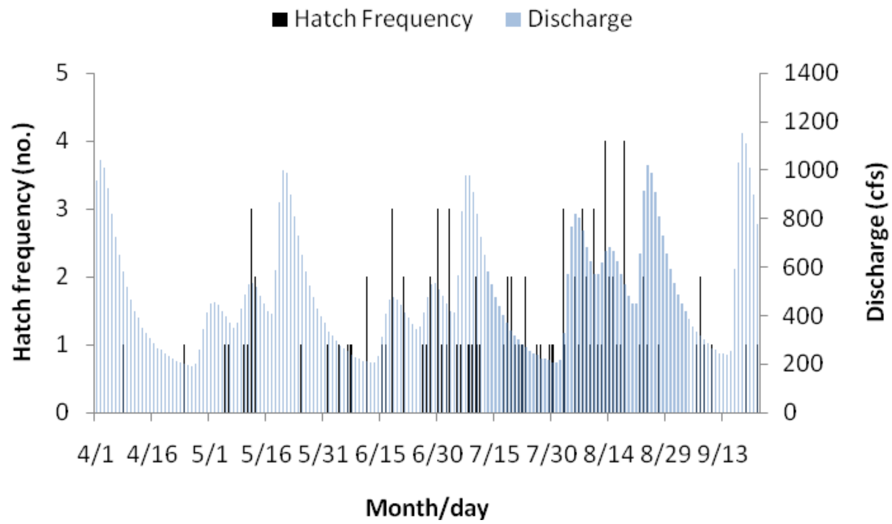


Figure 22.-Hourly flows and redbreast sunfish hatches from 1 April – 22 September 2007.

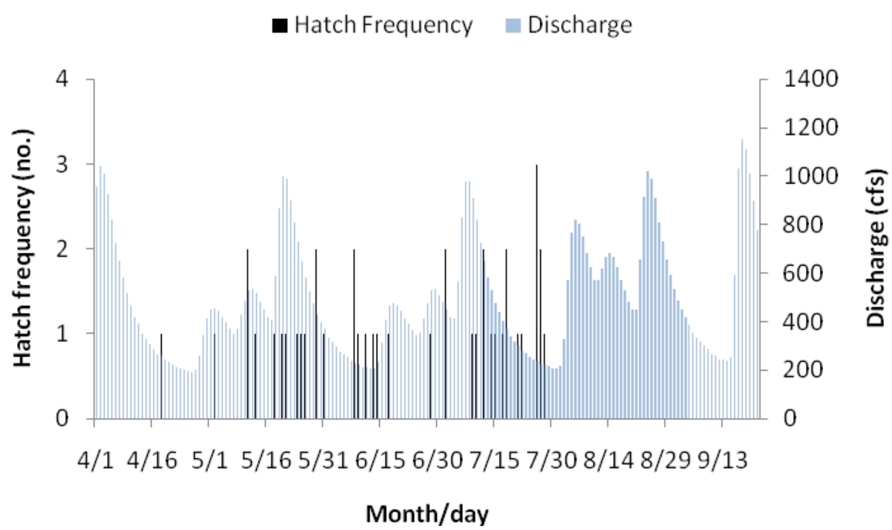


Figure 23.-Hourly flows and spotted bass hatches from 1 April – 22 September 2007.

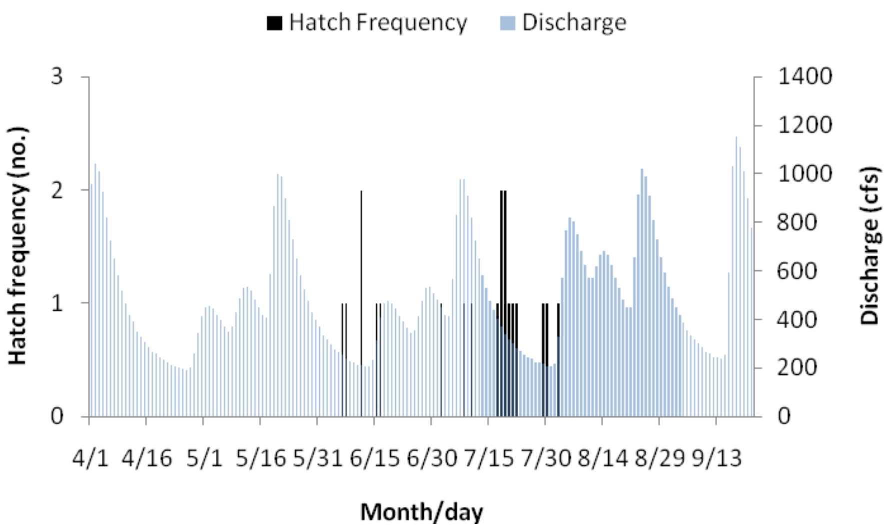


Figure 24.-Hourly flows and redeye bass hatches from 1 April – 22 September 2007.