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Fish and macroinvertebrate assemblage composition and diversity at revetted banks in the Pearl River and the response of these assemblages to a paper mill effluent spill

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FISH AND MACROINVERTEBRATE ASSEMBLAGE COMPOSITION AND DIVERSITY
AT REVETTED BANKS IN THE PEARL RIVER AND THE RESPONSE OF THESE
ASSEMBLAGES TO A PAPER MILL EFFLUENT SPILL

A Thesis

Submitted to the Graduate Faculty of the
Louisiana State University and
Agricultural and Mechanical College
In partial fulfillment of the
Requirements for the degree of
Master of Science

in

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by
Jose Alexander Vazquez
B.S., George Mason University, 2007
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ABSTRACT

The armoring of river banks with riprap can have detrimental effects on lotic ecosystems due to the subsequent alteration of hydrologic regimes; however, evidence suggests that riprap can also increase aquatic diversity in degraded systems. The goal of my study was to determine what impacts riprapped banks have on fish and macroinvertebrate assemblages in the Pearl River, which has a history of anthropogenic degradation. I collected fishes with an electrofishing boat from armored and natural banks at five regions during fall 2011, winter 2012, and summer 2012. I also collected macroinvertebrates with introduced substrates in fall 2011. Richness was analyzed with rarefaction curves, whereas seasonal abundance and evenness were analyzed with ANOVA, and differences in assemblage structure were assessed with PERMANOVA. Fish and macroinvertebrate richness never varied between bank types. However, fish evenness and abundance were higher at riprapped banks during the summer, and fish assemblage composition varied during base flows in the summer and fall. Macroinvertebrate and assemblage structure also varied between bank types in the fall. My results imply that, at base flows, riprapped banks in the Pearl River support unique aquatic assemblages, possibly due to differences in heterogeneous habitat availability.

On August 9, 2011 waste material from a paper mill in Bogalusa, LA was accidentally discharged into the Pearl River, causing anoxic conditions that resulted in complete fish extirpation downstream of the spill's source. ANOVA indicated that, by October 2011, fish species richness and evenness at sites ~10 km downstream of the spill did not differ from sites in undisturbed areas, although richness and evenness at sites ~40 km downstream were still significantly lower at this time. However, by January, richness and evenness at all disturbed and undisturbed sites were similar. PERMANOVA indicated that, despite similarities in richness and

evenness, fish assemblage composition at sites ~10 km downstream of the spill remained significantly different from undisturbed areas until January 2012, while sites ~40 km downstream of the spill were still significantly different by the study's end in July 2012. These results suggest a gradual recovery, with colonization rates related to the proximity of source populations.

CHAPTER 1: INTRODUCTION:

1.1 PULSE AND PRESS DISTURBANCE IN SOUTHEASTERN RIVERS

Aquatic ecosystems found within warmwater rivers and streams in the coastal plain of the southeastern United States host species assemblages that are among the most diverse in the nation (Masters et al. 1998, Warren et al. 2000). Many of the fish and invertebrate species that belong to these systems are endemic to the region, whereas others are important commercially or ecologically (Masters et al. 1998, Ross 2000). Unfortunately, southeastern lotic systems have a long history of anthropogenic press and pulse disturbances that have led to severe decreases in regional ecosystem diversity (Warren et al. 1997, Taylor et al. 2008, Geheber and Piller 2012).

Press disturbances were defined by Lake (2000) as potentially damaging forces applied on an ecosystem that may begin quickly, but thereafter maintain a fairly consistent level of intensity. Most press disturbances are anthropogenic in origin, such as impoundment, watershed development, and in-stream gravel mining (Lake 2000). Most of these disturbances result in decreased native diversity and homogenization of aquatic assemblages (Scott and Helfman 2001), although the mechanisms responsible for decreasing diversity vary depending upon the type of press disturbance and the characteristics of the impacted system (Detenbeck et al. 1992). In several of the press disturbances common to the southeastern United States, including impoundment and gravel mining, decreases in diversity are often due the effects of habitat alteration and increased siltation rates caused by the disturbance (Warren et al. 1997, Hayer and Irwin 2008). Species that are acutely susceptible to these effects include lithophilic spawners and benthic habitat specialists (Piller et al. 2004, Stewart et al. 2005). Declines in these species often occur when coarse gravel or rocky substrates are removed or buried as a result of press disturbances (Berkman and Rabeni 1987, Pusey and Arthington 2003).

Although press disturbances represent continuous perturbations, pulse disturbances are associated with short-term events (Bender et al. 1984, Lake 2000), such as floods, storms, and chemical spills that do not leave residual pollutants in the affected system. The immediate impacts of pulse disturbances can be more severe than those associated with press disturbances, and in some cases can include complete defaunation (Yount and Niemi 1990). Despite the severity of some pulse disturbances, recovery times are often shorter than those following press disturbances (Wallace 1990). This is primarily because pulse disturbances rarely cause habitat alterations (Niemi et al. 1990). Although assemblages can take decades to recover from press disturbances, several studies have documented recoveries from pulse disturbances taking place in just a few weeks (Olmsted and Cloutman 1974, Sheldon and Meffe 1995, Stone and Wallace 2002). In the case of either disturbance type, however, the rate of recovery is dependent upon the characteristics of the impacted system, the type and degree of disturbance, and the availability and proximity of source populations (Niemi et al. 1990). Consequently, rates of recovery can vary greatly between systems and disturbances.

A few studies have examined recolonization and recovery rates among low order stream systems and small scale reaches in the southeastern United States (Meffe and Sheldon 1990, Adams and Warren 2005), but few have examined the effects of pulse disturbances on large rivers in this region. The recovery rates of small stream systems, which are generally very fast, are not necessarily applicable to higher order rivers, which indicates a need for more post-disturbance research in southeastern higher-order, lotic systems.

My two-part study focused on the Pearl River system, a sixth order, 640-km long river in Louisiana and Mississippi that has a prolonged history of press and pulse disturbances (Benke and Cushing 2005), and was composed of two separate research efforts. In the first part of the

study I examined the effects of artificial rocky substrates present at armored banks on aquatic assemblage composition and diversity in the Pearl River. The effects of this substrate are of interest largely because Pearl River assemblages have been severely impacted by declines in naturally occurring coarse and rocky substrates caused by press disturbances, including gravel mining and impoundment (Tipton et al. 2004). Although bank revetments can represent press disturbances themselves, particularly when present in large concentrations (Schmetterling et al. 2001), in highly degraded systems suffering from the effects of multiple severe press disturbances, research indicates that the introduction of small quantities of riprap can be beneficial to local diversity (Dardeau et al. 1995, Shields et al. 1995, White et al. 2009).

The release of 300-375 million liters of black liquor paper mill effluent in August 2011 that affected 235 km of the Pearl River's mainstem and its numerous distributaries (Bart 2012, Brown and Daniels 2012) provided the opportunity to also analyze the effects of a large scale pulse disturbance on fishes in the Pearl River system. This effluent is known to be highly toxic to aquatic taxa (Chapman et al. 1982, Kleynhans et al. 1992), and is thought to have resulted in the near complete extirpation of the fish fauna within the impacted area (Bart 2012). Following the spill I examined the recolonization and recovery of fish assemblages at sample sites in undisturbed and disturbed areas. As a consequence of these events, my thesis focused on identifying the effects that common artificial substrates have on Pearl River communities impacted by press disturbances, and documenting rates of recolonization and recovery by these communities in areas impacted by a severe pulse disturbance.

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CHAPTER 2: THE EFFECTS OF ARTIFICIAL BANK STABILIZATION ON AQUATIC SPECIES ASSEMBLAGES WITHIN A REGULATED COASTAL PLAIN RIVER

2.1 INTRODUCTION

Southeastern coastal plain rivers are characterized by meandering channels; fine, unconsolidated substrates; and diverse aquatic assemblages, particularly among fishes (Hupp 2000, Warren et al. 2000, Benke and Cushing 2005). Unfortunately a long history of anthropogenic disturbance, including impoundment, gravel mining, dredging, snagging, and agricultural and urban runoff, has severely impacted southeastern coastal plain lotic communities (Benke 1990, Richter et al. 1997, Sawyer et al. 2004, Long et al. 2012). Species that are dependent upon rocky or gravel substrates have demonstrated especially large declines in population sizes due to decreases in coarse substrate availability following siltation and in-stream gravel mining (Brown et al. 1998, Tipton et al. 2004, Hayer and Irwin 2008). In several southeastern coastal plain rivers, as naturally occurring gravel substrates are removed or buried, artificially introduced coarse substrates along revetted banks that have been armored with boulder or concrete riprap are becoming more common. Although the effects of revetted banks on lotic communities have been studied in other regions, there is little information available regarding the potential impacts of this artificial substrate on aquatic biota in southeastern coastal plain river systems.

It is well documented that increasing amounts riverine riprap can negatively impact local hydrologic regimes and lotic ecosystems (Li et al. 1984, Buer et al. 1984, Baker et al. 1991) by eliminating channel migration, increasing channel velocity and bank incision, and, in extreme cases, lowering the water table and causing floodplain abandonment (Schmetterling et al. 2001, Fischenich 2003, Gidley et al. 2012). Biological impacts caused by revetted banks largely relate

to the degree of disturbance caused by their presence (Shields et al. 1995). Large concentrations of riprap in a given reach are associated with greater degrees of hydrologic disturbance and reductions in riparian vegetation, which lead to decreases in abundance and diversity of resident aquatic biota (Shields et al. 1995, Schmetterling et al. 2001). Responses to smaller amounts of riprap can vary depending on the ecological and hydromorphologic characteristics of a given system (Dardeau et al. 1995).

Most of the research examining the effects of riprap on aquatic ecosystems originates from the western United States or the Mississippi River Basin. Most of the research from the western United States focuses on the effects of revetted banks on salmonid populations (Fischenich 2003, White et al. 2009). A review of this research by Schmetterling et al. (2001) indicated that, the presence of even moderate quantities of riprap can be detrimental to salmonid populations. However, several studies included in this review, as well as several other studies from the region, reported greater overall fish diversity in riprap habitat (Tabor et al. 1993, Binns and Remmick 1994, Jude and DeBoe 1996). In all of these studies, however, greater diversity was due to increases in non-native species abundances, specifically ictalurids and centrarchids that are native to the southeastern United States and Mississippi River Basin. Although much of the research from the Mississippi River Basin, where these species are native, found similar or higher native fish and macroinvertebrate richness and abundance at revetted banks when compared to natural banks (Burress et al. 1982, Pennington et al. 1983, Litvan et al. 2007), most of these studies were from the highly degraded mainstems of the Mississippi and Missouri rivers. These rivers are confined by levies, deprived of woody debris inputs, and are hydrologically distinct from smaller, southeastern coastal plain rivers, and thus offer limited applicability to southern coastal plain systems.

The overarching goal of this study was to gain a better understanding of the effects that bank revetments have on aquatic communities and diversity in the Pearl River, a southeastern coastal plain river that runs through Mississippi and Louisiana. To accomplish this, I documented seasonal fish abundance, evenness, richness and assemblage structure at both revetted and natural banks in the Pearl River. I also examined similar measures in macroinvertebrate communities, which are considered to be severely understudied within the Pearl River and surrounding region (Benke and Cushing 2005).

2.2 METHODS

2.2.1 Site Description

The Pearl River is a sixth order, Gulf of Mexico coastal plain river that begins in east-central Mississippi and empties into Lake Borgne, which connects to the Gulf of Mexico through the Mississippi Sound. The river drains a catchment of $\sim 22,000 \text{ km}^2$, and is approximately 640 km long (Benke and Cushing 2005). The Pearl River is characterized by large amounts of woody debris as well as sand, silt and gravel substrates, although gravel substrate availability has declined due to anthropogenic influences (Piller et al. 2004, Geheber and Piller 2012). The river was impounded north of Jackson, Mississippi at the Ross Barnett Reservoir in 1964, resulting in regulated flows through downstream portions of the river. Annual mean flows in lower sections of the river have averaged $262 \text{ m}^3/\text{s}$ since 2000 (USGS gauge 02489500); however, daily flows are highly variable and dependent upon seasonal rainfall. Peak annual flow usually occurs between December and May, although as in other southeastern coastal plain systems, flows typically remain elevated during this entire period due to a combination of high precipitation and low transpiration rates (Hupp 2000). During the summer and fall months, flows are commonly near base flow levels.

The banks and riparian zones of the Pearl River are largely undeveloped except for areas near cities and towns, which account for less than two percent of the basin's total area (Benke and Cushing 2005). Other than Jackson, Mississippi, major settlements on the Pearl River include several towns and the city of Picayune in Mississippi as well as the cities of Bogalusa and Slidell in Louisiana. Since the impoundment of the river, houses and hunting camps have become increasingly common along banks near these population centers. Most of these banks are composed of unconsolidated sand and clay sediments, and are thus highly susceptible to erosion. In order to minimize property loss, landowners are increasingly turning to bank stabilization methods. Although bank stabilization methods vary between properties, most of the stabilized banks are armored with either boulder riprap or similarly sized fragments of waste concrete. Currently, bank stabilization structures cover < 1.5% of the Pearl River's banks, and are not limited to outside bendways as is common in many other systems (unpublished personal data).

2.2.2 Sample Site Selection

Sampling was limited to the lower Pearl River as defined by the Mississippi Department of Environmental Quality, which included all areas below and including Columbia, MS (MDEQ 2000). I selected five regions (Figure 2.1) that were separated by at least 10 river kilometers and bordered towns where armored banks were known to be present. I identified all of the revetted banks longer than 20 m in these regions, and then randomly selected three banks from each region. In order to reduce variation resulting from different substrate types, site selection was limited to banks stabilized with concrete or boulder riprap. After selection, I determined the length of each bank and then measured depth, current velocity, and wetted width at five equidistant points along three evenly spaced transects that were perpendicular to the revetted

bank. I used the results of these surveys in conjunction with satellite imagery to select three natural banks within each region that reassembled the channel morphology and hydrologic conditions present at the region's three revetted banks. Each natural bank was located at least one kilometer away from any armored banks, but was within 10 km of its corresponding revetted bank. Electrofishing transects at each sample site comprised the full length of each revetted bank or an equivalent distance at paired natural banks.

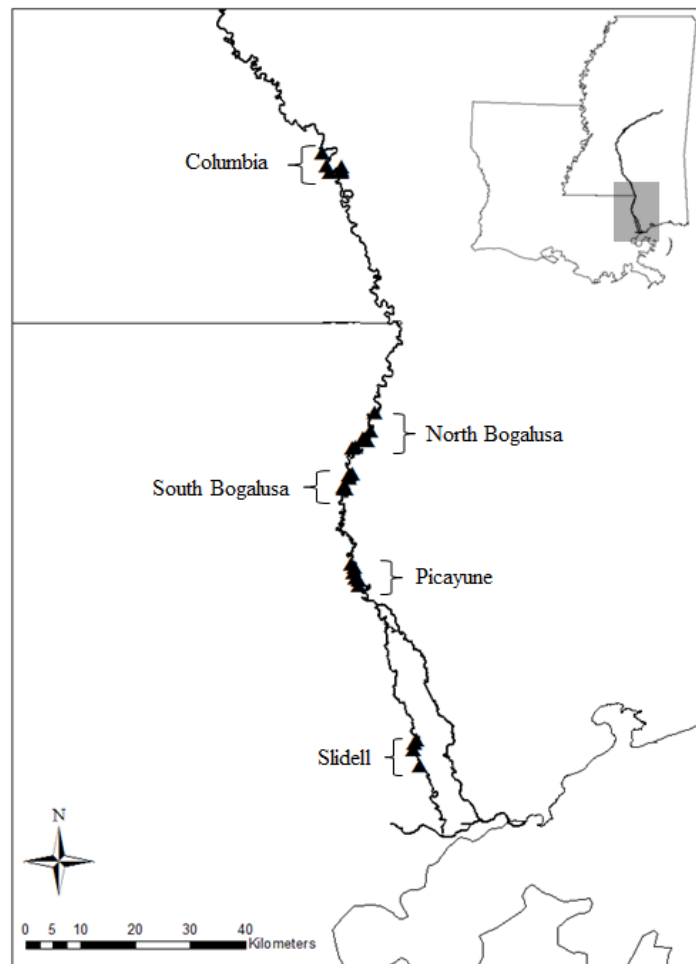


Figure 2.1: Map of the Pearl River showing individual sample sites (triangles) and sampled regions (text).

2.2.3 Fish Sampling

I sampled fish with boat-electrofishing, which has been shown to effectively sample both revetted and natural bank fish communities (White et al. 2009). Electrofishing was conducted

from an aluminum hull electrofishing boat along the full length of each transect, by sampling crews consisting of two netters and a driver. We performed sampling using 60 Hz pulsed DC, and I adjusted voltage and amperage as needed based on river conductivity on the day of sampling, usually maintaining a current of approximately 4.5 - 6.5 amps. I recorded the total length of all captured fish and identified them to the species level in the field whenever possible. Specimens that could not be accurately identified in the field were transferred in an ice slurry to the laboratory for identification (LSU AgCenter IACUC A2011-16).

To assess seasonal effects on fish assemblage composition, I electrofished revetted and natural banks during three periods: 1) in September and October 2011 during base flows that characterize the fall; 2) from January through March 2012 during temperatures and elevated flows representative of the winter and early spring; and 3) in May through June 2012 during summer base flows. When possible, I sampled all sites and regions during each sampling season. However, unusually low water during fall 2011 and an August 2011 paper-mill effluent spill either prevent data collection or led to the exclusion of data from Picayune and Columbia during fall 2011 and from Slidell during fall 2011 and winter 2012.

2.2.4 Benthic Macroinvertebrate Sampling

I sampled benthic macroinvertebrates at sites in South Bogalusa, North Bogalusa and Columbia in the fall of 2011 using introduced concrete and wood samplers, which have been reported to acquire representative assemblages of that habitat when deployed on the benthos (Rosenberg and Resh 1982). Wood and concrete were chosen as substrates because they are the predominant macroinvertebrate habitat at natural and revetted banks in southeastern lotic systems (Shields et al. 1995, Benke and Wallace 2003). Samplers consisted of 45.7-cm long x 17.8-cm wide cylinders constructed of 1.9-cm plastic mesh containing either concrete or wood

substrate. Rock samplers contained 5.45 kg of 3.68 cm wide concrete tile that was broken into pieces that approximated concrete riprap found on armored banks in the Pearl River. Wood samplers contained a total of six branches from three tree species [American sycamore (*Platanus Occidentalism*), Black Willow (*Salix nigra*), and American sweetgum (*Lliquidambar styraciflua*)] that vegetation surveys (unpublished personal data) indicated were dominant species along the banks of the Pearl River. All branches were between two and five centimeters in diameter and were cut to be 38 cm long. Branches were originally harvested from live, untreated trees in Bogue Chitto State Park, and were dried at 55°C for 72 hours before deployment.

I randomly selected two pairs of natural and revetted banks from the sites that were previously sampled for fish in each of three regions. I deployed one rock and one wood sampler halfway between the midpoint and the upstream end of each selected site's electrofishing transect. Another pair was deployed halfway between the midpoint and the downstream end of each transect, resulting in two rock and two wood samplers at each site. I deployed samplers 0.5 meters below mean base flow to ensure complete submersion during the duration of the study. Samplers were deployed for a minimum of six weeks, which Lamberti and Resh (1985) and Kaller and Kelso (2006) indicated is a sufficient period of time for introduced rock and wood substrates to develop representative macroinvertebrate assemblages. Samplers in Columbia and South Bogalusa were deployed between October 31, 2011 and November 8, 2011, and were retrieved six weeks after deployment. Samplers in North Bogalusa were deployed on November 18, 2011; and were retrieved nine weeks after their initial deployment due to winter storms. Samplers were retrieved by carefully surrounding and removing cylinders from the water column with a 250-µm mesh bag or net so as to minimize invertebrate loss. The presence of any abiotic

materials in each sampler, including sand or mud, was noted at this time, and the contents of each retrieved cylinder were transferred to plastic bags containing 95% ethanol for transport and storage.

Macroinvertebrates were sorted by rinsing and retaining organisms on a 250- μ m sieve and by manual extraction under magnification. I calculated the surface area of wood substrates with calipers and determined the surface area of rocky substrates with the foil method detailed in Bergey and Getty (2006). High densities of macroinvertebrates were recovered from the artificial substrates; therefore, I used a gridded Caton Tray described by Caton (1991) to obtain a 20% subsample of each sample for identification. Because regional keys were lacking, all subsampled macroinvertebrates were identified to family under magnification with keys found in Merritt et al. (2008) and Smith (2001). Family level was selected to reduce errors associated with misidentification of early instar insects and problems associated with mixing levels of taxonomic precision during analyses (Jones 2008).

2.2.5 Environmental Parameters

After sampling a bank for aquatic biota, I used a handheld YSI *in situ* water quality monitor to record specific conductance (mS/cm), water temperature ($^{\circ}$ C), dissolved oxygen concentration (mg/l), turbidity (NTUs) and pH at the sample site. All water quality measurements were taken one meter away from the wetted edge at the midpoint of each electrofishing transect at a depth of 0.5 m. In addition to measuring water quality, I also used a Sontek Acoustic Doppler Velocimeter to measure surface current velocity along a transect perpendicular to each sampled bank's midpoint. Current velocity was measured one meter from each bank's wetted edge and at three equidistant points along the perpendicular transect. I also

measured current velocity 0.5 m above the substrate at the location of each artificial substrate sampler during deployment and retrieval.

2.2.6 Statistical Analysis

Fish species richness at revetted and natural banks and macroinvertebrate family richness at each bank type as well as on rock and wood substrates were compared using sample-based rarefaction (EstimateS version 8.2.0). These curves provide a means of comparing taxonomic richness between different treatment levels when sample sizes are unequal (Gotelli and Colwell 2001, Colwell et al. 2012). I was able to conduct hypothesis tests on these curves at $\alpha = 0.05$ by comparing the overlap of unconditional 84% confidence intervals between treatments as described by Gotelli and Colwell (2011), and was able to detect significant differences in richness between bank and substrate types when confidence intervals no longer overlapped.

I compared fish abundance and evenness at revetted and natural banks with analysis of variance (ANOVA) nested by season (PROC MIXED; SAS version 9.3, SAS Institute Inc.). Fish abundance was calculated based on catch per unit effort (CPUE; fish per second) from electrofishing samples. To compensate for differences in transect lengths between sample sites, I calculated evenness with Hurlbert's evenness index, which is robust to differences in sample size (Beisel et al. 2003, Olszewski 2004). ANOVA tests were nested within season to account for differential life history-based fish habitat use among seasons, and categorical covariables indicating each site's sampling region were included in models to account for regional variation. All assumptions of ANOVA were assessed during analyses. I followed ANOVA with Tukey-Kramer pairwise comparisons to examine differences in mean evenness and CPUE between the two bank types within each season.

Differences in macroinvertebrate abundance and family level evenness between natural and revetted banks as well as between rock and wood substrates were compared with ANOVA's blocked by sampling region (PROC MIXED; SAS version 9.3). Hurlbert's index was used again to compare evenness, and abundance was defined as the number of macroinvertebrates per square meter of substrate.

Comparisons of fish and macroinvertebrate assemblage composition between different bank and substrate types were made with non-parametric permutational multivariate analysis of variance (PERMANOVA; R version 2.15, Vegan Package, R Foundation for Statistical Computing) based on 9999 permutations. Permutations of electrofishing data were constrained within each season, and sampling regions were identified and included in the model as a single categorical covariable in order to account for differences in seasonal and regional sampling effort. Permutations of macroinvertebrate data were constrained within each region to account for regional variation. Analysis of fish data was performed with the Morisita-Horn distance matrix in order to compensate for differences in sample sizes resulting from different transect lengths (Chao et al. 2006, Jost et al. 2011). Macroinvertebrate analysis was based on a Bray-Curtis dissimilarity matrix that was calculated with family density data. Following analysis of combined fish assemblage data, separate PERMANOVA tests were conducted on each individual sampling season in order to determine if differences in species assemblages between banks were persistent throughout the year.

Whenever PERMANOVA tests indicated significant differences existed in fish or macroinvertebrate assemblage composition between treatments, I performed individual non-parametric ANOVA's on fish species or macroinvertebrate families with seasonal CPUE or family density data in order to determine how composition varied. These follow-up tests were

conducted at $\alpha = 0.05$ with the Wilcoxon test statistic (PROC NPAR1WAY; SAS version 9.3). Follow up tests were conducted at $\alpha = 0.05$ based on the principle of protected tests in MANOVA (Stevens 2002), which is conceptually similar to PERMANOVA.

I analyzed environmental variables by first using parametric and, when indicated to be necessary by Kruskal-Wallis tests, non-parametric ANOVA's nested by season to determine if any measured water quality or current velocity variables differed between revetted and natural banks. Most velocity measurements were averaged into a single value for each site before analyses, although banks velocities were poorly correlated with other velocity measurements, and were thus retained as a separate variable. When ANOVA's indicated that at least one environmental variable differed significantly between revetted and natural banks, I then used canonical correspondence analysis (CCA; R version 2.15, Vegan package) to relate environmental variables that differed between bank types to corresponding fish or macroinvertebrate assemblage data. Monte Carlo permutational tests conducted within CCA were used to determine which, if any, tested environmental variables influenced assemblage structure. Following CCA, I related environmental variables that influenced assemblage structure to individual macroinvertebrate family and fish species abundances with Spearman rank correlation coefficients and corresponding significance tests (PROC CORR; SAS version 9.3). Correlation analysis was limited to macroinvertebrate families and fish species represented by at least 20 individuals.

2.3 RESULTS

2.3.1 Fish Assemblages

I collected 4,983 fish representing 48 species in 58 samples during the study, with 2,856 fish representing 39 species captured at revetted banks and 2,127 fish representing 39 species

captured at natural banks. Longear sunfish (*Lepomis megalotis*), blacktail shiners (*Cyprinella venusta*) and bluegill (*Lepomis macrochirus*) were the three most common species at both bank types, representing 80.1% of the fish at revetted banks and 78.7% of those at natural banks (Table 2.1). I captured four specimens [three southeastern blue suckers (*Cycleptus meridionalis*) and a single silverjaw minnow (*Ericymba buccata*)] representing two of the eleven species in the Pearl River that are considered to be threatened by Louisiana or Mississippi state agencies. I also captured multiple highfin carpsuckers (*Carpoides velifer*), which are under consideration for threatened status by the state of Louisiana, although this species is known to be locally abundant in the Pearl River (Ross 2000, Bart and Rios 2003).

Table 2.1: Numerical abundance (and relative abundance) of species comprising > 1.0% of fish collected at natural or revetted banks.

Species	Natural Banks	Revetted Banks
Black tail shiner	1152 (54.2)	825 (28.9)
Longear sunfish	387 (18.2)	1105 (38.7)
Bluegill	134 (6.3)	358 (12.5)
Bullhead minnow	76 (3.6)	146 (5.1)
Channel catfish	55 (2.6)	59 (2.1)
Mississippi silvery minnow	55 (2.6)	12 (0.4)
Emerald shiner	50 (2.4)	22 (0.8)
Spotted bass	34 (1.6)	84 (2.9)
Large mouth bass	27 (1.3)	21 (0.8)
Other species	151 ^a (7.4)	223 ^a (7.8)

^a 40 species

Rarefaction curves based on pooled seasonal data indicated that there was not a significant difference between species richness at revetted and natural banks (Figure 2.2). Curves plotted with data from individual seasons confirmed species richness did not differ significantly between bank types during any point in the study.

Total abundance, both numerically and as represented by CPUE (Figure 2.2), was generally greater at revetted banks than at natural banks, particularly during the summer season

(revetted N = 2091, natural N = 1469). However, these differences were not significant at $\alpha = 0.05$ (Figure 2.2), although the summer sampling was close ($P = 0.072$, $t = -1.85$, $DF = 44$). In contrast, evenness varied significantly between natural and revetted banks ($P = 0.008$, $F = 5.48$, $DF = 48$), although this difference reflected greater species evenness at revetted banks during summer ($P = 0.005$, $t = -2.95$, $DF = 44$), but not during fall or winter (Figure 2.3). Examination of raw abundances indicated that differences in mean abundance and evenness between bank types during the summer were due to higher proportions of moderately common species such as bluegill, spotted bass (*Micropterus punctulatus*), channel catfish (*Ictalurus punctatus*) and bullhead minnows (*Pimephales vigilax*) at revetted banks.

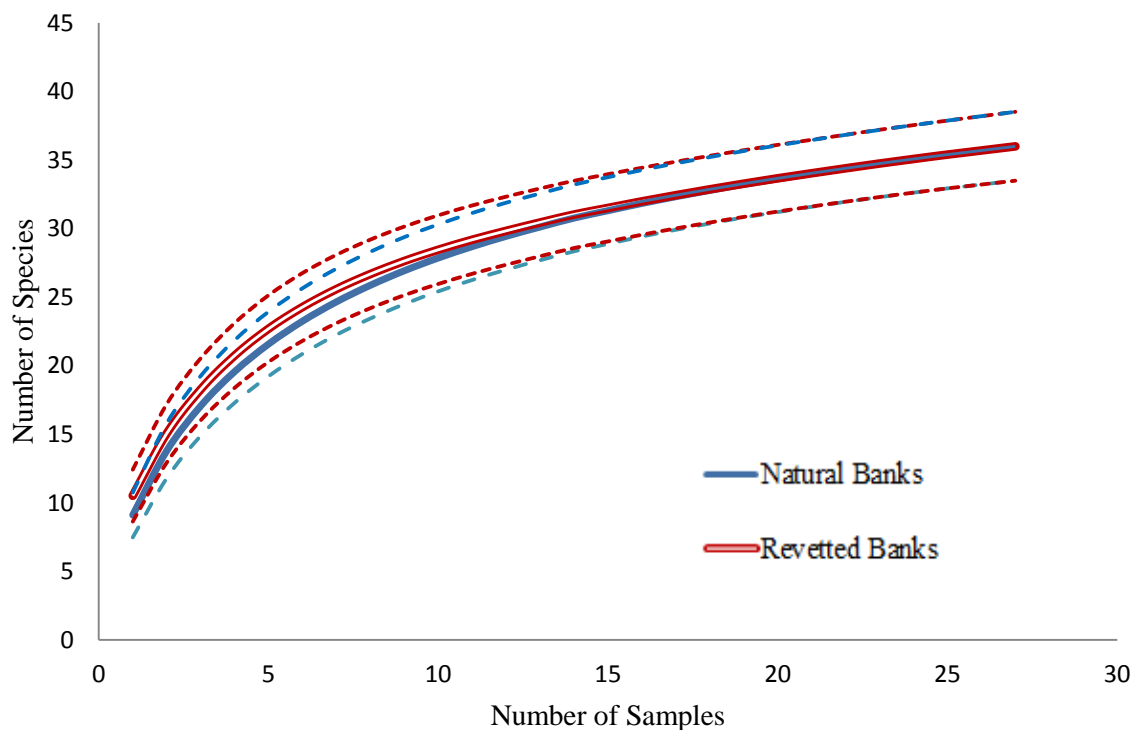


Figure 2.2: Rarefaction curves depicting fish species richness at revetted and natural banks. Dashes represent 84% confidence intervals. Complete overlap between the confidence intervals implies similarity between bank types ($P > 0.05$).

PERMANOVA indicated significant differences between fish assemblages at revetted and natural banks ($P < 0.001$, $F = 12.19$, $DF = 48$), with seasonal PERMANOVA tests indicating

that differences in fish assemblage structure between bank types were limited to the fall and summer sampling periods (Table 2.2). Non-parametric ANOVA based on species CPUE data

Table 2.2: Results from Seasonal PERMANOVA's testing differences in fish assemblage structure between bank types.

Season	DF	Mean Square	F	P value
Fall	1, 7	0.206	6.74	0.021
Winter	1, 12	0.058	0.79	0.548
Summer	1, 22	0.495	16.47	< 0.001

indicated that bullhead minnows, highfin carpsuckers, longear sunfish, bluegill and spotted bass were significantly more abundant at revetted banks during summer (Table 2.3). Many of the highfin carpsuckers and bullhead minnows captured at this time possessed tubercles or ripe eggs, indicating that these species may have been using revetted banks as spawning habitat. Gizzard shad (*Dorosoma cepedianum*) was the only species that demonstrated a significant preference for natural banks during the summer season (Table 2.3), but several species, including spotted gar, Mississippi silvery minnows, and blacktail and emerald shiners (*Cyprinella venusta*) were noticeably more abundant at these sites. Longear sunfish was the only species that exhibited differences in abundance among bank types in the fall, being more common at revetted banks ($P = 0.016$, $F = 5.77$, $DF = 9$), although this is likely because I was only able to sample 10 sites during this season (versus 28 in the summer) due to abnormally low river stages during fall 2011. Several species that demonstrated preferences for either revetted banks or natural banks in other studies, including largemouth bass (*Micropterus salmoides*), smallmouth buffalo (*Ictiobus bubalus*), blacktail redhorse (*Moxostoma poecilurum*), American eel (*Anguilla rostrata*), and freshwater drum (*Aplodinotus grunniens*) (Pennington et al. 1983, Killgore and Hoover 1992, White et al. 2009), demonstrated no apparent bank preference during any season in this study.

2.3.2 Benthic Macroinvertebrate Assemblages

I collected and identified 12,854 benthic macroinvertebrates representing 37 families during the course of the study, with 7,731 (60.1%) organisms collected at revetted banks and 5,123 (39.9%) collected at natural banks. Wood samplers contained 8,243 (64.1%) of these macroinvertebrates, whereas rock samplers contained 4,611 (35.9%). Individuals within the family Chironomidae made up 71.7% of the total number of organisms sampled, although the families Hydropsychidae, Heptageniidae, and Perlidae were also common among all bank and substrate types (Table 2.4).

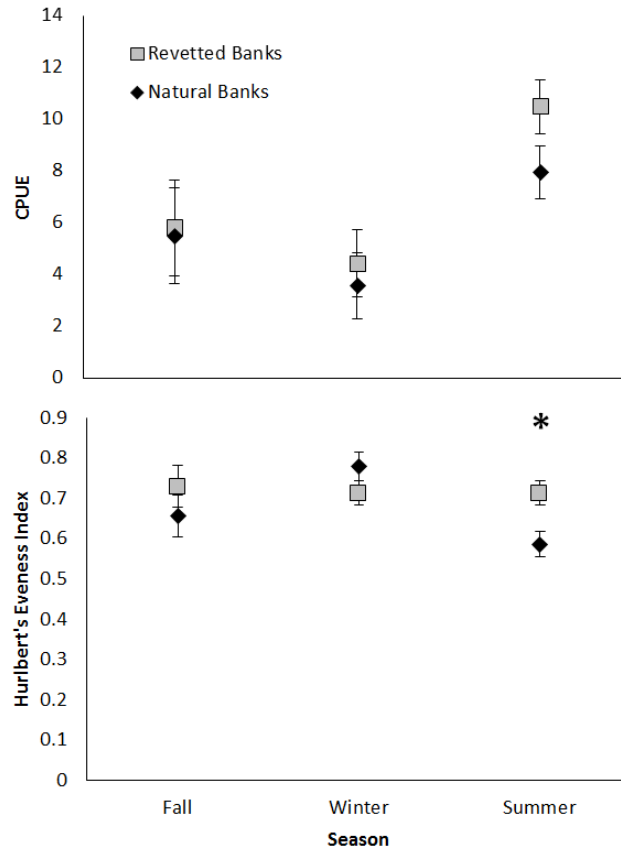


Figure 2.3: Seasonal mean fish abundance (CPUE) and evenness (Hurlbert's evenness index) at revetted and natural banks. Greater values are associated with increased abundance and evenness. Tick marks represent standard error. Asterisks indicate significant differences between revetted and natural bank means within the indicated season ($P < 0.05$).

Rarefaction curves indicated that benthic macroinvertebrate family richness was not significantly different between revetted and natural banks or between wood and rock substrates (Figure 2.4). This is reflected by the fact that both bank types and substrate types each contained 30 of the 37 families sampled during the study.

Table 2.3: Non-parametric ANOVA results indicating fish species with significant differences in abundance between revetted and natural banks during the summer sampling season. Kruskal-Wallis test statistic used.

Species	Bank Preference	χ^2	P value	Abundance	
				Revetted	Natural
Bluegill	Revetted	5.07	0.024	283	82
Bullhead minnow	Revetted	7.93	0.005	112	61
Highfin carpsucker	Revetted	6.07	0.018	21	5
Longear sunfish	Revetted	6.16	0.013	854	263
Spotted bass	Revetted	5.90	0.043	44	17
Gizzard shad	Natural	5.79	0.016	0	7

Table 2.4: Numerical abundance (and relative abundance) of macroinvertebrate families representing > 0.3% of those collected at natural and revetted banks, and rock and wood substrates.

Family	Natural		Revetted		Rock		Wood	
Chironomidae	3838	(74.92)	5382	(69.62)	3051	(66.17)	6169	(74.84)
Hydropsychidae	455	(8.88)	1309	(16.93)	749	(16.24)	1015	(12.31)
Heptageniidae	394	(7.69)	565	(7.31)	351	(7.61)	608	(7.38)
Perlidae	90	(1.76)	185	(2.39)	125	(2.71)	150	(1.82)
Leptoceridae	76	(1.48)	34	(0.44)	50	(1.08)	60	(0.73)
Ceratopogonidae	53	(1.03)	33	(0.43)	37	(0.80)	49	(0.59)
Polycentropodidae	24	(0.47)	52	(0.67)	29	(0.63)	47	(0.57)
Coengarionidae	45	(0.88)	18	(0.23)	43	(0.93)	20	(0.24)
Ephemeridae	60	(1.17)	1	(0.01)	55	(1.19)	6	(0.07)
Hydrachnidia	5	(0.10)	51	(0.66)	23	(0.50)	33	(0.40)
Simuliidae	12	(0.23)	27	(0.35)	22	(0.48)	17	(0.21)
Other Families	71	(1.39)	74	(0.96)	76	(1.65)	69	(0.84)

ANOVA indicated that benthic macroinvertebrate abundance was greater at revetted banks when compared to natural banks ($P = 0.006$, $F = 8.37$, $DF = 41$) as well as on wood substrates when compared to rock substrates ($P > 0.001$, $F = 13.79$, $DF = 41$; Figure 2.5a). Further examination indicated that the greatest macroinvertebrate densities occurred within wood

samplers located at revetted banks, even when chironomids were omitted from the data (Table 2.5), whereas the lowest macroinvertebrate densities tended to occur within rock samplers located at natural banks (Table 2.5). Macroinvertebrate family evenness was also greater at revetted banks than at natural banks ($P = 0.029$, $F = 5.14$, $DF = 41$; Figure 2.5b), although evenness was marginally greater on rock substrates than on wood substrates ($P < 0.056$, $F = 3.87$, $DF = 41$). Lower evenness values among natural banks and wood substrates were primarily the result of large proportions of Chironomidae relative to other families. When chironomids were excluded from the analysis, evenness remained higher on rock substrates compared to wood substrates ($P = 0.005$, $F = 8.72$, $DF = 43$). However, evenness at natural banks became higher than evenness at revetted banks ($P = 0.016$, $F = 6.30$, $DF = 43$). Further examination indicated that this proportional increase in evenness at natural banks was a result of a lower abundance of Hydropsychidae at these sites. Increased evenness at natural banks did not correspond to higher abundances of uncommon or rare families compared to revetted banks.

Table 2.5: Abundance and density of macroinvertebrates at each substrate/bank combination. Parenthetical values represent corresponding abundances omitting Chironomidae.

Bank Type	Substrate Type	Numerical Abundance		Density per m ²	
Revetted	Wood	4,977	(1,304)	11,801	(3,191)
Revetted	Rock	2,754	(1,045)	6,661	(2,532)
Natural	Wood	3,266	(770)	7,619	(1,829)
Natural	Rock	1,857	(515)	4,078	(1,130)

The PERMANOVA analyses indicated significant differences in the composition of macroinvertebrate assemblages found on wood and rock substrates ($P < 0.001$, $F = 5.23$, $DF = 45$) as well as between those recovered from natural and revetted banks ($P < 0.001$, $F = 6.43$, $DF = 45$). Families that were more common on rock substrates included Coengarionidae and Ephemeridae, while Heptageniidae and Chironomidae were significantly more common on wood substrates (Table 2.6). Hetageniidae, Hydropsychiidae, Hydrachnidia, Perlidae and

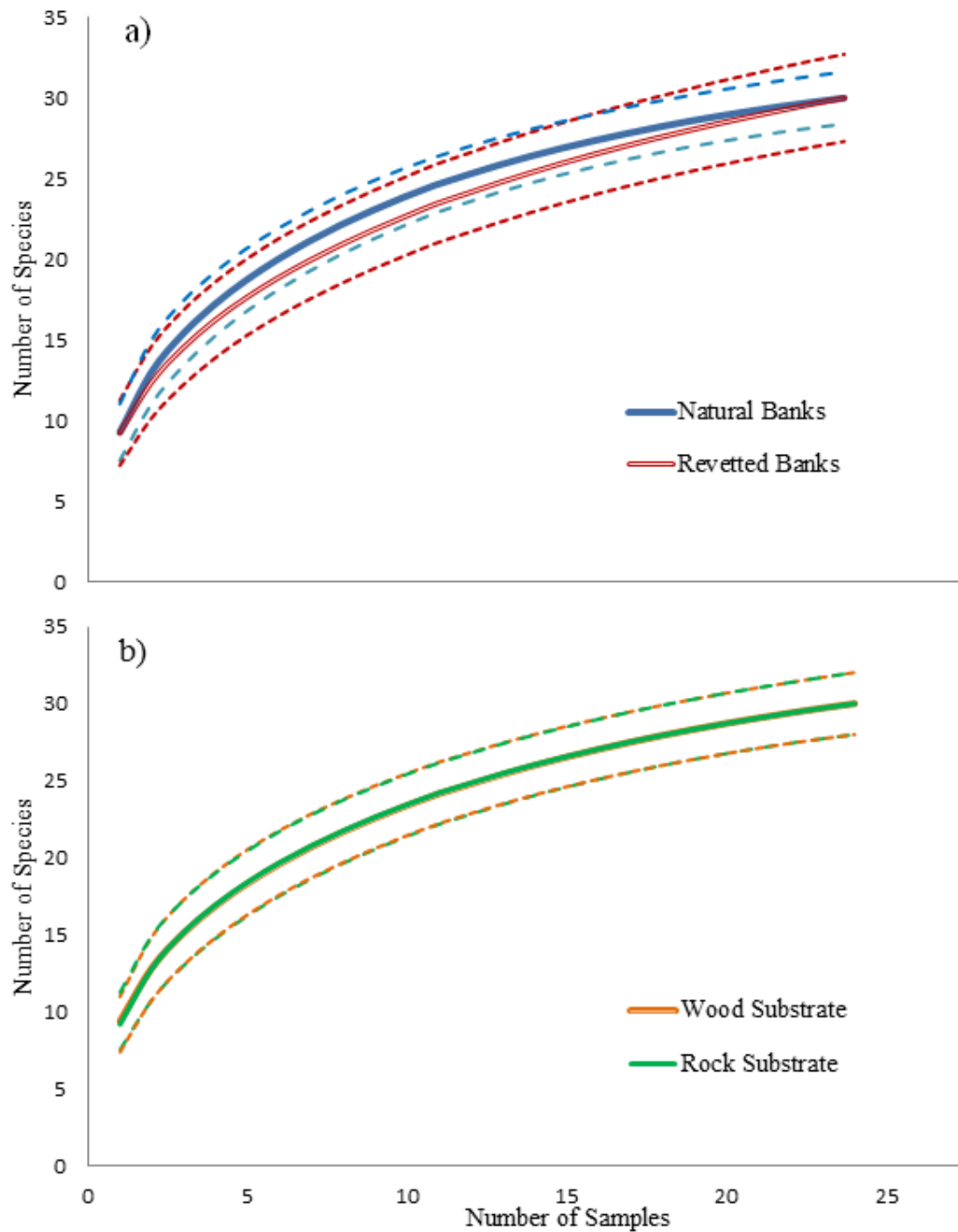


Figure 2.4: Rarefaction curves depicting macroinvertebrate family richness at a) revetted and natural banks and b) rock and wood substrates. Dashes represent 84% confidence intervals. Overlap between the confidence intervals implies that species richness does not differ between bank or substrate types ($P > 0.05$).

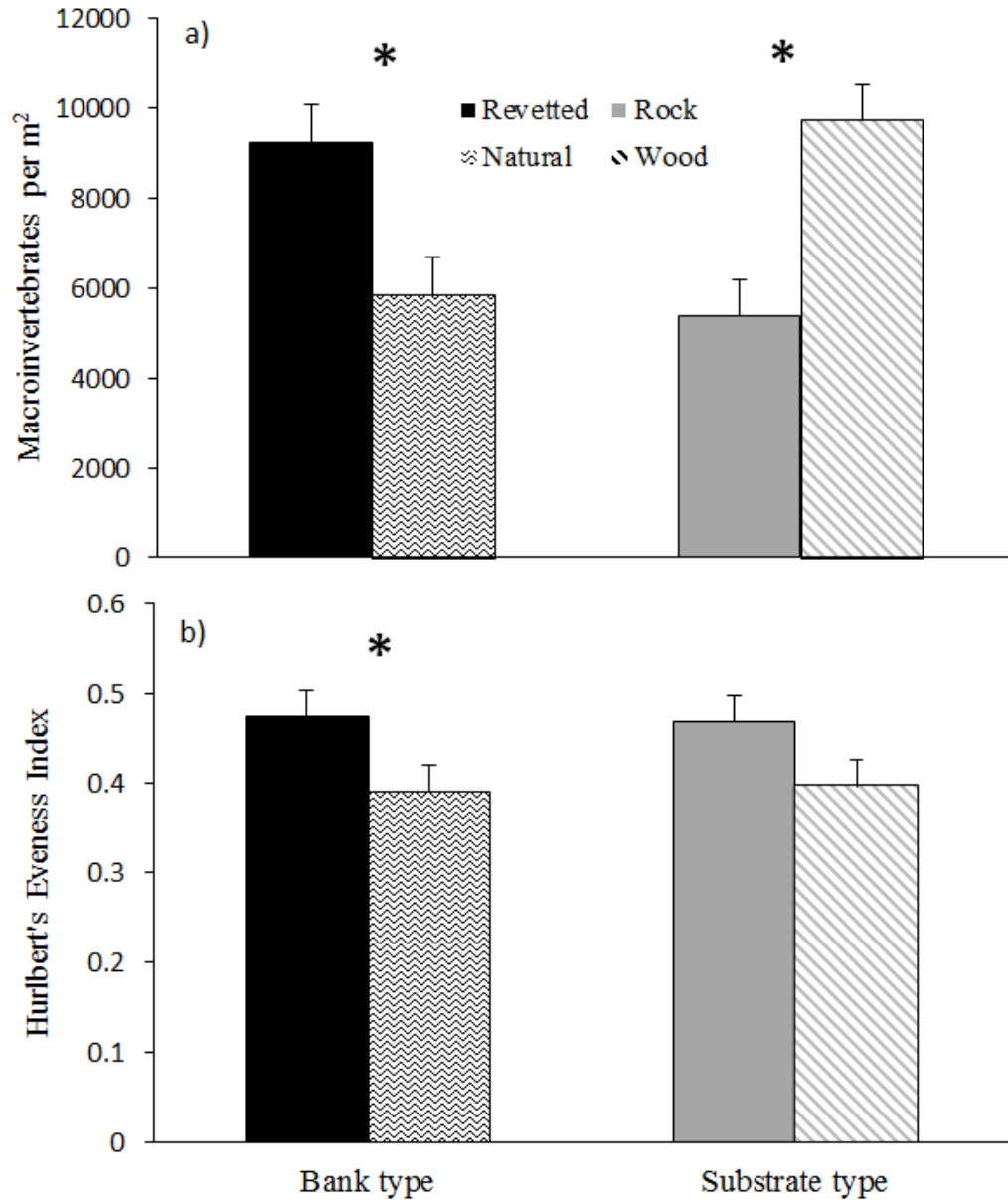


Figure 2.5: Macroinvertebrate a) abundance and b) evenness (mean + SE) at different bank and substrate types. Asterisks indicate significant differences between factor means ($P < 0.05$).

Polycetropodidae were all more common at revetted banks, while Ephemeridae and Leptoceridae were more common at natural banks (Table 2.7).

Table 2.6: Macroinvertebrate families indicated by non-parametric ANOVA's to be significantly more abundant on either rock or wood substrate. DF = 46 for all tests.

Family	P value	Substrate Preference	Density (per m ²)	
			Rock	Wood
Calopteryidae	0.039	Wood	0	4.9
Chironomidae	0.002	Wood	2830.1	5767.0
Coengarionidae	0.038	Rock	37.9	19.1
Ephemeridae	0.003	Rock	51.6	5.6
Heptageniidae	0.022	Wood	327.4	582.9

Table 2.7: Macroinvertebrate families indicated by non-parametric ANOVA's to be significantly more abundant at either natural or revetted banks. DF = 46 for all tests.

	P value	Bank Preference	Density (per m ²)	
			Natural	Revetted
Ephemeridae	< 0.001	Natural	56.3	1.0
Heptageniidae	0.026	Revetted	374.8	535.5
Hydrachnidia	< 0.001	Revetted	4.1	48.6
Hydropsychidae	< 0.001	Revetted	408.7	1309.0
Leptoceridae	0.002	Natural	70.8	32.9
Perlidae	0.006	Revetted	83.5	176.1
Pleuroceridae	0.039	Natural	3.5	0
Polycentropodidae	0.024	Revetted	21.4	49.0

2.3.3 Environmental Variables

None of the environmental variables that I measured during fish sampling differed between natural and revetted banks. In contrast, analyses indicated that velocity measurements taken at macroinvertebrate sampler locations differed between the bank types during deployment ($P = 0.002$, $F = 7.93$, $DF = 42$) and retrieval ($P < 0.001$, $F = 14.71$, $DF = 42$). CCA of macroinvertebrate assemblage data conducted with deployment and retrieval velocities as constraining variables indicated that the flow recorded at the time of deployment ($P = 0.015$, $F = 4.15$, $DF = 43$) and during retrieval ($P = 0.064$, $F = 1.68$, $DF = 43$) had significant or nearly significant effects on macroinvertebrate assemblages. Pearson rank correlation coefficients and

corresponding significance tests indicated that several macroinvertebrate families were positively correlated with increased velocities, including Hydropsychidae, Heptageniidae, and Perlidae (Table 2.8). The only family correlated with lower velocities was Ephemeridae.

Table 2.8: Significant Spearman rank correlations (and corresponding P values) between macroinvertebrate family densities and velocities at sampler locations during deployment and retrieval.

Family	Deployment Velocity		Retrieval Velocity	
Ephemeridae	0.201	(0.171)	0.325	(0.024)
Heptageniidae	0.431	(0.002)	0.335	(0.020)
Hydrachnidia	0.430	(0.002)	0.378	(0.008)
Hydropsychidae	0.551	(< 0.001)	0.563	(< 0.001)
Perlidae	0.539	(< 0.001)	0.428	(0.002)
Simuliidae	0.355	(0.013)	0.328	(0.023)

2.4 DISCUSSION

2.4.1 Pearl River Fishes

My findings regarding fish abundance, evenness, and assemblage composition imply that, during low river stages, revetted banks in the Pearl River represent a preferential habitat type for several fish species compared to similar natural banks. Specifically, several fish species were more abundant at revetted banks, particularly centrarchid sunfishes, during the summer and fall months. Although species common to revetted banks were also found at natural banks, natural bank assemblages contained much higher relative abundances of mobile cyprinid species such as blacktail and emerald shiners as well as detritivores including Mississippi silvery (*Hybognathus nuchalis*) minnows and gizzard shad.

The finding that fish assemblages vary between revetted and natural bank at low flows has been reported by several other studies (e.g., Lehtinen et al. 1997, Madejczyk et al. 1998), and likely reflects species responses to differences between revetted and natural banks in habitat structure and heterogeneity as well as prey availability during low river stages (Bolding et al.

2004). On-site observations indicated that when river stage was low, much of the complex structure at natural banks was exposed above the water line, limiting available habitat to the bottom portions of steep, eroding banks and sand and mud shoals. Survey data also indicated that flows were generally homogenous at these natural bank habitats, and that the only available complex structure or coarse substrate was large woody debris that remained submerged.

Although these conditions can be beneficial to detritivores and cyprinids (Ross 2000), and may account for their greater abundances at these sites, fine substrates can depress abundances of epibenthic fauna, which can lead to reductions in populations that feed on these organisms (Henley et al. 2000). Small amounts of woody debris, however, can support relatively large abundances of epibenthic species (Benke et al. 1985, Stewart et al. 2012), which may account for the presence of several invertivores at both natural and revetted banks, although at different densities.

Although woody debris was mostly absent at revetted banks compared with natural banks, these banks contained greater amounts of coarse substrates and complex structure during base flows, although this structure was in the form of interstitial spaces associated with concrete or boulder riprap. Woody debris generally supports the greatest abundance of macroinvertebrates in southeastern systems (Benke 1998, Benke and Wallace 2003, Testa et al. 2011), but my findings and those of others indicate that riprap is also capable of supporting large macroinvertebrates populations due to the biofilm that accumulates on the substrate, as well as the interstitial spaces and the heterogeneous flows associated with revetted banks (Way et al. 1995, Litvan et al. 2007). Increased abundances of biofilms and biofilm grazers associated with the large concentrations coarse substrates at revetted banks can correspond to increases in invertivorous fish (Dardeau et al. 1995). These increases, combined with greater amounts of

complex structure, may explain the greater abundances of several fish species, including sunfish and bullhead minnows, at revetted banks in the summer season (Baker et al. 1991). Large piscivorous species that were more common at revetted banks, such as spotted bass, may also be benefiting from the heterogeneous habitat, as well as the increased abundance of their prey.

Although fish assemblages differed between revetted and natural banks during the summer and fall sampling seasons when river stage was near base flows (mean $m^3/s = 84.1$), assemblages were similar during the winter season when water levels were elevated (mean $m^3/s = 291.7$). There are a limited number of revetment studies that have collected winter and spring flood fish assemblage data due to the difficulties associated with sampling during these periods. In one of the few studies examining armored bank fish assemblages during winter or spring high flows in a system possessing a similar hydroperiod to the Pearl River, Pennington et al. (1983) found that differences in species composition between revetted and natural banks continued through the flood stage in the Lower Mississippi River. It is likely, however, that these results reflect the highly degraded state of the habitat provided by the Mississippi River's natural banks (Pennington et al. 1983). In the Pearl River, where most banks are unmodified and not levied, it is possible that submerged large woody debris, root structures, and riparian vegetation result in proportionately greater amounts of fish habitat at natural banks during high river stages compared to base flows (Jackson et al. 2001). Greater amounts of complex habitat in surrounding reaches could decrease the relative value of the habitat provided by revetted banks at this time (Dardeau et al. 1995). Fish life history traits may also result in more homogenous fish assemblages between revetted and natural banks during high flow seasons. McCargo and Peterson (2010) suggested that during high flows in the late winter and early spring, many southeastern fish species are seeking mates and undergoing annual migrations. These fish thus

demonstrate reduced habitat fidelity at this time, except at breeding sites. Conversely, in the summer months, many southeastern fish species prioritize food and refuge availability, and are thus more likely to seek out habitats that maximize both, such as revetted banks (Dardeau et al. 1995, McCargo and Peterson 2010). It is also possible that my inability to detect differences in assemblage composition between bank types during the winter season may be the result of seasonal differences in sampling efficiency. I believe that this is unlikely, however, because several fish species that were well represented in samples from every season demonstrated abundance differences between revetted and natural banks during base flows, but did not demonstrate these differences during the winter.

The fact that fish assemblage structure differed between natural and revetted banks, yet species richness did not, is concurrent with some studies (Burres et al. 1982, Farabee 1986, Shields et al. 2000), but not with others (Jennings et al. 1999, Erős et al. 2008, White et al. 2009). There are several potential reasons for similarity in species richness between revetted and natural banks. Most notably, downstream reaches in medium and large sized rivers in the southeastern coastal plain often contain greater amounts of generalist fishes than upstream reaches, resulting in the homogenization of species assemblages between sites (Scott and Helfman 2001). Recent research also indicates that the effects of long term anthropogenic impacts following the river's impoundment, combined with recent pulse disturbances such as hurricane Katrina in 2005, have resulted in general declines in fish diversity throughout the river (Gunning and Suttkus 1991, Stewart et al. 2005, Geheber and Piller 2012). Particularly hard hit have been populations of rare habitat specialists (Piller et al. 2004, Tipton et al. 2004), which represent the species most likely to preferentially select one habitat over another, resulting in richness differences (Scott and Helfman 2001). Although I cannot predict whether larger

populations of habitat specialists would have influenced my richness data, generalists were sufficiently common at both revetted and natural banks to result in similar aggregated richness values despite differences in assemblage structure.

Given the low number of rare habitat specialists and other uncommon species detected during my study, I am unable to determine the impact that revetted banks have on rare fish populations in the Pearl River. Of note, however, is the fact that the highfin carpsucker, which is rare in other regional river systems, was more common at revetted banks in the summer season. Moreover, examination of captured individuals indicated that this species was using revetted banks as spawning habitat. Other rare species including several sturgeons have been known to preferentially spawn on riprap (Wesche 1985), although, research quantifying egg survival of any rare or common species on this substrate is lacking. I suggest that future research examine this topic in order to determine whether riprap is detrimental or beneficial to highfin carpsucker and rare species recruitment rates.

2.4.2 Pearl River Benthic Macroinvertebrates

Benthic macroinvertebrate data from introduced substrate samplers reflected the trends seen in fish species data collected during base flows. Although family richness did not differ between bank types, evenness and abundance were generally highest at revetted banks, and family assemblages were significantly different between the two bank types. My results largely concur with the few studies that examine the influence revetted banks on benthic macroinvertebrate assemblages, although several of these studies also note increased richness at revetted banks (Burress et al. 1982, Duehr et al. 2006, Litvan et al. 2007, Nelson 2011). In most cases, however, these studies either take place in systems with fewer generalist species, or they identify specimens to a lower taxonomic level than family (Litvan et al. 2007, Nelson 2011). It

is possible that variations in richness between bank types may be detectable given a greater degree of taxonomic resolution. However, in a study that included several southern tributaries of the Pearl River, Markos (2010) found that high degrees of taxonomic overlap between reaches extended to the genus level.

Differences in macroinvertebrate assemblage structure, evenness, and abundance between samplers retrieved from revetted and natural banks likely reflected differences in habitat features between the two bank types, particularly differences in near-bank currents and siltation rates. Many of the families that were significantly more abundant at revetted banks are associated with high flow areas, including Hydropsychidae, Heptageniidae, and Perlidae (Ebrahimnezhad and Harper 1997, Thorp and Covich 2009). Sedimentation resulting from bank erosion at several of the sampled natural banks also likely affected the structure of the resident assemblages, e.g., burrowing mayflies of the family Ephemeridae were almost exclusively found in samplers containing sediment that originated from visible bank erosion. High sedimentation rates are often negatively correlated with macroinvertebrate diversity, and likely influenced assemblage structure in the Pearl River by depressing the abundance of some species at natural banks (Henley et al. 2000, Downes et al 2006).

Samplers containing wood substrates had greater abundances of macroinvertebrates than those containing concrete substrate. It is possible, however, that this result reflects the fact that, despite the use of an established deployment time, biofilm development on rock substrates during deployment was minimal. Although assemblages on artificial substrates likely reflected the assemblages on the benthos they were deployed on (Rosenberg and Resh 1982, Waters et al. 2005), it is possible that the relative abundances of some species on introduced riprap varied from those on extant riprap possessing a mature biofilm. However, macroinvertebrates should

still have been attracted to interstitial refugia present in introduced riprap samplers (Lancaster and Hildrew 1993), and comparisons can still be made between the two banks based on the fact that the same sampling gear was used at both treatment levels. I caution that comparisons made between substrate types may reflect macroinvertebrate assemblage differences between woody debris and newly created revetted bank habitat, rather than older, established banks. However, my results do indicate that, at newly created revetted banks, woody debris may increase macroinvertebrate populations, which should benefit other trophic levels as well. These results are consistent with lotic and lentic restoration research that indicates that the inclusion of additional complex woody structure when armoring banks is beneficial to local aquatic fauna diversity and abundance (Henderson 1986, Fischenich 2003, Miller et al. 2010).

2.5 CONCLUSION

My results suggest that during the winter and early spring, revetted banks in the Pearl River support biotic assemblages that are comparable to those at natural banks. During base flows, however, my data indicates that revetted banks support unique fish assemblages. Although I cannot attest to the impact that riprap has on rare or uncommon species, my data does suggest that many common macroinvertebrate families, several moderately common fish species, and most native centrarchids demonstrate a detectable preference for revetted banks. This implies that the heterogeneous habitat provided by riprap during low river stages may be beneficial to diversity in the Pearl River, although potentially only among already abundant taxa. It should be emphasized that many studies in several lotic systems have indicated that an overabundance of revetted banks can have serious and detrimental hydrologic and ecological impacts on a local and even system-wide scale (Schmetterling et al. 2001). The percentage of banks armored with riprap in the Pearl River (<1.5%) is comparable to other systems in which

riprap was noted to have a beneficial effect on diversity (Gidley et al. 2012). It is possible that adding significantly more riprap to the system, or adding it to areas where it is already common, may be detrimental to aquatic communities. I also stress that my study was limited to existing revetted banks and distant, unimpacted banks, and thus, increases in diversity indicated by my results are on a local scale and may not reflect net regional increases. I strongly suggest that future research in the Pearl River and other southeastern systems examine the effects that riprap has on region-wide diversity, particularly in systems with larger proportions of revetted banks. I also stress that my results may not be applicable to otherwise similar systems where revetted banks compose a significantly greater relative proportion of the total bank area.

2.6 WORKS CITED

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CHAPTER 3: RECOVERY OF FISH ASSEMBLAGES IN A SOUTHEASTERN COASTAL PLAIN RIVER FOLLOWING AN ANTHROPOGENIC PULSE DISTURBANCE

3.1 INTRODUCTION

Short term disturbances, known as pulse disturbances, can have significant impacts on the structure of lotic fish assemblages (Bender et al. 1984, Lake 2000, Kubach et al. 2011).

Consequently, a substantial amount of research has attempted to examine the effects of pulse disturbances on these assemblages, and quantify their rates of post-disturbance recovery. In most post-pulse disturbance studies, fish assemblages have demonstrated a high degree of resilience (Gresswell 1999, Roghair and Dolloff 2005, Kroon and Ludwig 2010). In a review of disturbance case studies, Niemi et al. (1990) indicated that most studies monitoring fish communities after pulse disturbances found fish diversity to return to pre-disturbance levels in less than two years. However, there is a high degree of variability in fish recolonization and recovery rates among pulse disturbance studies, with some assemblages recovering in a period of weeks (Yount and Niemi 1990). Variability in fish recolonization and recovery rates reflect the influence of several factors, including the degree of disturbance (Lake 2000, McHugh et al 2010); the proximity and availability of refugia and source populations (Magoulick and Kobza 2003, Adams and Warren 2005); the environmental characteristics of the disturbed system (Lake 2000, Ferreira et al 2007, Davey and Kelley 2007); and life history traits of the disturbed species (Matthews and Marsh-Matthews 2003, Leathwick et al. 2008, Albanese et al. 2009).

Given the large degree of variation in recolonization and recovery rates between different systems, it is notable that few studies have examined the recovery of fish assemblages following pulse disturbances in higher order systems (Detenbeck et al. 1992), with most studies focusing on second to fourth order streams (Ensign et al. 1997). Additionally, comparatively little

research has been conducted in southeastern coastal plain systems (Adams and Warren 2005). Higher order river systems, especially those in the southeastern coastal plain, typically support diverse and often unique fish assemblages (Johnson et al. 1995, Galat and Zweimüller 2001, Scott and Helfman 2001). Given the already highly degraded state of many of these rivers, recovery of their biota from pulse disturbances warrants further investigation.

On August 9, 2011, the Temple-Inland Paper Mill in Bogalusa, Louisiana discharged approximately 80 - 100 million gallons of concentrated kraft mill effluent known as “black liquor” into the Pearl River. Black liquor is a mixture of lignins and sulphuric bases, which, in addition to being toxic to most aquatic biota, can also cause hypoxia due to increased biochemical oxygen demand and substantially elevated ambient pH levels (Kelso et al. 1977, Chapman et al. 1982, Kleynhans et al. 1992, Brown and Daniel 2012). The spill occurred when the Pearl River was at base flows, and as a result, the black liquor constituted 7.0 - 8.8% of the river’s daily discharge on August 9th. The introduction of large quantities of effluent into the Pearl River resulted in complete fish extirpation throughout the combined 235 km of the river’s mainstem and its tributaries downstream of the spill’s source (Bart 2012). Following the spill, the mill effluent was flushed downstream and out of the river, and local state agencies reported that system-wide water quality parameters returned to standard seasonal levels by August 18, 2011.

The August 2011 mill effluent spill in the Pearl River provided the opportunity to study recolonization by aquatic biota in an understudied southeastern coastal plain river system following a catastrophic pulse disturbance. The goal of my study was to assess the recovery of fish assemblages and diversity in the Pearl River following the 2011 paper mill effluent spill and corresponding fish kill.

3.2 METHODS

3.2.1 Sample Site Selection

Sampling was limited to the lower Pearl River as defined by the Mississippi Department of Environmental Quality, which included all areas below Columbia, MS (MDEQ 2000).

Sample sites included in this study were within one of four regions (Figure 3.1). Two of these regions, Columbia (COL) and North Bogalusa (NBOG), were located upstream of the source of the mill effluent and were unimpacted by the fish kill. The other two regions, South Bogalusa (SBOG) and Picayune (PIC), were downstream and thus subject to the mill effluent's effects. Each region contained six electrofishing transects of varying lengths that included three pairs of armored and natural banks located in developed and undeveloped portions of the river respectively. All regions and sample sites were selected based on their proximity (within 15 km) to one the few public access points in the river.

3.2.2 Fish Sampling

I collected fishes via electrofishing during three time periods to assess the effects of the effluent spill. The first period was between September 23 and October 13, 2011, approximately two months after the fish kill. The second period, January 17 to March 8, 2012, was five to seven months after the disturbance. The final collecting period was 9-10 months after the disturbance, between May 22 and June 11, 2012. When possible, all sites were sampled during each period, although abnormally low river stages during the summer of 2011 prevented sampling at sites in Columbia during the first sampling period.

A crew of two netters and one boat driver conducted boat-mounted, pulsed DC electrofishing at 60Hz along the full length of each electrofishing transect, and recorded the total sampling seconds after each collection event. Voltage and amperage were adjusted as needed

based on river specific conductance on the day of sampling, usually maintaining a current of approximately five to seven amps. We recorded the total length of all captured fish and identified them to the species level in the field when possible. Specimens that could not be accurately identified in the field were placed in an ice slurry and transferred to the laboratory for identification (LSU AgCenter IACUC A2011-16).

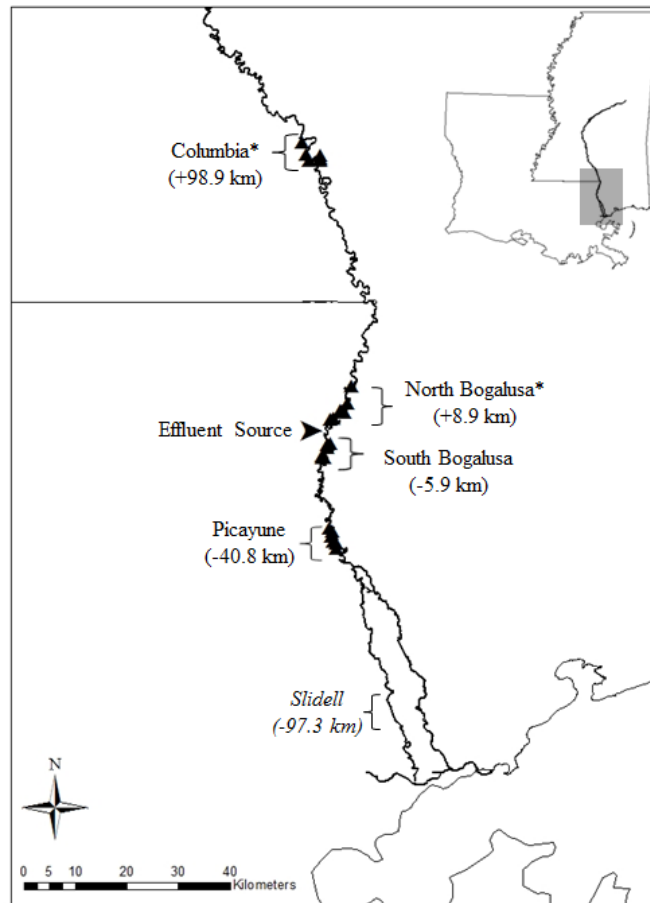


Figure 3.1: Map of sample regions and sites on the Pearl River. The mill effluent source is noted with an arrow. Triangles indicate sample sites. Regions unaffected by mill effluent are followed by an asterisk. Values in parentheses indicate average river kilometers upstream (+) or downstream (-) from the effluent source. The italicized region was not included in analysis due to the effects of salinity.

3.2.3 Statistical Analysis

Abundance, richness, and evenness between disturbed and undisturbed regions were compared with Tukey-Kramer contrasts conducted on seasonal data with general linear mixed

models (GLMM; PROC MIXED; SAS version 9.3, SAS Institute Inc., Cary, North Carolina). Fish abundance was calculated as catch per unit effort (CPUE; number of fish per second) from electrofishing samples. To compensate for differences in transect length between sample sites, richness was calculated as rarefied richness and evenness was calculated with Hurlbert's evenness index, since both of these metrics are robust to differences in sample size (Beisel et al. 2003, Magurran 2004, Olszewski 2004). All model assumptions were assessed during analysis.

Within season comparisons of fish assemblage composition between undisturbed and disturbed sites were made with permutational multivariate analysis of variance (PERMANOVA; R version 2.15, Vegan Package, R Foundation for Statistical Computing) and corresponding within season-among site contrasts. All analyses were based on 9999 permutations and were conducted with Bray-Curtis dissimilarity matrices that were calculated with CPUE data. In seasons where undisturbed sites in both North Bogalusa and Columbia were sampled, data from these regions was pooled if contrasts did not indicate significant differences between their assemblages. In order to visualize assemblage differences found by PERMANOVA, I performed unconstrained ordinations on seasonal Bray-Curtis distances with non-metric multidimensional scaling (NMDS, R version 2.15, Vegan Package). All NMDS plots were calculated with a maximum of 999 random starts, and were constrained to two axes when stress was less than 20%. I used principal component rotation so that the first axis accounted for the greatest amount of variation. I also plotted fish species scores in NMDS space based on weighted averages of site scores.

Whenever PERMANOVA tests indicated significant differences in assemblage composition between disturbed and undisturbed regions, I performed non-parametric ANOVA's (PROC NPAR1WAY; SAS version 9.3) on individual fish species abundances using regional

CPUE data in order to determine how composition varied. I restricted tests to non-rare species represented by at least 10 individuals in a given seasonal sample in order to minimize type I error. These follow-up tests were conducted at $\alpha = 0.05$ with the Kruskal-Wallis test statistic. Follow up tests were conducted at $\alpha = 0.05$ based on the principle of protected tests in MANOVA (Stevens 2002), which is conceptually similar to PERMANOVA.

Non-parametric ANOVA's implementing the Kruskal-Wallis test statistic were also used to compare total lengths of commonly captured fish species (> 50 individuals) between undisturbed and disturbed regions. I limited these tests to data from the final sampling season due to samples from prior seasons either being too small or dissimilar to adequately compare species lengths.

3.3 RESULTS

During the course of the study, 2,790 fish representing 37 species were collected from undisturbed sites, while 2,944 fish representing 39 species were collected from disturbed sites. Centrarchid species and blacktail shiners (*Cyprinella venusta*) were numerically dominant in both undisturbed and disturbed regions during every season, except at disturbed sites in Picayune during the first sampling season, where juvenile threadfin shad (*Dorosoma petenense*; average length = 46.9 mm) dominated the assemblage (Table 3.1). After the first sampling season, though, threadfin shad were rarely found at any site.

Tukey-Kramer contrasts indicated that, during the first sampling period, mean abundance, evenness, and richness at disturbed sites in Picayune, which were 37.1 to 44.6 km downstream of the spill source, were significantly different from undisturbed sites upstream of the spill in North Bogalusa (Figure 3.2). Both evenness and richness were lower in Picayune relative to North Bogalusa, whereas abundance was much higher due to the large schools of

juvenile threadfin shad that were found at every site sampled in the region. During the same sampling period, disturbed sites in South Bogalusa, which were 3.2 to 9.2 km downstream of the spill source, did not differ significantly from undisturbed sites in North Bogalusa in abundance, evenness, or richness. During later sampling seasons, disturbed sites in both Picayune and South Bogalusa did not differ significantly in mean abundance, evenness, or richness from undisturbed sites upstream of the spill (Figure 3.2).

Table 3.1: Numerical abundances of fish captured in each region during each season. PIC = Picayune (disturbed). SBOG = South Bogalusa (disturbed). NBOG = North Bogalusa (undisturbed). COL = Columbia (undisturbed).

	Fall			Winter				Summer				Total
	PIC	SBOG	NBOG	PIC	SBOG	NBOG	COL	PIC	SBOG	NBOG	COL	
Blacktail shiner	56	30	257	50	31	112	83	442	150	348	401	1960
Longear sunfish	1	73	96	20	34	75	91	326	100	157	398	1371
Threadfin shad	936	0	0	0	0	1	0	0	0	0	1	938
Bluegill	1	58	34	13	23	12	8	103	69	59	68	448
Bullhead minnow	0	20	6	3	8	7	6	71	10	20	55	206
Channel catfish	2	4	7	2	6	23	23	22	6	4	26	125
Spotted bass	0	9	26	1	5	8	13	16	8	10	18	114
Mississippi silvery minnow	0	11	2	0	0	3	42	18	6	26	0	108
Emerald shiner	0	3	20	7	5	4	10	17	1	4	1	72
Largemouth bass	0	0	4	0	0	0	1	5	6	4	6	26
Other species	5	14	30	26	14	32	59	71	26	32	57	366

Comparisons of fish assemblages with PERMANOVA contrasts indicated that, despite similarities in abundance, evenness, and richness between undisturbed sites in North Bogalusa and disturbed sites in South Bogalusa during the first sampling period, fish assemblages were significantly different between all three sampled regions at this time (Table 3.2). However, PERMANOVA contrasts did imply that, by the second sampling period, assemblages at disturbed sites in South Bogalusa immediately below the spill were no longer different from those at undisturbed sites (Table 3.2), although test statistics implied that assemblages at undisturbed sites in Columbia and North Bogalusa remained more similar to each other than to

assemblages in South Bogalusa. Fish assemblages at further downstream, disturbed sites in Picayune continued to remain different from those at undisturbed sites for the duration of the study. Decreasing contrast test statistics, however, imply that assemblage dissimilarity between Picayune and undisturbed sites decreased with each successive period (Table 3.2).

Table 3.2: Results of PERMANOVA contrasts testing differences between regional fish assemblages within each sampling season. PIC = Picayune (disturbed), SBOG = South Bogalusa (disturbed). NBOG = North Bogalusa (undisturbed). Undisturbed = Columbia and North Bogalusa.

Test	DF	MS	F	P value
Sep- Oct. 2011				
NBOG v SBOG	1, 8	0.37	4.02	0.024
NBOG v PIC	1, 8	1.57	16.85	< 0.001
SBOG v PIC	1, 8	1.44	15.45	< 0.001
Jan. - Mar. 2012				
NBOG v COL	1, 12	0.03	0.26	0.989
SBOG v PIC	1, 12	0.16	1.08	0.364
Undisturbed v SBOG	1, 12	0.20	1.36	0.186
Undisturbed v PIC	1, 12	0.41	2.76	0.009
May 22 - Jun. 2012				
NBOG v COL	1, 16	0.09	0.73	0.575
SBOG v PIC	1, 16	0.13	1.07	0.359
Undisturbed v SBOG	1, 16	0.20	1.78	0.121
Undisturbed v PIC	1, 16	0.31	2.79	0.032

Spatiotemporal NMDS ordination patterns supported the PERMANOVA results. NMDS confirmed that separation between undisturbed and disturbed regional fish assemblage centroids, which corresponds to fish assemblage dissimilarity between these regions, was greatest during the first sampling period and declined with successive periods (Figure 3.3 - 3.5). The NMDS ordination of data from the first period also indicated that variation between disturbed assemblages in Picayune and undisturbed assemblages in North Bogalusa was most apparent on the first axis, whereas variation between disturbed assemblages in South Bogalusa and undisturbed assemblages in North Bogalusa was restricted to the second axis (Figure 3.3). Because principal component axis rotation causes the first axis to represent the majority of the

variation in the model, this ordination implies that undisturbed fish assemblages in North Bogalusa were more similar to assemblages in South Bogalusa than to assemblages further downstream in Picayune during the first sampling period. In the NMDS ordination of data from the second sampling period, separation between regional centroids was less evident (Figure 3.4). The centroid representing disturbed assemblages in South Bogalusa was within the standard error boundaries of the centroid representing undisturbed assemblages in Columbia and North Bogalusa, implying minimal dissimilarity in species assemblages between these regions. Simultaneously, separation between disturbed assemblages in Picayune and undisturbed assemblages was less prominent than in the previous period. In the ordination of data from the final sampling period, South Bogalusa's centroid continued to remain within the standard error boundary of the undisturbed centroid (Figure 3.5). Separation between assemblages in Picayune and undisturbed regions was still visible, but confined to the second axis, indicating that variation between Picayune and undisturbed regions was not a primary source of variation in the model during this last sampling period.

Analysis of differences in individual species abundances between undisturbed and disturbed regions with nonparametric ANOVA's indicated that several fish species were significantly more abundant at undisturbed sites in North Bogalusa compared to disturbed sites in Picayune during the first sampling period (Table 3.3), with only threadfin shad being more abundant in Picayune. Assemblage differences during the same period between South Bogalusa and North Bogalusa were less pronounced but still present. ANOVA tests indicated that significant differences in species abundances between these two regions were limited to cyprinid species (Table 3.3), although NMDS indicated that several other species, including largemouth

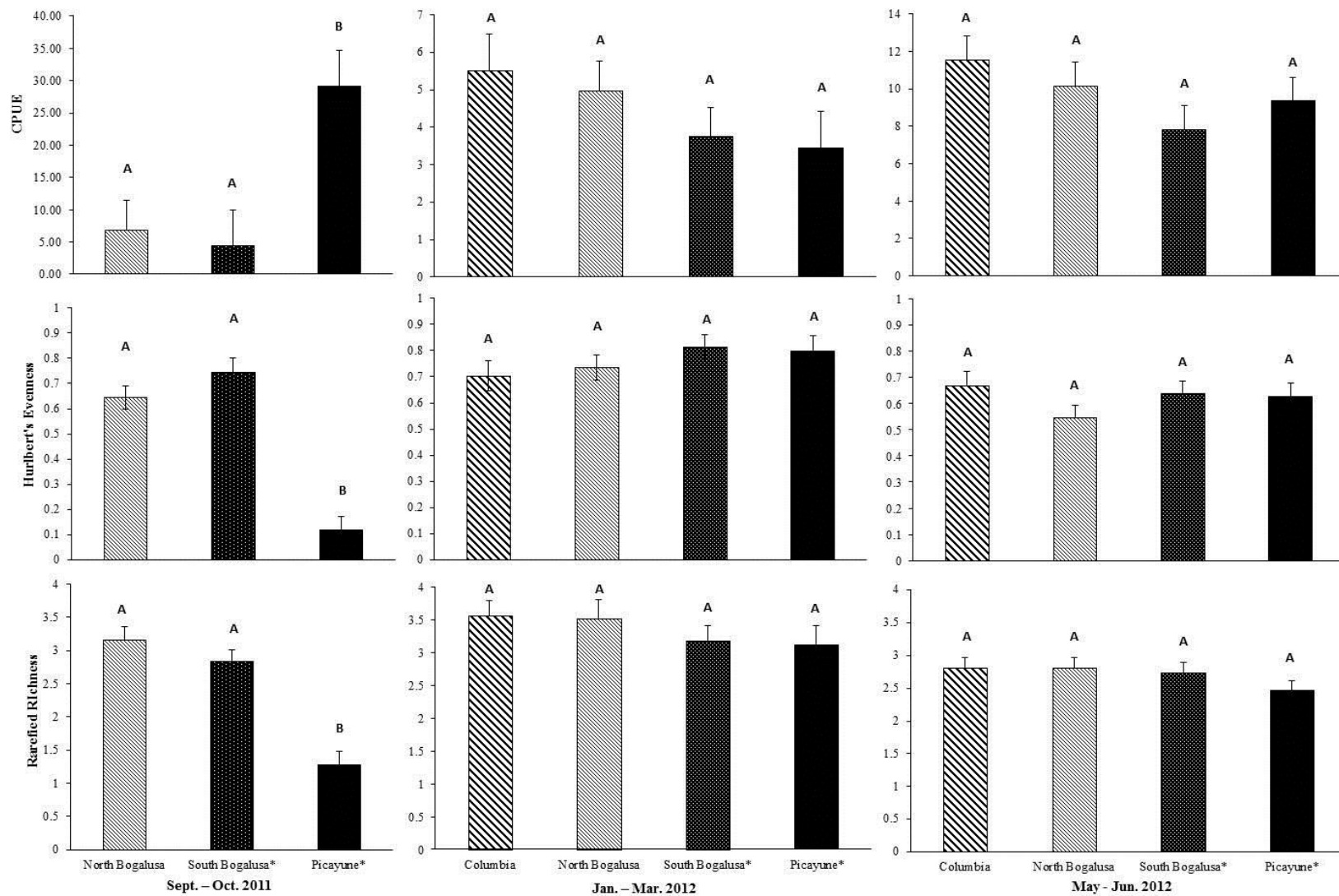


Figure 3.2: Fish abundance (top row, CPUE), evenness (middle row, Hurlbert's Evenness index), and rarefied richness (bottom row) in each region during each sampling season. Letters indicate similar means ($P > 0.05$, Tukey-Kramer). * indicates disturbed regions.

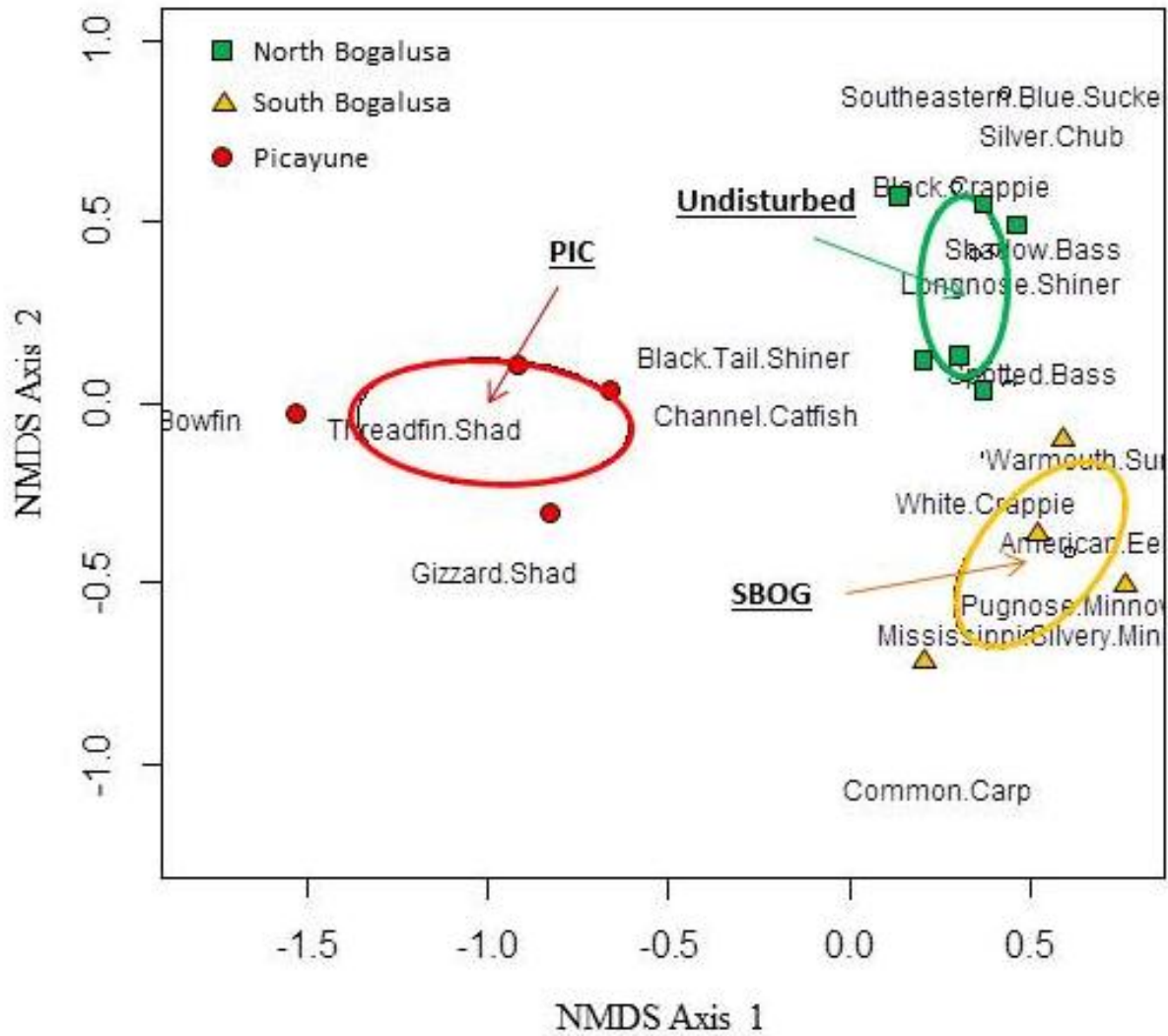


Figure 3.3: Non-metric multidimensional scaling ordination of regional fish assemblages ~2 months after the spill. Points represent sampling sites. Arrows point to centroids of corresponding underlined regions (Undisturbed = North Bogalusa, PIC = Picayune, SBOG = South Bogalusa). Ellipses represent standard error of regional centroids. Text represents species scores.

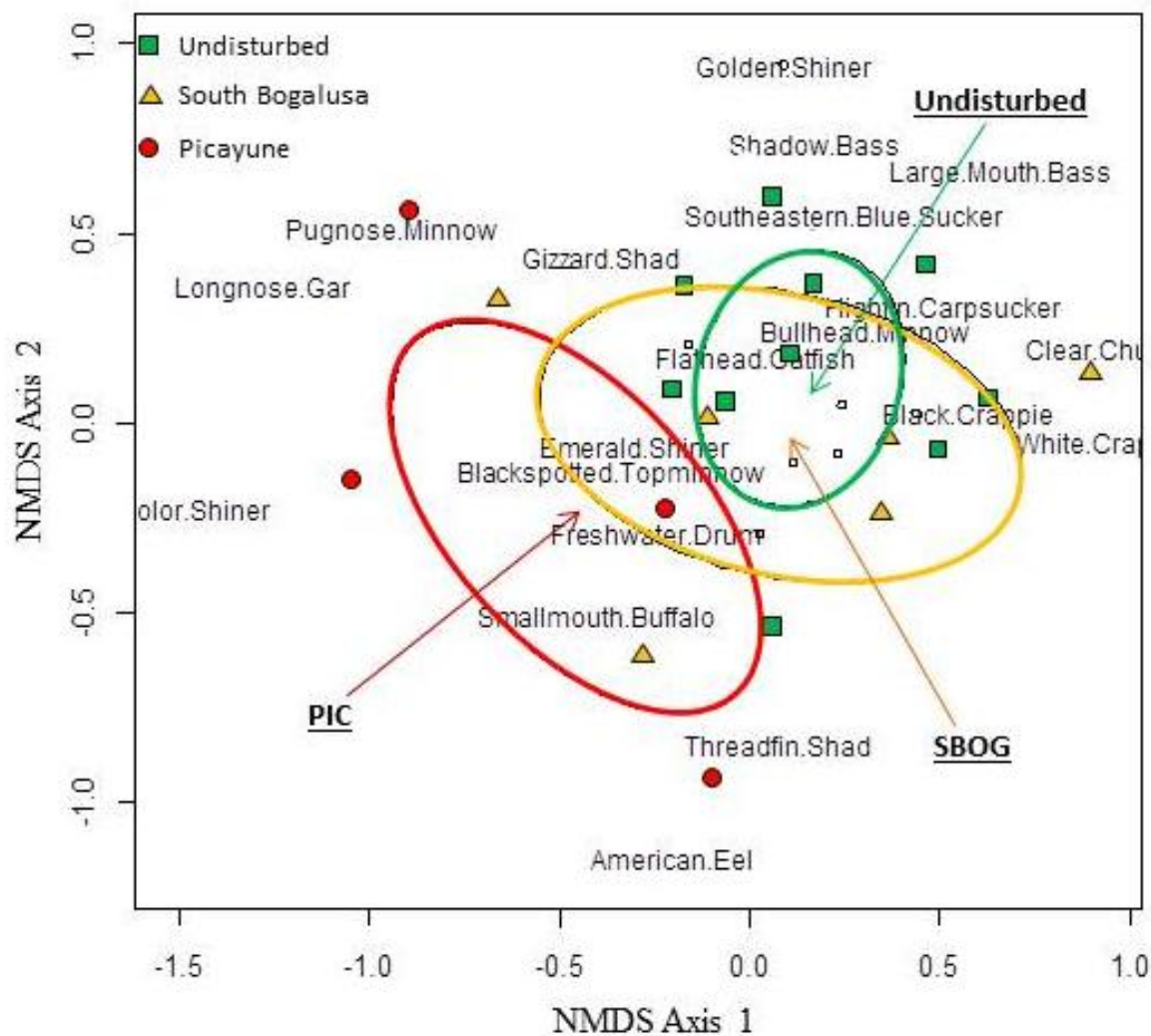


Figure 3.4: Non-metric multidimensional scaling ordination of regional fish assemblages ~6 months after the spill. Points represent sampling sites. Arrows point to centroids of corresponding underlined regions (Undisturbed = North Bogalusa and Columbia, PIC = Picayune, SBOG = South Bogalusa). Ellipses represent standard error of regional centroids. Text represents species scores.

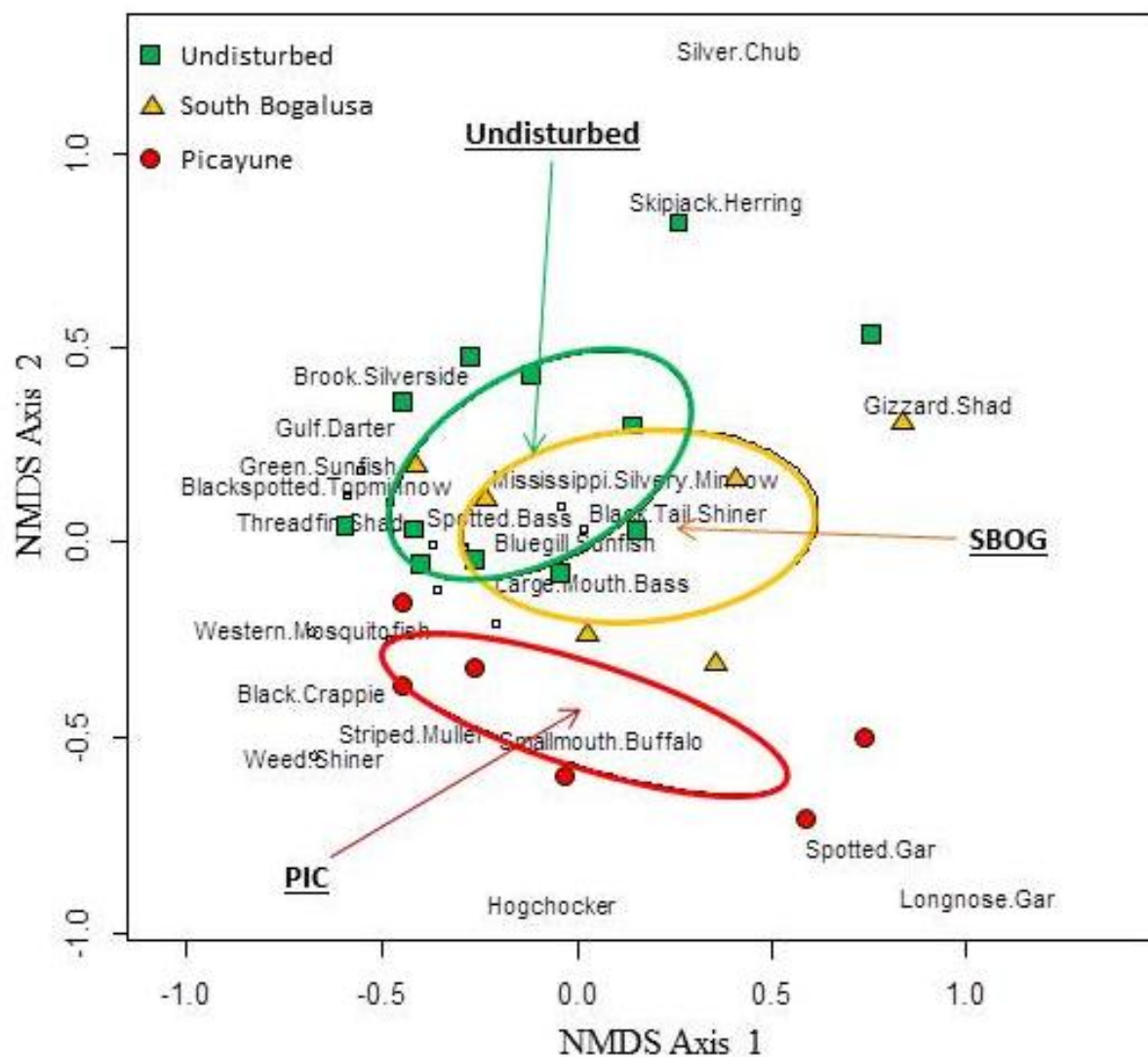


Figure 3.5: Non-metric multidimensional scaling ordination of regional fish assemblages ~10 months after the spill. Points represent sampling sites. Arrows point to centroids of corresponding underlined regions (Undisturbed = North Bogalusa and Columbia, PIC = Picayune, SBOG = South Bogalusa). Ellipses represent standard error of regional centroids. Text represents species scores.

bass (*Micropterus salmoides*) and flathead catfish (*Pylodictis olivaris*), exhibited a non-significant trend of greater abundance in North Bogalusa than South Bogalusa (Figure 3.3).

Table 3.3: Significant ($P < 0.05$) and nearly significant ($P < 0.10$) results of non-parametric ANOVA's comparing fish species abundances between disturbed and undisturbed regions. x's indicate P values > 0.10 . Dashes indicate that no test was performed. PIC = Picayune (disturbed). SBOG = South Bogalusa (disturbed).

Species	Contrast		Abundance		
	PIC v	SBOG v			
	Undisturbed	Undisturbed	PIC	SBOG	Undisturbed
Sep. – Oct. 2011 ^A					
Bluegill	0.010	x	1	58	34
Blacktail shiner	x	0.033	56	30	257
Emerald shiner	0.023	x	0	3	20
Longear Sunfish	0.010	x	1	73	96
Longnose shiner	0.023	0.037	0	1	8
Spotted Bass	0.023	x	0	9	26
Threadfin shad	0.004	N/A	936	0	0
Jan. – Mar. 2012 ^B					
Channel catfish	0.011	-	2	-	36
Longear sunfish	0.066	-	20	-	152
Spotted bass	0.054	-	1	-	16
May – Jun. 2012 ^B					
Spotted gar	0.010	-	5	-	0

^A Undisturbed sites consist only of North Bogalusa

^B Undisturbed sites consist of North Bogalusa and Columbia

Due to a lack of assemblage dissimilarity between South Bogalusa and undisturbed regions during the next two sampling periods, subsequent non-parametric ANOVA tests were used only to analyze species abundance differences between undisturbed regions and Picayune. These ANOVA's were only able to detect significant or nearly significant abundance differences between Picayune and undisturbed regions for three species during the second sampling period, and only one during the final period (Table 3.3). When these results are viewed together with the species scores plotted on the NMDS ordinations, it can be inferred that although species

abundances varied between disturbed and undisturbed regions, large-scale differences in non-rare species were largely limited to the first sampling season.

Six species were sufficiently common during the final sampling season to allow comparisons between their lengths at disturbed and undisturbed sites during the final sampling period with non-parametric ANOVA's (Figure 3.6). These tests indicated that blacktail shiners, longear sunfish (*Lepomis megalotis*), and spotted bass (*Micropterus punctulatus*) were all significantly smaller in South Bogalusa than in undisturbed regions by the final sampling period. Further downstream in Picayune, all of the tested species except for channel catfish (*Ictalurus punctatus*) were significantly smaller than in undisturbed regions.

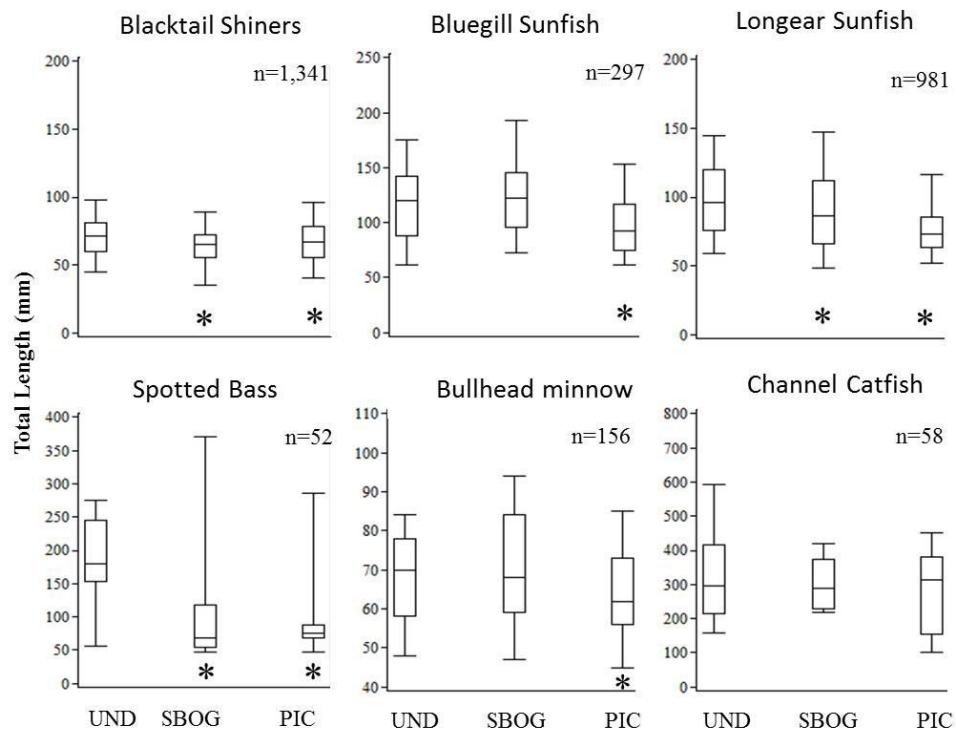


Figure 3.6: Regional total lengths of common fish species (> 50 specimens) during the final sampling season. Box-whisker plots represent percentile data (bottom whisker = 5th percentile, bottom of box = 25th percentile, middle of box = median, top of box = 75th percentile, top whisker = 95th percentile). UND = undisturbed sites in Columbia and North Bogalusa. SBOG = South Bogalusa. PIC = Picayune. Asterisks indicate significant differences from unaffected regions (Kruskal-Wallis test, $P < 0.05$).

3.4 DISCUSSION

Fish evenness, richness, and abundance indicated that the rate of recovery of fish populations in the Pearl River following the August 2011 fish kill was consistent with pulse disturbance recolonization in other North American river systems (Niemi et al. 1990, Detenbeck et al. 1992). Further, given the size of the Pearl River, the magnitude of recolonization that I documented represented a relatively fast recovery. Although recovery times of only a few weeks have been documented by some post-disturbance studies, these studies typically took place in systems that were smaller, less speciose, or more prone to annual disturbances such as drought, than the Pearl River system (Peterson and Bayley 1993, Sheldon and Meffe 1995, Albanese et al. 2009, Kubach et al. 2011). Studies documenting recoveries in medium and larger rivers often note that richness and evenness take several months to several years to return to undisturbed levels (Binns 1967, Detenbeck et al. 1992).

The rate of recolonization at disturbed sites is also notable given the timing of the mill effluent spill. Lotic recolonization rates in North America are thought to be greatest during the spring spawning season (Niemi et al. 1990, Meade 2004). Detenbeck et al. (1992) indicated that population recovery in large systems can be delayed by as much as a year when pulse disturbances happen after this spawning season. My data, however, indicates that despite the spill occurring in August, significant differences between richness and evenness in undisturbed and disturbed regions were undetectable by the winter sampling season. This implies that either fall spawning-related movements or other dispersal mechanisms were driving initial recolonization rates.

In addition to occurring relatively quickly, my data indicates that the recovery in the Pearl River was largely latitudinal. Latitudinal recovery is suggested by the faster recolonization of

my northernmost, upstream sample sites relative to my southernmost, downstream sites. Gradual recovery in an upstream to downstream direction has also been documented in several larger systems. Kubach et al. (2011) and Binns (1967) noted an upstream to downstream pattern of recovery following pulse disturbances in the Reedy River, South Carolina and the Green River, Wyoming respectively. The majority of post-pulse disturbance studies, however, have documented uniform rather than directional recoveries, although most of these studies examined small, experimentally defaunated reaches and lower order stream systems (Meffe and Sheldon 1990, Sheldon and Meffe 1995, Lonzarich et al. 1998). Uniform recoveries in these small systems likely reflect the high speeds at which species can disperse across short distances (Peterson and Bayley 1993, Jackson et al. 2001).

The faster rate of recovery of disturbed reaches near undisturbed portions of the river's mainstem relative to the recovery rates of downstream reaches implies that undisturbed portions of the mainstem served as the largest source of fish colonizers following the initial disturbance. This is supported by evidence that recolonization rates are typically greatest near large source populations (Dias 1996, Ensign et al. 1997). My results also indicate that numerous low-order tributaries that flow into the mainstem of the Pearl River may have served as additional sources of colonists. This is demonstrated by the presence of low-vagility species such as longear sunfish and spotted gar (*Lepisosteus oculatus*) ~40 km downstream of the spill in Picayune shortly after the fish kill. Regional inventories by Ward (2009) indicated that these species, and several of the cyprinids found in Picayune during the first two seasons, are common in low order tributaries of the Pearl River. The slower recovery of downstream reaches despite these populations, however, indicates the reduced role of these source populations relative to those in the mainstem. Other studies have noted greater amounts of tributary-based colonization in other

systems (Olmsted and Cloutman 1974, Kubach et al. 2011). The apparent lack of tributary population influence may be due to drought conditions in effect when the fish kill occurred. During carcass surveys completed during the spill, it was noted that several local tributaries were dewatered, and others had minimal flows (unpublished personal data). Studies examining the influence of drought on lotic systems indicate that the refuge value of tributaries and their ability to support representative source populations can decline under drought conditions (Magoulick and Kobza 2003).

Tidally influenced portions of the Pearl River near its confluence with Lake Borgne demonstrated a similar degree of recovery to sites included in this study by the summer of 2012. This was indicated by surveys that were conducted as part of a separate study (unpublished personal data) at sites approximately 97 km downstream from the source of the spill (Figure 3.1). This region, however, is characterized by salt tolerant assemblages that would not be reliant upon upstream colonizers during the recovery process, indicating that the recovery in this region was independent of upstream reaches. Anecdotal accounts from fish kill response teams indicate that the spatial arrangement of the interconnected channels that compose the extreme downstream regions of the Pearl River may have resulted in refugia for salt tolerant species, creating downstream population sources in this area. These findings suggest that recovery of sites near the river's mouth may have occurred at a different pace from upstream portions of the Pearl River. I do not believe that downstream refugia influenced upstream populations due to faunal differences between the two regions.

As is common in most post-disturbance studies, rates of recolonization and recovery varied between species (Resh et al. 1988, Niemi et al. 1990, Albanese et al. 2009). Most notable among the interspecific responses to the fish kill was that of threadfin shad, which represented

93.5% of the sampled fish population in Picayune during the first sampling period but was nearly absent from later samples. High threadfin shad densities in Picayune during the first sampling season probably reflected a cohort that was able to invade and thrive in areas nearly free from predators following the initial disturbance. This conclusion is supported by the near absence of large piscivorous fishes during the first sampling period, when only four fish > 150 mm in length were captured in the entire region. When predators were present in Picayune during the next sampling season, threadfin shad were absent from the sample. Several studies have found high shad densities (up to 1000 per m³) in ponds and lakes recently disturbed by hurricanes (Rogers and Allen 2008, Alford et al. 2009), or in large reservoirs (Netsch et al. 1971, Michaletz and Gale 1999) where schools of juveniles are able to migrate diurnally to areas of low predation. However, I am not aware of any record of similarly extreme densities in lotic systems.

Most species that were dominant members of fish assemblages upstream of the disturbance, including blacktail shiners, bluegill (*Lepomis macrochirus*) and bullhead minnows (*Pimephales vigilax*), appeared to recolonize defaunated reaches quickly following the kill. All were common in South Bogalusa by the first sampling period, and none of these species differed significantly in abundance between disturbed and undisturbed regions by the second sampling period. Rapid recolonization by dominant and common species following a disturbance is typical in post-disturbance studies (Albanese et al. 2009), which is thought to reflect large source populations as well as the ability of these species to inhabit a variety of common habitat types in the impacted system (Gore and Milner 1990, Schlosser and Angermeier 1995, Albanese et al. 2009).

Species that were historically common in the Pearl River, but have since undergone population declines, were largely absent from my study. Most prior studies have found slow

recovery rates among similarly uncommon or rare species following disturbances (Ensign et al. 1997, Albanese et al. 2009). Slow recovery rates among these species are likely related to the same factors that cause them to be rare, i.e., slow maturation rates and low migration rates resulting from habitat scarcity and separation (Gore and Milner 1990, Allen et al. 2005). One of the few threatened species that I did detect was the southeastern blue sucker (*Cycleptus meridionalis*). Studies on southeastern blue sucker populations in the Pearl River indicate that the species is unlikely to quickly colonize disturbed regions due to a high degree of site specificity and low migration rates (Peterson et al. 2000, Oliver 2012). Although I only detected three southeastern blue suckers during my study, all were found in undisturbed regions. This result may reflect low recolonization rates by this species, although the sample size is too small to draw any definitive conclusions.

The absence of many of the Pearl River's historically prevalent species from my samples, combined with the high relative abundance of several generalists, likely reflects system-wide declines in diversity caused by long-term anthropogenic press disturbances and recent pulse disturbances, such as Hurricane Katrina (Tipton et al. 2004, Piller et al. 2004, Geheber and Piller 2012). Several studies based on long term monitoring data from the Pearl River and its tributaries have noted decreases in populations of once common benthic species, and relative increases in generalists that were common in my study such as blacktail shiners and longear sunfish (Gunning and Suttkus 1990, Stewart et al. 2005). A shift in the Pearl River's fish assemblages towards disturbance tolerant taxa may explain the apparent speed at which the Pearl River's fish assemblages recovered following the August 2011 fish kill. If assemblages in undisturbed regions had contained greater proportions of intolerant species historically common in the river, it is possible that diversity indices that I measured would have taken longer to

stabilize following the spill. As a consequence, the rate of recovery that I noted in this study may reflect the rate at which Pearl River fish assemblages are returning to the current, disturbance-tolerant regime rather than the rate at which disturbed assemblages are recovering to an undisturbed state.

Although fish diversity and assemblage recovery appeared to occur relatively quickly in disturbed regions, potentially due to the abundance of disturbance-tolerant taxa, it is important to note that fish assemblage recovery was not complete by the final sampling period, despite recoveries in species richness and evenness. Incomplete recovery is reflected primarily by NMDS and PERMANOVA results as well as by non-parametric ANOVA's of common species lengths. Incomplete recovery was most obvious at sites further away from the effluent source and mainstem source populations in Picayune, where fish assemblages were still significantly different from undisturbed regions by the final sampling period. Although fish assemblages in South Bogalusa were statistically similar to undisturbed sites during this period, PERMANOVA implied a greater degree of dissimilarity between assemblages in South Bogalusa and North Bogalusa, which were separated by only ~14.8 km, than between North Bogalusa and Columbia, which were separated by ~90 km. In the absence of external forces, proximate assemblages in lotic systems typically demonstrate a greater degree of similarity (Soininen et al. 2007, Brown and Swan 2010, Brown et al. 2011). This comparatively high similarity between unimpacted Bogalusa and Columbia assemblages is also particularly noteworthy given that other studies that have more samples, and therefore more statistical power, have detected differences between fish assemblages near these regions (Geheber and Piller 2012).

Results of non-parametric ANOVA's testing regional differences in fish lengths also implied residual impacts from the fish kill among Pearl River fish assemblages, particularly in

Picayune. Almost all of the selected fish species were significantly smaller in Picayune compared with undisturbed sites. Although lengths did not demonstrate the same level of difference at sites in South Bogalusa, three species were still significantly smaller in this region than in undisturbed locations. Other studies have also found biomass or fish length to be lower at recently recolonized sites relative to undisturbed sites (Olmsted and Cloutman 1974, Meade 2004). Monitoring studies indicate that biomass and length distributions eventually recover, but the process often takes several years (Detenbeck et al. 1992, Meade 2004).

3.5 CONCLUSION

My results indicate that, in less than a year, Pearl River fish assemblages demonstrated a substantial level of recovery following a catastrophic pulse-disturbance, particularly in areas closest to undisturbed portions of the river's mainstem. Despite a fast rate of recolonization, however, residual impacts of the disturbance on fish populations were still detectable ten months after the mill effluent spill. Additionally, these impacts were greater with increasing distance from the spill's source and mainstem source populations. These results imply that present-day fish assemblages in higher order rivers in the southeastern coastal plain can show a substantial level of resilience to pulse disturbances, given the presence of adequate source populations. However, recovery rates may not match the faster rates witnessed in the region's smaller order systems. Additionally, the apparent lack of tributary-based recolonization in early sampling periods emphasizes that recolonization patterns following a pulse disturbance can vary substantially between systems depending on environmental conditions at the time of the disturbance. Importantly, although the Pearl River and other similar southeastern coastal plain systems have the ability to recovery quickly following severe pulse disturbances, more research is needed to document how biotic recovery following pulse disturbances varies by disturbance

type, abundances, and distributions of mainstem and tributary source populations, and environmental conditions during and subsequent to the event.

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CHAPTER 4: SUMMARY

4.1 EFFECTS OF REVETTED BANKS AND THE MILL EFFLUENT SPILL

The results of my study indicate that diversity within the Pearl River's fish and macroinvertebrate communities, which have been severely impacted by past anthropogenic press disturbances (e.g., dam construction and gravel mining), may benefit from the presence of revetted banks during base flows. My results also imply that the fish assemblages in the Pearl River can demonstrate a high level of resilience to catastrophic pulse disturbances when source populations are present.

The high level of resilience of Pearl River fish assemblages to pulse disturbances was demonstrated the degree of recovery that I documented following the 2011 Temple Inland paper mill effluent spill and resulting fish kill. Despite complete defaunation, fish assemblages ~ 10 km downstream of the effluent's source were statistically similar to assemblages in undisturbed regions within six months of the pulse disturbance. Additionally, although fish assemblages ~40 km downstream of the spill were still statistically dissimilar to undisturbed assemblages by the final sampling season, dissimilarity appeared to be decreasing with successive seasons. However, despite these promising results, my data indicates that residual disturbance impacts were still present at regions affected by mill effluent by the final sampling season. Other than species abundance differences between my furthest downstream sampling sites and undisturbed regions; residual impacts were demonstrated by discrepancies between fish lengths at disturbed and undisturbed regions, as well as the greater degree of similarity between geographically distant undisturbed sites than between disturbed and undisturbed sites that were comparatively close to each other.

The potential benefits that revetted banks have on Pearl River diversity were

demonstrated by the fact that revetted banks supported unique assemblages when compared to natural banks in the summer and fall seasons. Observation, environmental measurements, and findings from other studies indicate these assemblage differences may reflect responses by several common fish species and macroinvertebrate families to the greater quantities of heterogeneous habitat provided by revetted banks when compared to natural banks during these months (Baker and Killgore 1992, Shields et al 1995). These potential benefits, however, seem to be limited to base flow conditions in the summer and fall seasons. During elevated flows in the winter and spring, assemblages did not differ between bank types. This seasonal discrepancy may reflect different behavioral patterns during the winter and spring season, or increases in the relative value of habitat present at natural banks at this time (McCargo and Peterson 2010).

I wish to stress that researchers and managers should be cautious when extrapolating my results to other systems. Excessive bank revetments can have severe hydrologic implications, and assemblages may exhibit different responses to these structures when a greater proportion of a river's banks are armored (Schmetterling et al 2001). Additionally, pulse disturbance impacts and recovery rates are influenced by many factors (Yount and Niemi 1990, Lake 2000). This means that the recolonization rates that I documented in this study may not reflect the rate or pattern of recovery following other pulse disturbances, even in the Pearl River and other local systems. Finally, given regime shifts among the Pearl River's fish community towards assemblages composed of disturbance tolerant species (Gunning and Suttkus 1990, Stewart et al. 2005, Geheber and Piller 2012); my results may not correspond to responses of undisturbed assemblage to the investigated stimuli. Historic Pearl River communities and current communities in less disturbed systems may not demonstrate the same increases in local diversity

along revetted banks or the fast recoveries rates following pulse disturbances documented in this study.

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APPENDIX A: FISH SURVEY DATA

Appendix A1: Fish abundance at revetted and natural banks, excluding data from the first sampling season at Picayune, which was not included in analysis.

Species	Scientific Name	Revetted Banks	Natural Banks	Total Collected
American eel	<i>Anguilla rostrata</i>	10	4	14
Atlantic needlefish	<i>Strongylura marina</i>	0	1	1
Black crappie	<i>Pomoxis nigromaculatus</i>	2	7	9
Black tail shiner	<i>Cyprinella venusta</i>	825	1152	1977
Blackspotted topminnow	<i>Fundulus olivaceus</i>	7	4	11
Blacktail redhorse	<i>Moxostoma poecilurum</i>	9	8	17
Blue catfish	<i>Ictalurus furcatus</i>	0	4	4
Bluegill	<i>Lepomis macrochirus</i>	358	134	492
Brook silverside	<i>Labidesthes sicculus</i>	1	1	2
Bullhead minnow	<i>Pimephales vigilax</i>	146	76	222
Channel catfish	<i>Ictalurus punctatus</i>	59	55	114
Clear chub	<i>Hybopsis winchelli</i>	0	3	3
Common carp	<i>Cyprinus carpio</i>	1	1	2
Emerald shiner	<i>Notropis atherinoides</i>	22	50	72
Flathead catfish	<i>Pylodictis olivaris</i>	14	5	19
Freshwater drum	<i>Aplodinotus grunniens</i>	21	10	31
Gizzard shad	<i>Dorosoma cepedianum</i>	5	14	19
Golden shiner	<i>Notemigonus crysoleucas</i>	2	0	2
Green sunfish	<i>Lepomis cyanellus</i>	3	0	3
Gulf darter	<i>Etheostoma swaini</i>	1	0	1
Highfin carpsucker	<i>Carpiodes velifer</i>	23	10	33
Hogchocker	<i>Trinectes maculatus</i>	1	0	1
Ironcolor shiner	<i>Notropis chalybaeus</i>	0	1	1
Large mouth bass	<i>Micropterus salmoides</i>	21	27	48
Longear sunfish	<i>Lepomis megalotis</i>	1105	387	1492
Longnose gar	<i>Lepisosteus osseus</i>	0	9	9
Longnose shiner	<i>Notropis longirostris</i>	6	3	9
Menhaden	<i>Brevoortia patronus</i>	1	0	1
Mimic shiner	<i>Notropis volucellus</i>	0	3	3
Mississippi silvery minnow	<i>Hybognathus nuchalis</i>	12	55	67
Pirate perch	<i>Aphredoderus sayanus</i>	15	0	15
Pugnose minnow	<i>Opsopoeodus emiliae</i>	0	8	8
Redear sunfish	<i>Lepomis microlophus</i>	15	2	17
Redspotted sunfish	<i>Lepomis miniatus</i>	24	7	31
Shadow bass	<i>Ambloplites ariommus</i>	8	0	8
Silver chub	<i>Macrhybopsis storeriana</i>	1	1	2
Silver jawed minnow	<i>Notropis buccatus</i>	0	1	1
Skipjack herring	<i>Alosa chrysochloris</i>	3	3	6
Smallmouth buffalo	<i>Ictiobus bubalus</i>	15	15	30
Southeastern blue sucker	<i>Cycleptus meridionalis</i>	1	2	3
Spotted bass	<i>Micropterus punctulatus</i>	84	34	118
Spotted gar	<i>Lepisosteus oculatus</i>	2	9	11
Striped mullet	<i>Mugil cephalus</i>	4	3	7
Threadfin shad	<i>Dorosoma petenense</i>	2	0	2
Warmouth	<i>Lepomis gulosus</i>	17	5	22
Weed shiner	<i>Notropis texanus</i>	0	3	3

Appendix A2: Regional numerical fish abundance during the fall sampling season

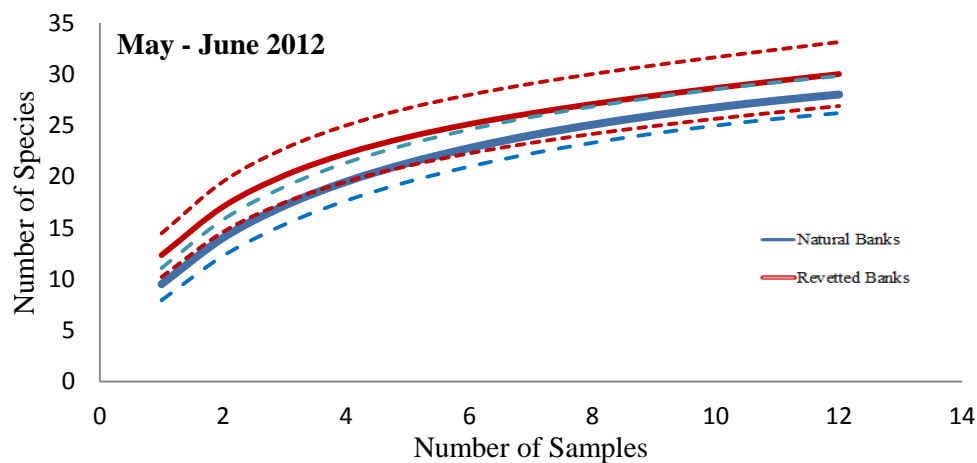
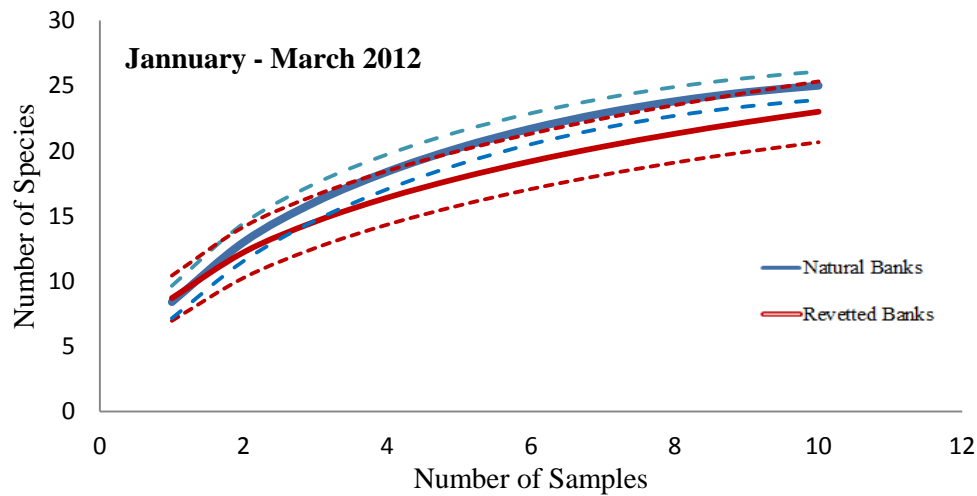
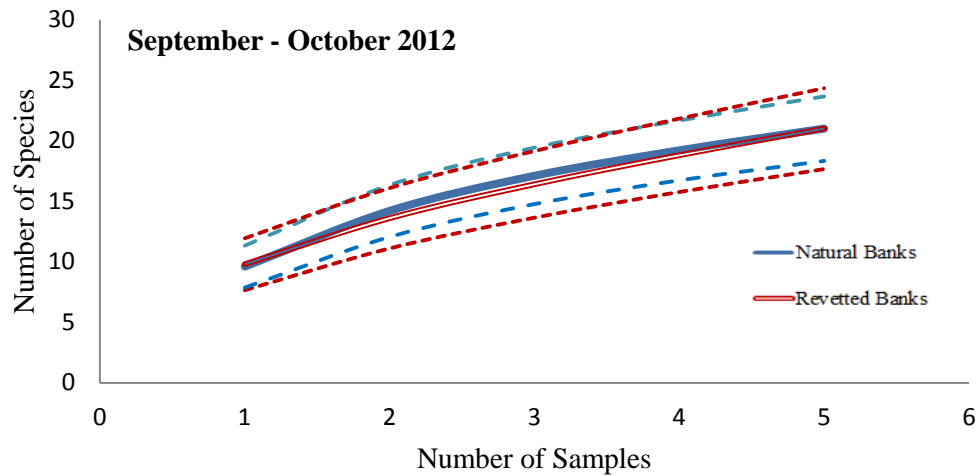
Species	Picayune	South	North
		Bogalusa	Bogalusa
American eel	0	1	1
Black crappie	0	0	5
Black tail shiner	56	30	257
Blackspotted topminnow	0	0	1
Blacktail redhorse	0	0	3
Bluegill	1	58	34
Bowfin	1	0	0
Bullhead minnow	0	20	6
Channel catfish	2	4	7
Common carp	0	1	0
Emerald shiner	0	3	20
Flathead catfish	0	2	1
Gizzard shad	3	0	0
Highfin carpsucker	0	3	0
Large mouth bass	0	0	4
Longear sunfish	1	73	96
Longnose gar	0	0	1
Longnose shiner	0	1	8
Mississippi silvery minnow	0	11	2
Pugnose minnow	0	3	0
Shadow bass	0	0	3
Silver chub	0	0	1
Silver jawed minnow	0	0	1
Smallmouth buffalo	1	1	2
Southeastern blue sucker	0	0	2
Spotted bass	0	9	26
Threadfin shad	936	0	0
Warmouth	0	1	0
White crappie	0	1	1

Appendix A3: Regional numerical fish abundance during the winter sampling season

Species	South		North	
	Picayune	Bogalusa	Bogalusa	Columbia
American eel	1	1	1	0
Black crappie	0	1	0	1
Black tail shiner	50	31	112	61
Blackspotted topminnow	3	0	1	0
Blacktail redhorse	0	0	2	1
Blue catfish	2	0	1	0
Bluegill	13	23	12	7
Bullhead minnow	3	8	7	5
Channel catfish	2	6	23	13
Clear chub	0	1	1	1
Common carp	0	0	1	0
Emerald shiner	7	5	4	4
Flathead catfish	0	1	3	0
Freshwater drum	2	1	6	2
Gizzard shad	3	2	5	2
Golden shiner	0	0	1	0
Highfin carpsucker	0	0	1	3
Ironcolor shiner	1	0	0	0
Large mouth bass	0	0	0	1
Longear sunfish	20	34	75	77
Longnose gar	4	1	0	0
Mississippi silvery minnow	0	0	3	0
Pugnose minnow	5	0	0	0
Redspotted sunfish	0	0	2	3
Shadow bass	0	0	2	3
Smallmouth buffalo	3	2	5	0
Southeastern blue sucker	0	0	0	1
Spotted bass	1	5	8	8
Spotted gar	0	1	0	0
Threadfin shad	0	0	1	0
Warmouth	2	0	0	1
White crappie	0	3	0	0

Appendix A4: Regional numerical fish abundance during the summer sampling season

Species	South				
	Picayune	Bogalusa	NBOG	COL	SLI
American eel	1	2	3	3	0
Atlantic needlefish	0	0	0	0	1
Black crappie	2	0	0	0	0
Black tail shiner	442	150	348	401	95
Blackspotted topminnow	1	0	1	3	1
Blacktail redhorse	2	0	1	6	2
Blue catfish	1	0	0	0	0
Bluegill	103	69	59	68	46
Brook silverside	0	0	0	2	0
Bullhead minnow	71	10	20	55	17
Channel catfish	22	6	4	26	1
Emerald shiner	17	1	4	1	6
Flathead catfish	1	1	4	5	1
Freshwater drum	8	3	2	7	0
Gizzard shad	0	4	2	1	0
Golden shiner	1	0	0	0	0
Green sunfish	0	0	2	1	0
Gulf darter	0	1	0	0	0
Highfin carpsucker	4	5	0	17	0
Hogchocker	1	0	0	0	0
Large mouth bass	5	6	4	6	22
Longear sunfish	326	100	157	398	136
Longnose gar	2	0	0	0	1
Menhaden	0	0	0	0	1
Mimic shiner	3	0	0	0	0
Mississippi silvery minnow	18	6	26	0	1
Pirate perch	11	1	2	0	1
Redear sunfish	0	0	0	0	17
Redspotted sunfish	3	1	2	3	17
Silver chub	0	0	1	0	0
Skipjack herring	0	0	3	1	2
Smallmouth buffalo	7	2	1	4	3
Spotted bass	16	8	10	18	9
Spotted gar	5	2	0	0	3
Striped mullet	2	0	0	0	5
Threadfin shad	0	0	0	1	0
Warmouth	10	1	2	2	3
Weed shiner	3	0	0	0	0
Western mosquitofish	1	0	0	0	0
White crappie	2	3	6	2	1



Appendix A5: Seasonal rarefaction curves depicting fish species richness at revetted and natural banks. Dotted lines represent 84% confidence intervals. Complete overlap between the confidence intervals implies species richness does not differ between bank types ($P > 0.05$).

APPENDIX B: MACROINVERTEBRATE SURVEY DATA

Appendix B1: Numerical and relative abundance of macroinvertebrate families at revetted and natural banks

Family	Natural Banks		Revetted Banks	
	Count	Relative	Count	Relative
Ameletidae	2	0.04	0	0
Ancylidae	2	0.04	1	0.01
Asellidae	0	0	2	0.03
Baetidae	8	0.16	10	0.13
Caloptyeridae	3	0.06	2	0.03
Carabidae	1	0.02	0	0
Ceratopogonidae	53	1.03	33	0.43
Chironomidae	3838	74.92	5382	69.62
Coengarionidae	45	0.88	18	0.23
Corbiculidae	0	0	2	0.03
Corduliidae	0	0	2	0.03
Elmidae	13	0.25	7	0.09
Ephemeridae	60	1.17	1	0.01
Gomphidae	7	0.14	1	0.01
Heptageniidae	394	7.69	565	7.31
Hydrachnidia	5	0.10	51	0.66
Hydropsychidae	455	8.88	1309	16.93
Hydroptilidae	0	0	1	0.01
Isonychidae	5	0.10	3	0.04
Leptoceridae	76	1.48	34	0.44
Leptohyphidae	6	0.12	2	0.03
Limnephilidae	1	0.02	0	0
Macromiidae	2	0.04	0	0
Macrovellidae	0	0	1	0.01
Palaemonidae	1	0.02	0	0
Perlidae	90	1.76	185	2.39
Perlodidae	0	0	1	0.01
Philopotamidae	5	0.10	4	0.05
Physidae	1	0.02	7	0.09
Pleuroceridae	4	0.08	0	0
Polycentropodidae	24	0.47	52	0.67
Simuliidae	12	0.23	27	0.35
Siphonuridae	1	0.02	0	0
Sisyridae	0	0	1	0.01
Tipulidae	2	0.04	5	0.06
Unionidae	2	0.04	18	0.23
Viviparidae	5	0.10	4	0.05

Appendix B2: Numerical and relative abundance of macroinvertebrate families on wood and rock substrates

Family	Rock		Wood	
	Count	Relative	Count	Relative
Ameletidae	0	0	2	0.02
Ancylidae	2	0.04	1	0.01
Asellidae	2	0.04	0	0
Baetidae	8	0.17	10	0.12
Caloptyeridae	0	0	5	0.06
Carabidae	1	0.02	0	0
Ceratopogonidae	37	0.80	49	0.59
Chironomidae	3051	66.17	6169	74.84
Coengarionidae	43	0.93	20	0.24
Corbiculidae	2	0.04	0	0
Corduliidae	2	0.04	0	0
Elmidae	11	0.24	9	0.11
Ephemeridae	55	1.19	6	0.07
Gomphidae	7	0.15	1	0.01
Heptageniidae	351	7.61	608	7.38
Hydrachnidia	23	0.50	33	0.40
Hydropsychidae	749	16.24	1015	12.31
Hydroptilidae	1	0.02	0	0
Isonychidae	4	0.09	4	0.05
Leptoceridae	50	1.08	60	0.73
Leptohyphidae	3	0.07	5	0.06
Limnephilidae	0	0	1	0.01
Macromiidae	0	0	2	0.02
Macrovellidae	1	0.02	0	0
Palaemonidae	0	0	1	0.01
Perlidae	125	2.71	150	1.82
Perlodidae	0	0	1	0.01
Philopotamidae	5	0.11	4	0.05
Physidae	1	0.02	7	0.08
Pleuroceridae	3	0.07	1	0.01
Polycentropodidae	29	0.63	47	0.57
Simuliidae	22	0.48	17	0.21
Siphonuridae	1	0.02	0	0
Sisyridae	0	0	1	0.01
Tipulidae	2	0.04	5	0.06
Unionidae	18	0.39	2	0.02
Viviparidae	2	0.04	7	0.08

Appendix B3: Numerical abundance of macroinvertebrate families in each region

Family	Columbia	North Bogalusa	South Bogalusa
Ameletidae	2	0	0
Ancylidae	0	0	3
Asellidae	0	1	1
Baetidae	7	6	5
Caloptyeridae	0	5	0
Carabidae	0	0	1
Ceratopogonidae	13	50	23
Chironomidae	1907	2598	4715
Coengarionidae	20	27	16
Corbiculidae	0	0	2
Corduliidae	0	2	0
Elmidae	3	12	5
Ephemeraidae	22	4	35
Gomphidae	1	6	1
Heptageniidae	314	349	296
Hydrachnidia	26	19	11
Hydropsychidae	412	1091	261
Hydroptilidae	1	0	0
Isonychidae	4	3	1
Leptoceridae	25	59	26
Leptohyphidae	1	6	1
Limnephilidae	0	1	0
Macromiidae	1	1	0
Macrovellidae	0	0	1
Palaemonidae	0	0	1
Perlidae	83	143	49
Perlodidae	1	0	0
Philopotamidae	0	8	1
Physidae	0	2	6
Pleuroceridae	4	0	0
Polycentropodidae	24	16	36
Simuliidae	4	29	6
Siphonuridae	0	0	1
Sisyridae	0	1	0
Tipulidae	3	2	2
Unionidae	1	1	18
Viviparidae	1	7	1

VITA

Jose Vazquez was born to a military family in Virginia Beach, Virginia in February 1985. Jose spent the next fourteen years living in and around military bases in California, Pennsylvania, Rhode Island, Florida and Puerto Rico, until returning to Virginia Beach where he graduated from Princess Anne High School's International Baccalaureate Program in 2003. Jose attended college at George Mason University, where he graduated cum laude with a Bachelors of Science in Biology in May 2007. During and after college, Jose worked as an intern, environmental researcher and educator at the Naval Research Laboratory, Earth Day Network, Marine Science Consortium, and James Hubert Blake High School, until eventually serving as a Fisheries Biologist with US Fish and Wildlife Service in Reno, Nevada. Jose enrolled in Louisiana State University in the fall of 2010, and after dealing with the research implications of a drought, a tropical storm, and a fish kill, finally graduated magna cum laude with a Masters of Science in Renewable Natural Resources with a concentration in Fisheries and Aquaculture in the spring of 2013.