A Thesis

by

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ABSTRACT

EFFECTS OF REOPENING A TIDAL INLET ON ESTUARINE-DEPENDENT NEKTON IN MESQUITE BAY, TEXAS, USA

Cedar Bayou, a natural tidal inlet, was recently dredged and reopened. The inlet allows for direct water exchange between the Gulf of Mexico and Mesquite Bay, TX, USA. The goals of this study were to: 1) quantify what changes occurred to juvenile nekton densities in Mesquite Bay after Cedar Bayou was reopened, 2) document changes in nekton communities as a forage base, 3) define how historical nekton populations were influenced by Cedar Bayou's flow status, and 4) determine if adult Red Drum utilized Cedar Bayou as a migration route to access the Gulf of Mexico. Juvenile fish and crustaceans were sampled using an epibenthic sled in *Halodule wrightii* seagrass beds at both control and impact sites one year before (October 2013 – April 2014) and after (October 2014 – April 2015) opening. Using a before-after-control-impact design, significantly higher densities of total nekton were observed after-opening. Individual estuarine-dependent nekton species, including Red Drum (*Sciaenops ocellatus*), Atlantic Croaker (*Micropogonias undulatus*), post-larval penaeid shrimp (*Farfantepenaeus aztecus*, *F. duorarum*, and *Litopenaeus setiferus*), and Blue Crab (*Callinectes sapidus*), were also observed to have significantly higher densities after-opening. Multivariate analysis demonstrated a significant change in impact site community assemblage after-opening with an increased presence of estuarine-dependent prey species. The Boosted Regression Tree (BRT) models showed that historic populations for numerous species were significantly higher when Cedar Bayou was open, and densities were highest when nearer to open inlets. The recent field sampling data aligned well with the BRT results, showing that models are able to predict a tidal inlet's influence on an ecosystem and can help justify the need for continued maintenance of inlet flow. Adult Red Drum were also implanted with acoustic transmitters to determine their movement patterns before and after-opening Cedar Bayou to assess the impact to adult estuarine-dependent fish populations. Spatial movement data revealed that after-opening fish used Cedar Bayou during times commonly associated with spawning migrations, which were not apparent before-opening. The overall results of this study demonstrate that juvenile estuarine-dependent nekton are using Cedar Bayou to access previously inaccessible nursery habitat within Mesquite Bay and newly matured Red Drum are using Cedar Bayou as a migration pathway to offshore spawning grounds. In addition, BRT modeling indicates that historical nekton populations were influenced by Cedar Bayou's flow status. These results suggest higher recruitment and productivity of numerous species of economic and environmental importance with the opening of Cedar Bayou. Overall, these studies clearly demonstrate opening Cedar Bayou and reconnecting Mesquite Bay to the Gulf of Mexico led to an increased presence of numerous species that are vital to Texas' ecology and economy. The inlet is providing robust connectivity between productive estuary areas and the open Gulf of Mexico and should reinstitute natural processes vital to the ecological stability of the Aransas, Mesquite, and San Antonio Bay regions.

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GENERAL INTRODUCTION AND RATIONALE:
THE VALUE OF TIDAL INLETS

Global fish stocks have been depleted as over 1 billion people rely on fish as their primary source of protein (Pauly et al. 2002; FAO 2015). It is estimated that 158 million metric tons of fish were harvested world-wide in 2012 alone and the demand for fish has continued to grow with the burgeoning human population (FAO 2015). This led the United States to pass the Sustainable Fisheries Act (SFA) in 1996 to help ensure long-term persistence of fish stocks. Fisheries resources are more likely to be renewable and sustainable if the key habitats they rely upon are available. Therefore the SFA stipulates all fisheries management plans must include management strategies for Essential Fish Habitat (EFH). An increased understanding of the importance of EFH has become a key component of effective fisheries management (Rosenberg et al. 200l; Jose 2014). Ever increasing focus on EFH and ecosystem-based fisheries management, which manages entire ecosystems including their habitat and recognizes the importance of interactions between multiple species and abiotic environmental factors, has led to the recovery of numerous fish stocks (Pikitch et al. 2004; Jose 2014).

Tidal inlets are vital access points for nekton to reach key nursery habitat such as seagrasses and marshes. These habitats are considered to be some of the most productive habitats within estuaries (Levin et al. 1997). They serve as essential nurseries for numerous recruiting estuarine-dependent species by providing structurally complex habitat, (Heck et al. 2003), protection from predators, (Stunz and Minello 2001), and support for increased growth rates (Heck et al. 2003, Stunz et al. 2002 A & B). Access to these nursery habitats is critical at key life phases, and tidal inlets allow ingression of young recruits from offshore spawning grounds into these nursery areas (King 1971; Stunz and Minello 2001; Reese et al. 2008; Bushon 2006). However, anthropogenic and natural factors have led to the closure of several inlets along the Texas coast (Bermudez et al. 2005; Kraus 2007) including Cedar Bayou, which is the only direct link between the Mesquite Bay system and the Gulf of Mexico. Given the logistical and financial challenges associated with dredging closed inlets, reopening projects are very difficult and rare. Despite these challenges, support from the local government, conservation groups, and concerned citizens led to a comprehensive restoration of Cedar Bayou, including dredging, hydrology restoration, and a monitoring program to assess changes in nekton abundance, density, length, and movement in the region before and after Cedar Bayou was reopened.

CHAPTER 1

CHANGES TO JUVENILE NEKTON ABUNDANCE, MEAN DENSITY, MEAN LENGTH, AND COMMUNITY STRUCTURE

Introduction

Access to estuaries through tidal exchanges is an essential and defining life-history trait for estuarine-dependent organisms. Approximately 75% of commercially and recreationally valuable species in the Gulf of Mexico are dependent on access to estuaries for at least one critical stage of their development (Chambers 1991). Water exchange and biological connectivity between the various bays along the Texas coast and the Gulf of Mexico are regulated by tidal inlets that traverse through the barrier islands typical of the region. Research has shown that inlets play an important role in providing juvenile estuarine-dependent nekton access to estuarine nursery habitats such as seagrass meadows (*Halodule wrightii*) (King 1971; Reese et al. 2008), salt marshes (Weinstein 1979; Baltz et al. 1993; Jenkins and Black 1994; Jenkins et al. 1997; Minello 1999; Stunz and Minello 2001; Brown et al. 2004), and oyster reef (Nevins et al. 2013). These habitats provide juvenile nekton with protection from predation and increased food resources leading to increased growth rates and recruitment into the larger size classes (Rooker et al. 1998 A; Rooker et al. 1998 B; Stunz et al. 2002 A; Neahr et al. 2010).

Cedar Bayou is a natural ephemeral tidal inlet (Shepsis and Carter 2007, Kraus 2007) that separates Matagorda from San Jose Island and historically allowed water exchange between Mesquite Bay (a satellite bay of the San Antonio Bay complex) and the Gulf of Mexico. The mouth of the inlet was intentionally closed in 1979 to prevent contaminants from the Ixtoc oil spill from reaching Texas bays and the mainland (Gundlach et al. 1981; Ward 2010). The earliest assessment of Cedar Bayou was conducted by the Coast Survey in 1858 (Gilbert 1859), with additional hydrological, geological, and biological studies being conducted intermittently through the 1990's (Ward 2010). Despite large gaps in quantitative information regarding the bayou's features throughout time, Cedar Bayou was generally open and flowing prior to 1950 (Ward 2010). Subsequent weather events and coastal engineering-induced changes led to numerous closures and openings of the inlet, including extensive and mainly unsuccessful dredging projects in 1939, 1959, 1987-88, and 1995. Studies have suggested that scouring from flooding and freshwater inflow into the coastal bay system played a critical role in naturally reopening and maintaining flow in Cedar Bayou (Simmons and Hoese 1959; Copeland 1966). Recent analyses have shown that despite numerous severe weather events and periods of appropriately strong inflow (>493,393 m^3/mo; Ward 2010), the inlet has experienced long periods of severely restricted flow and reoccurring closures (see Bermudez et al. 2005).

Due to Cedar Bayou's diminishing cross-sectional area ($\leq$ 9.3 m^2 between 1980-2010), its contemporary role as a major avenue for juvenile nekton recruitment warrants further investigation (Ward 2010). There have been a few previous studies that examine the effects of opening tidal inlets on estuarine nekton. Opening a small tidal inlet in southwestern Florida did not significantly increase densities of estuarine-dependent species in surrounding seagrass nursery habitat, likely due to its proximity to other larger inlets and existing high levels of water exchange (Milbrandt et al. 2012). In contrast, I hypothesized that opening Cedar Bayou would play a more substantial role in establishing biological connectivity between the Gulf of Mexico and Mesquite Bay given its distance from the nearest open inlets which are Aransas Pass, located approximately 32 km to the southwest, and Cavallo Pass that is located approximately 55 km to the northeast. For example, results from a similar study in an analogous system found that reopening Packery Channel (an isolated tidal inlet near Corpus Christi, TX) resulted in significantly increased juvenile densities of many species including Red Drum (*Sciaenops ocellatus*), Atlantic Croaker (*Micropogonias undulatus*), post-larval penaeid shrimp (*Farfantepenaeus aztecus*, *F. duorarum*, and *Litopenaeus setiferus*), and Blue Crab (*Callinectes sapidus*) in the surrounding area (Bushon 2006; Reese et al. 2008).

The increase of Atlantic Croaker, post-larval penaeid shrimp, and Blue Crab described by Bushon (2006) and Reese et al. (2008) is particularly important given that these species are listed as being “key” to Texas’s $200 million per year commercial fishing industry. More important are the prominent roles Red Drum and Atlantic Croaker play in Texas’s recreational fishing industry, which generated $1.4 billion in expenditures during 2012 alone (NMFS 2014). Another estuarine-dependent species, Southern Flounder (*Paralichthys lethostigma*), is also targeted recreationally and commercially in Texas. Despite revisions to fishing regulations, the Texas Parks and Wildlife Department’s (TPWD) fishery independent monitoring program has recorded a decrease in Southern Flounder throughout the state over the past 25 years (GSMFC 2000; TPWD 2003; Nañez-James et al. 2009). The response of these particular species to the reopening of Cedar Bayou is of particular interest given their economic importance within the state of Texas and Gulf of Mexico in general.

In addition to Cedar Bayou's role in delivering economically important nekton to nursery areas, the inlet serves as a feeding ground for endangered Whooping Cranes (*Grus americana*) that overwinter at the Aransas National Wildlife Refuge (Gil-Weir et al. 2012). These cranes rely on Blue Crab in and near Cedar Bayou for 41% of their winter diet (Westwood and Chavez-Ramirez 2005), with periods of lower Blue Crab abundance being correlated to increased winter mortality for Whooping Cranes (Pugesek et al. 2008). Given that the reopening of inlets has been shown to increase juvenile Blue Crab density (Reese et al. 2008), the reopening of Cedar Bayou may play a key role in increasing the food supply for Whooping Cranes overwintering in the region.

Cedar Bayou, when open, is the only connection through Matagorda Island in the northern Aransas Bay region that allows nekton to reach estuarine nurseries from spawning grounds in the Gulf of Mexico. This connection, or lack thereof, likely drives the ecology in the region. Environmental concern prompted the local government and concerned citizens to petition the U.S. Army Corps of Engineers for permits authorizing a dredging project that would permanently reopen the inlet by April 2014. There was immense interest by the scientific, birding, and fishing communities in how the Mesquite Bay region would change due to the reopening of Cedar Bayou. The restoration of flow to Cedar Bayou provides a rare opportunity to elucidate the direct impacts of tidal inlets on surrounding estuarine seagrass habitats and the juvenile nekton that often use them. Thus, the overall goal of this project was to assess changes in nekton densities in the Mesquite Bay region before and after Cedar Bayou was reopened. Specifically, this study addressed the following objectives:

1. Determine if seasonal relative abundance, density, and length of juvenile nekton (fish, shrimp, and crab) species change near Cedar Bayou after the inlet is dredged and re-opened.

   H_{A1}: Seasonal nekton relative abundance, density, and length at the impact sites are significantly different before and after-opening.

2. Assess nekton productivity as a prey resource for economically and ecologically important species in Mesquite Bay before and after-opening.

   H_{A2}: Community assemblages, particularly prey species assemblages, are significantly different at the impact sites between years.

Methods

Study Site

Mesquite Bay is the southernmost component of the larger San Antonio Bay estuary complex (Armstrong 1987; Britton and Morton 1989). It is bordered to the north by San Antonio Bay proper and to the southwest by the Aransas Bay complex (Figure 1). Given its position on the landward side of the Matagorda and San Jose barrier islands, Mesquite Bay is very isolated from the waters of the Gulf of Mexico when Cedar Bayou is closed.

A survey conducted by Coast and Harbor Engineering (Austin, TX) in April of 2005 found that Cedar Bayou's seaward mouth was open with dimensions of 91 m in width and a depth of 1.2 m below mean sea level (MSL) (Bermudez et al. 2005). The gradual closure of natural inlets has been common along the Texas coast. Of the 17 inlets found along the Texas coast, Cedar Bayou was one of six natural inlets considered "open" in January 2007 (Kraus 2007). Ward's (2010) comprehensive timeline of the inlet agrees that marginal flow was intermittently present during the 2005 to 2007 time period; however, it also indicates the inlet reclosed sometime in 2007 and remained so until the reopening in September 2014.

Cedar Bayou has changed course several times throughout its history and came to rest in its present position within the last 2,000 to 2,500 years (Figure 1) (Wilkinson 1975; Ward 2010). Its northeast (45^o) to southwest orientation (Bermudez et al. 2005) is characteristic of inlets in the Gulf Coast region and is due to in part to the northern location of Mesquite Bay to the barrier island complex (Price 1952). The inlet is approximately 4.8 km in length. Prior to dredging, the northern two-thirds of Cedar Bayou measured approximately 179.8 m in width with a mean depth of 1.9 m below MSL. The lower third of the inlet is morphologically dynamic and is subject to constant variation in width and depth (Bermudez et al. 2005).

Study Design and Sample Site Delineation

This study was designed to assess the effect of reopening Cedar Bayou on nekton relative abundance, densities, and lengths using a before-after-control-impact (BACI) experimental design. The BACI concept seeks to determine if an event, such as reopening Cedar Bayou, influences specified and predetermined ecological variables (Smith 2002). There are numerous variations of before-after designs (Eberhardt 1976; Green 1979; Schwarz 2012), and this particular study used a BACI design originally proposed by Stewart-Oaten et al. (1986) that was further refined for determining impacts of reopening tidal inlets by Reese et al. (2008) and Milbrandt et al. (2012).

The impact sites established using four sampling locations within the immediate vicinity of the Cedar Bayou complex, and the control group comprised of four sampling locations adjacent to Aransas Pass. The distance of the control sites from Cedar Bayou (approximately 32 km) was anticipated to be far enough as to not be altered by its reopening. The nekton densities at the control sites were assumed to be representative of a healthy estuarine system influenced by a nearby tidal inlet.

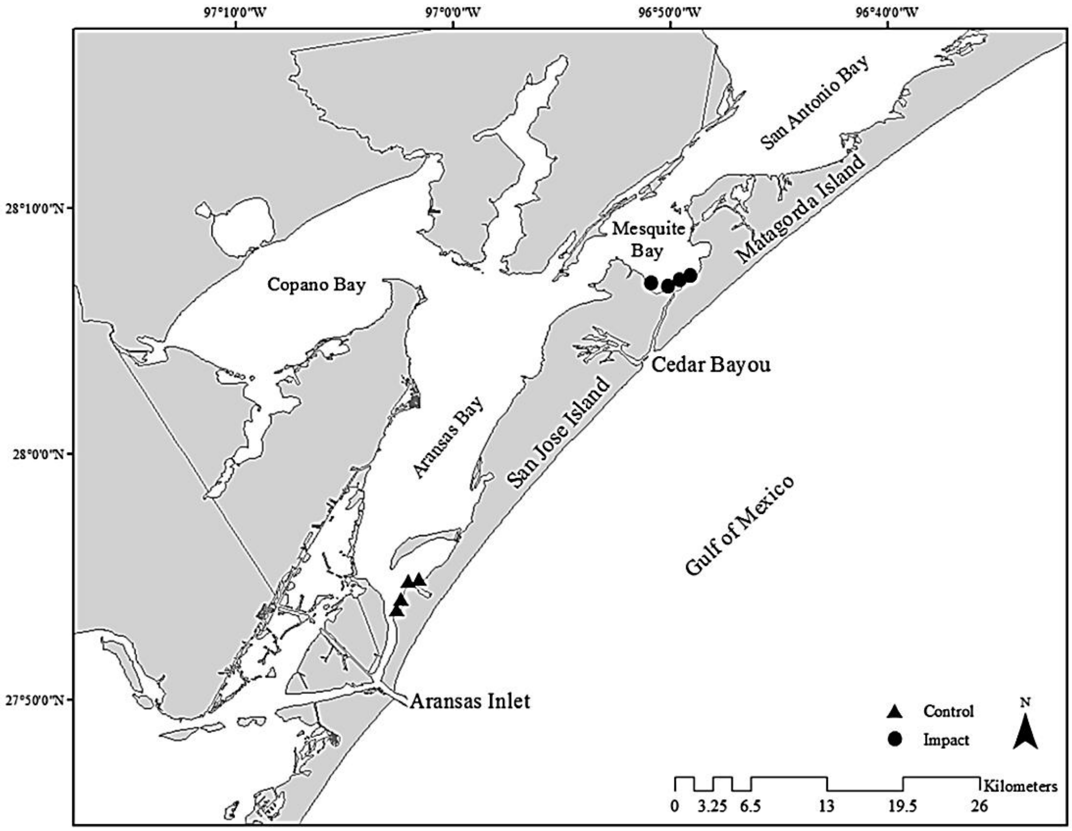

Figure 1: The Cedar Bayou study area. Individual control sites (triangles) were established near the Aransas inlet while impact sites (circles) were located near Cedar Bayou in seagrass meadows (*Halodule wrightii).* Two sampling trips were conducted in each of the three recruitment seasons (fall, winter, spring) over a two year period (one year of before-opening, one year after-opening). Each site was sampled three times per trip totaling 288 samples over the entire study period.

Sampling Procedure

Nekton samples were collected using an epibenthic sled. This device has been well-established in the literature as an efficient gear for sampling small nekton (Stunz et al. 2002 B; Reese et al. 2008; Neahr et al. 2010) in seagrasses and other estuarine habitat types. It is comprised of a metal frame 0.6 m wide by 0.75 m high, which supports a 1-mm mesh conical plankton net mounted to skids. Each tow consisted of pulling the sled 16.6 m covering a 10 m^2 sampling area.

Samples were collected six times in the sampling year before Cedar Bayou was reopened, and six times in the one year after-opening totaling 288 samples over the course of the two year study period (Table 1). Samples were collected in the spring, fall, and winter, which are established seasonal high-tide recruitment and settling periods in seagrass beds for ecologically important estuarine-dependent species. Three replicate epibenthic sled tows were taken at each of the eight sites during each sampling event. There were two sampling events during each recruitment season (fall, winter, spring). Samples from each individual tow were rough sorted in the field and preserved in 10% buffered formalin (Reese et al. 2008).

Fish and crustaceans in each sample were sorted, counted, identified to the lowest possible taxon, and measured to 0.1 mm total length (TL) in the laboratory. Shrimp were measured for TL between the tip of rostrum and the telson, while crab species were measured using carapace width (CW). If more than 22 individuals of the same species were collected in a single tow, the largest, smallest, and 20 randomly selected individuals were measured. This method made the assumption that the length measurements of the randomly sampled individuals were representative of the entire size distribution in the tow. Once a sample was processed, organisms were preserved in 70% ethanol for long-term storage. At each site, water temperature (^{o}C), dissolved oxygen (mg L^{-1}), and salinity were recorded during each sampling event using a HydroLab MS 5 sonde.

Table 1: Nekton density sample dates before and after-opening.

	Before-opening		**After-opening**	
Season	**Trip 1**	**Trip 2**	**Trip 1**	**Trip 2**
Fall	10/29/2013	11/14/2013	10/23/2014	11/10/2014
Winter	2/14/2014	2/25/2014	2/11/2015	2/18/2015
Spring	3/31/2014	4/7/2014	4/2/2015	4/13/2015

Statistical Analysis

Physical Parameters

Changes in water temperature (°C), dissolved oxygen (mg L^{-1}), and salinity were tested for significance at the control and impact locations before and after-opening. A one-way analysis of variance (ANOVA) ($\alpha = 0.05$) was conducted using the linear mixed effects (*.lme*) function in R 3.1.3 (Pinheiro et al. 2015) for the control and impact sites for each parameter over each season, where the parameter value was explained by the main before-after factor.

Nekton General Comparisons

Mean density (m^{-2}), mean length (mm), relative abundance (RA %), and change in relative abundance (Δ RA %) estimations were calculated for each species for each recruitment season at the control and impact sites following Reese et al. (2008). Each mean density was calculated from a total of 24 samples collected each season at the control and impact sites (totaling 48 samples per season per year). Mean lengths (total length for fish, total length for shrimp, and carapace width for crabs) were calculated from the number of individuals of a species measured during each season before and after-opening. The RA (%) was calculated by dividing the number of individuals of a species caught by the total number of fish or crustaceans within a particular season and multiplied by one hundred to obtain a percentage value. After-opening RA (%) was subtracted from the before-opening RA (%) to calculate Δ RA %. A negative change in relative abundance indicates a decline in relative abundance, while a positive number indicates an increase in relative abundance.

Primary Recruitment Seasons

Data used to test for significant differences in this BACI design were restricted to each species peak recruitment season (Reese et al. 2008). Total nekton, total fish, total crustaceans, and aggregated shrimp densities (ASD) were tested over all sample seasons. Aggregated shrimp density was calculated by summing the densities of Arrow Shrimp (*Tozeuma carolinense*), *Hippolytidae* sp., and *Palaemonetes* sp., which comprised over 85% of the total crustaceans sampled during the course of this study. Mean density and size of Red Drum were determined using fall samples only (Holt et al. 1983; Rooker et al. 1997; Rooker et al. 1998 B; Reese et al. 2008; Stunz et al. 2002 B). Mean densities and sizes of Southern Flounder and Atlantic Croaker were determined using winter samples only (Haven 1957; Hansen 1969; Rooker et al. 1998 B; Searcy et al. 2007; Nañez-James et al. 2009). Post-larval penaeid shrimp mean density and size were calculated by combining fall, winter, and spring samples. Given that Blue Crab have an exceedingly complex life history, disperse widely, and are known to spawn all year, mean density and size were determined by combining fall, winter, and spring samples (Pile et al. 1996; Blackmon and Eggleston 2001; Reese et al. 2008).

Nekton Density and Length Comparisons

Data were analyzed with ANOVA in a BACI design using the linear mixed effects (*.lme*) function in R 3.1.3 (Pinheiro et al. 2015) to identify changes in density and length due to the opening of Cedar Bayou. Each main effects model used a two-way nested ANOVA (BA*site(CI)) that tested for a significant interaction (α = 0.05) between the before-after (BA) and control-impact (CI) main factors using type III sum of squares. Site was treated as a random factor nested in the CI factor. The main effects ANOVA was conducted for every group (total organisms, etc.) and species of interest. Specifically, the *.lme* function allows for random, nested factors and uses restricted maximum likelihood (REML) to generate a set of contrasts calculated from the original data. The REML technique is capable of producing unbiased estimates of variance parameters while ensuring nuisance parameters have no effect. All data were log (*x*+1) transformed to ensure homogeneity of variance and normality of the residuals (Zuur et al. 2007; Reese et al 2008). Density data were balanced as a result of experimental design. Length data were unbalanced given that the number of individuals of a species varied by tow. For this reason, all ANOVAs used to test length data used Helmert contrasts.

To further test for differences within impact locations, BA and CI factors were combined using the "*paste*" function in R 3.1.3 to create a single factor with four levels: before-control (BC), after-control (AC), before-impact (BI), and after-impact (AI). Site remained nested within the new combined BA:CI factors. Then one-way "post-hoc" ANOVAs (α = 0.05) were used to test for differences within impact and control locations before and after-opening for all groups and individual species of interest (BC*AC and BI*AI).

Community Analysis

Multivariate analyses were implemented to test for differences in community assemblage at the impact and control locations over the course of the study using statistical procedures from PRIMER (v.6 PERMANOVA +). Mean densities for each species were calculated by date for control and impact sites. All data were fourth root transformed before analysis to reduce the importance of more abundant species and allow for changes in rarer species to be statistically discerned (Clarke and Green 1988; Reese et al. 2008).

A Bray-Curtis resemblance matrix was generated from daily mean densities for both control and impact sites before and after-opening. This matrix was used to generate a nonmetric multidimensional scaling (MDS) ordination which allowed for visual comparison of control site and impact site communities throughout time (Clarke and Warwick 2001, Reese et al. 2008; Ajemian et al. 2015). A permutational multivariate analysis of variance (PERMANOVA) was then calculated using type III sum of squares. This method is widely used to detect differences in the locations of multivariate groups (Anderson and Walsh 2013; Ajemian et al. 2015). PERMANOVA tests the null hypothesis that ‘‘the centroids of the groups, as defined in the space of the chosen resemblance measure, are equivalent for all groups’’ and generate pseudo-F ratios by partitioning distance matrices for multivariate data and conducting permutations to make the data distribution-free (Anderson and Walsh 2013). They are robust to heterogeneity when used in balanced designs (Anderson and Walsh 2013). Pair-wise tests were also performed on significant factors to determine if control site communities and impact site communities changed between the before and after-opening years (Ajemian et al. 2015). To determine the exact response of the impact location community, separate Bray-Curtis resemblance matrices were constructed for just the impact sites both before and after-opening. These matrices were tested for differences using the RELATE package, (Clarke and Gorely 2006; Reese et al. 2008). The RELATE routine performs a rank correlation and compares the results to randomly permuted samples, and tests the null hypothesis that no correlation exists between the similarity matrices (Clarke and Gorely 2006; Reese et al. 2008). An MDS ordination based on Bray-Curtis similarity was also used to compare impact site community changes. Bray-Curtis cluster groups were superimposed on the MDS ordination for interpretation (Clarke and Warwick 2001, Reese et al. 2008). Finally, a two-way crossed similarity of percentages (SIMPER) analysis was conducted to determine which overall species densities were the most dissimilar between before and after samples. All seasons were combined for these comparisons as community assemblages are known to change seasonally (Reese et al. 2008).

Results

Physical Parameters

Expected seasonal differences were apparent for water temperature (^oC), dissolved oxygen (mg L^{-1}), and salinity at the sample sites (Table 2). For the control sites, within season water temperature was not different for the fall (p=0.632), winter (p=0.344), or spring (p = 0.124) between the before and after-opening samples.

Dissolved oxygen at the control sites did not vary during the fall ($p = 0.164$) or winter ($p = 0.159$); however, dissolved oxygen was lower during the spring ($p <0.001$). Control site salinity levels increased during the fall ($p = 0.020$) and winter ($p = 0.016$) after-opening, while spring salinities were similar ($p = 0.292$) during the before and after-opening years (Table 2). Statistically significant differences in physical water parameters were also observed at the impact locations (Table 2). Water temperatures during the fall and winter seasons were similar before and after-opening ($p = 0.318$ and $p = 0.808$, respectively); however, spring water temperatures increased after opening ($p < 0.001$). Dissolved oxygen levels decreased during the fall season before and after-opening ($p = 0.003$), but remained similar during the winter ($p = 0.931$) and spring ($p = 0.410$). Significant increases in salinity were found at the impact sites during the fall ($p = 0.013$) and winter ($p = 0.022$), while spring samples indicated a decrease ($p = 0.048$) between the before and after-opening years. Additionally, we found differences in water quality parameters between the control and impact sites, however these differences were minimal and likely not biologically significant. See Appendix A for complete ANOVA table.

Table 2: Mean physical parameters (water temperature (^{o}C), dissolved oxygen (mg L^{-1}), and salinity), for control and impact sites before-opening (October 2013 – April 2014) and after-opening (October 2014 – April 2015). Mean values and standard error (SE) were calculated from measurements taken at each sample site twice per season. A one-way ANOVA ($\alpha = 0.05$) was conducted to determine changes at the control and impact sites before and after reopening. An * indicates a significant change ($p<0.05$). See Appendix A for complete ANOVA table.

	CONTROL					**IMPACT**				
	Before Opening		After Opening			Before Opening		After Opening		
	Mean	SE	Mean	SE		Mean	SE	Mean	SE	
FALL										
Dissolved Oxygen (mg L^{-1})	10.35	(0.69)	11.74	(0.65)		7.97	(0.35)	6.58	(0.19)	*
Water Temperature (^{0}C)	21.87	(2.32)	23.05	(0.65)		19.72	(2.20)	22.04	(0.47)	
Salinity	32.19	(0.75)	34.35	(0.34)	*	30.89	(0.74)	33.12	(0.28)	*
WINTER										
Dissolved Oxygen (mg L^{-1})	7.96	(0.67)	9.02	(0.25)		8.90	(0.30)	8.86	(0.26)	
Water Temperature (^{0}C)	16.66	(0.71)	14.63	(1.95)		16.31	(1.45)	16.78	(1.24)	
Salinity	29.95	(0.12)	31.04	(0.38)	*	27.72	(0.07)	28.64	(0.35)	*
SPRING										
Dissolved Oxygen (mg L^{-1})	9.78	(0.50)	5.94	(0.19)	*	7.82	(0.03)	7.68	(0.16)	
Water Temperature (^{0}C)	22.70	(0.55)	21.58	(0.40)		18.90	(0.80)	24.94	(0.23)	*
Salinity	30.28	(0.15)	30.92	(0.56)		29.35	(0.10)	28.12	(0.56)	*

Overall Nekton Relative Abundance and Species Characterization

A total of 126,812 organisms were collected during the course of this study representing 31 species of fish and 7 species of crustaceans. For some taxa, juveniles were not identified to species, but the lowest possible taxon. These included *Clupeiformes*, *Gobiidae*, and *Gobiosoma* for fish and *Xanthidae*, *Hippolytidae*, *Penaeidae*, and *Palaemonetes* for crustaceans. As is characteristic of estuarine species, seasonal differences were found in nekton mean density (m^{-2}), mean length (mm), RA %, and Δ RA % both before and after-opening in control sites (Table 3). Darter Gobies (*Gobionellus boleosoma*) (33.2%), Red Drum (23.0%), and *Syngnathus* sp. (19.4%), were the most abundant fish at the fall before-opening control sites (Table 3). During fall after-opening, these same species were also found to be the most abundant at control sites with Darter Gobies being the most abundant (51.3%), followed by *Syngnathus* sp.

(27.0%), and Red Drum (10.2%). The most abundant crustaceans during the fall before-opening control sites were *Hippolytidae* sp. (68.2%), Arrow Shrimp (18.3 %), and *Palaemonetes* sp. (12.3%). As with the fish abundances, the same three crustacean species were most abundant for the fall after-opening control sites with *Hippolytidae* sp. being the most abundant (44.3%), followed by *Palaemonetes* sp. (36.1%), and Arrow Shrimp (13.0%). The most abundant fish species at the winter before-opening control sites were Spot (*Leiostomus xanthurus*) (44.9%), Pinfish (*Lagodon rhomboides*) (28.9%), and Darter Gobies (15.9%). After-opening winter samples at the control sites also showed these same species as being most abundant with Darter Gobies having the highest abundance (36.6%), followed by Spot (22.5%), and Pinfish (16.3%). Crustacean abundances for winter before-opening control site samples were dominated by *Palaemonetes* sp. (50.5%), *Hippolytidae* sp. (38.9%), and Arrow Shrimp (8.5 %). After-opening winter control site crustacean abundances followed the same pattern with *Palaemonetes* sp. (59.2%), *Hippolytidae* sp. (27.6%), and Arrow Shrimp (10.6%), being most abundant. Before-opening spring control site fish abundances were dominated by Pinfish (55.4%), Darter Gobies (31.6%), and Spot (3.9%). After-opening spring control site fish abundances remained similar to the before-opening values with Pinfish (56.9%), and Darter Gobies (22.1%) remaining the most abundant fish, but with *Syngnathus* sp. (9.0%) being more abundant than Spot. Before-opening spring control crustacean abundances were comprised of mostly of *Hippolytidae* sp. (34.0%), *Palaemonetes* sp. (23.7%), and post-larval penaeids (23.0%). These same species continued to dominate the crustacean abundances at the spring after-opening control sites with the highest abundance of *Hippolytidae* sp. (41.9%), followed by *Palaemonetes* sp. (30.2%), and post-larval penaeids (14.2%).

The influence of opening Cedar Bayou was most apparent at the impact sites as there were seasonal differences in nekton mean density (m^{-2}), mean length (mm), and RA % (Table 4). The most abundant fish at the fall before-opening impact sites were *Gobiosoma* sp. (73.7%), *Syngnathus* sp. (23.2%), and Atlantic Croaker (2.0%). After-opening, *Gobiosoma* sp. (43.8%) and *Syngnathus* sp. (18.7%) remained the most abundant fish species at the fall season impact sites. However, Red Drum RA % went from 0% at the impact sites during the fall before-opening portion of the study to 14.8% after-opening, becoming the third most abundance fish species. Fall before-opening crustacean abundances were led by *Hippolytidae* sp. (40.8%), Arrow Shrimp (39.3%), and *Palaemonetes* sp. (12.8%). Fall after-opening crustacean abundances were comprised of mainly *Hippolytidae* sp. (47.9%), *Palaemonetes* sp. (21.2%), and post-larval penaeids (15.2%). Winter before-opening impact sites had high relative abundances of Spot (42.5%), Sheepshead Minnow (*Cyprinodon variegatus*) (18.9%), and Southern Flounder (12.3%). After-opening samples showed changes in fish abundances at the impact sites during the winter season. While Spot were still the most abundant fish species after-opening (24.7%), Atlantic Croaker were nearly as abundant (23.5%). Pinfish also increased to be the third most abundant fish at the winter-after-opening impact sites (12.1%). Crustacean abundance in winter before-opening impact samples was comprised of *Hippolytidae* sp. (42.7%), *Palaemonetes* sp. (27.2%), and Arrow Shrimp (23.9%). After-opening winter impact samples show Blue Crab (40.7%), *Hippolytidae* sp. (28.1%), and Arrow Shrimp (12.2%) as the most abundant crustacean species. Spring before-opening impact site fish abundances were dominated by Pinfish (73.0%), Spot (15.7%), *Syngnathus* sp. (4.5%), and Southern Flounder (4.5%). Spring after-opening impact samples showed that *Clupeiformes* sp. (19.7%), Pigfish, (*Orthopristis chrysoptera*), (18.7%), and Pinfish (16.1%) were the most abundant fish species. Spring before-opening crustacean abundances at the impact locations were dominated by post-larval penaeids (50.7%), Arrow Shrimp (18.8%), and *Hippolytidae* sp. (15.6%). Spring after-opening impact crustacean abundances were led by post-larval penaeids (75.8%), adult *Farfantepenaeus* sp., (14.5%), and *Hippolytidae* sp. (6.6%).

Table 3: The total number of organisms collected (total catch) at control sites (see Figure 1) seasonally both before and after-opening. The relative abundance (RA) is listed seasonally for fishes and crustaceans for before and after-opening. The change in relative abundance (Δ RA %) was also calculated seasonally for each species and group of nekton by subtracting the before-opening RA (%) from after-opening RA (%). A negative change in relative abundance indicates a decline in relative abundance, while a positive number indicates in increase in relative abundance. Mean densities (m^{-2}) and mean length (mm) for control sites before and after-opening are also shown. Each mean density is calculated from a total of 24 samples collected each season. Mean lengths (total length for fish, total length for shrimp, and carapace width for crabs) were calculated from the number of individuals of a species measured during each season before and after-opening.

		Control Pre-Opening						**Control Post-Opening**						
FALL		**Total Catch**	**RA (%)**	**Mean Density (#/m^2)**	**SE**	**Mean Length (mm)**	**SE**	**Total Catch**	**RA (%)**	**Mean Density (#/m^2)**	**SE**	**Mean Length (mm)**	**SE**	**Δ RA %**
Total Fish		**566**						**911**						
Atlantic Croaker	*Micropogonias undulatus*	33	5.8	0.138	(0.062)	12.42	(0.305)	0	0.0	0.000	(0.000)	0.00	(0.000)	-5.8
Bay Whiff	*Citharichthys spilopterus*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.1	0.004	(0.004)	22.80	(0.000)	0.1
Blackcheek Tonguefish	*Symphurus plagiusa*	1	0.2	0.004	(0.004)	42.40	(0.000)	9	1.0	0.038	(0.016)	18.47	(1.296)	0.8
Darter Goby	*Gobionellus boleosoma*	188	33.2	0.783	(0.222)	18.31	(0.475)	467	51.3	1.946	(0.410)	19.12	(0.379)	18.0
Dwarf Seahorse	*Hippocampus zosterae*	3	0.5	0.013	(0.007)	18.57	(0.524)	2	0.2	0.008	(0.006)	16.95	(0.650)	-0.3
Frillfin Goby	*Bathygobius soporator*	0	0.0	0.000	(0.000)	0.00	(0.000)	3	0.3	0.013	(0.009)	27.33	(8.206)	0.3
Gobiidae	*Gobiidae sp.*	0	0.0	0.000	(0.000)	0.00	(0.000)	2	0.2	0.008	(0.008)	5.15	(0.150)	0.2
Gobiosoma sp.	*Gobiosoma sp.*	78	13.8	0.325	(0.085)	15.27	(0.518)	66	7.2	0.275	(0.071)	17.24	(0.631)	-6.5
Inland Silverside	*Menidia beryllina*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.1	0.004	(0.004)	27.00	(0.000)	0.1
Lined Seahorse	*Hippocampus erectus*	1	0.2	0.004	(0.004)	26.00	(0.000)	0	0.0	0.000	(0.000)	0.00	(0.000)	-0.2
Pinfish	*Lagodon rhomboides*	9	1.6	0.038	(0.025)	19.43	(7.056)	12	1.3	0.050	(0.018)	20.33	(11.068)	-0.3
Pipefish	*Syngnathus sp.*	110	19.4	0.458	(0.080)	42.37	(1.940)	246	27.0	1.025	(0.170)	44.76	(2.016)	7.6
Red Drum	*Sciaenops ocellatus*	130	23.0	0.542	(0.123)	10.55	(0.279)	93	10.2	0.388	(0.070)	10.54	(0.359)	-12.8
Sheepshead Minnow	*Cyprinodon variegatus*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.1	0.004	(0.004)	11.30	(0.000)	0.1
Spot Croaker	*Leiostomus xanthurus*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.1	0.004	(0.004)	9.40	(0.000)	0.1
Spotfin Mojarra	*Eucinostomus argenteus*	12	2.1	0.050	(0.034)	13.17	(0.550)	5	0.5	0.021	(0.013)	15.64	(1.435)	-1.6
Spotted Seatrout	*Cynoscion nebulosus*	1	0.2	0.004	(0.004)	60.10	(0.000)	1	0.1	0.004	(0.004)	57.60	(0.000)	-0.1
Striped Blenny	*Chasmodes bosquianus*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.1	0.004	(0.006)	42.20	(12.500)	0.1

Total Crustaceans		**22066**						**16992**						
Arrow Shrimp	*Tozeuma carolinense*	4048	18.3	16.867	(3.554)	26.30	(0.211)	2207	13.0	9.196	(2.792)	25.83	(0.376)	-5.4
Blue Crab	*Callinectus sapidus*	67	0.3	0.279	(0.077)	6.48	(0.515)	118	0.7	0.492	(0.115)	8.78	(0.679)	0.4
Brown / Pink Shrimp	*Farfantepenaeus sp.*	113	0.5	0.471	(0.106)	34.42	(1.209)	512	3.0	2.133	(0.424)	30.12	(0.466)	2.5
Grass Shrimp	*Palaemonetes sp.*	2717	12.3	11.321	(2.608)	16.61	(0.218)	6135	36.1	25.563	(6.242)	17.35	(0.224)	23.8
Mud Crab	*Xanthidae sp.*	9	0.0	0.038	(0.019)	3.43	(0.624)	32	0.2	0.133	(0.046)	6.11	(0.759)	0.1
Post-Larval Penaeids	*Penaeidae sp.*	51	0.2	0.213	(0.044)	11.66	(0.514)	434	2.6	1.808	(0.340)	12.83	(0.292)	2.3
Porcelain Crab	*Porcellanidae sp.*	0	0.0	0.000	(0.000)	0.00	(0.000)	2	0.0	0.008	(0.006)	4.90	(2.300)	0.0
Snapping Shrimp	*Alpheus heterochaelis*	11	0.0	0.046	(0.023)	12.53	(0.692)	3	0.0	0.013	(0.009)	15.13	(4.698)	0.0
White Shrimp	*Litopenaeus setiferus*	9	0.0	0.038	(0.029)	36.60	(3.456)	14	0.1	0.058	(0.050)	31.89	(1.988)	0.0
Unidentified *Hippolytidae*	*Hippolytidae sp.*	15041	68.2	62.671	(12.993)	11.01	(0.156)	7535	44.3	31.396	(4.022)	12.08	(0.200)	-23.8
	WINTER													
Total Fish		**1211**						**1153**						
Atlantic Croaker	*Micropogonias undulatus*	15	1.2	0.063	(0.028)	21.97	(0.858)	1	0.1	0.004	(0.004)	36.50	(0.000)	-1.2
Bay Whiff	*Citharichthys spilopterus*	1	0.1	0.004	(0.004)	11.90	(0.000)	12	1.0	0.050	(0.023)	12.04	(0.582)	1.0
Darter Goby	*Gobionellus boleosoma*	193	15.9	0.804	(0.273)	13.08	(0.460)	422	36.6	1.758	(0.422)	13.81	(0.344)	20.7
Dwarf Seahorse	*Hippocampus zosterae*	20	1.7	0.083	(0.050)	19.11	(0.719)	33	2.9	0.138	(0.038)	21.95	(0.573)	1.2
Gobiosoma sp.	*Gobiosoma sp.*	19	1.6	0.079	(0.019)	16.57	(0.713)	31	2.7	0.129	(0.036)	17.54	(0.857)	1.1
Gulf Flounder	*Paralichthys albigutta*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.1	0.004	(0.004)	31.90	(0.000)	0.1
Gulf Killifish	*Fundulus grandis*	0	0.0	0.000	(0.000)	0.00	(0.000)	5	0.4	0.021	(0.010)	21.18	(1.177)	0.4
Gulf Menhaden	*Brevoortia patronus*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.1	0.004	(0.004)	23.00	(0.000)	0.1
Inland Silverside	*Menidia beryllina*	1	0.1	0.004	(0.004)	34.70	(0.000)	3	0.3	0.013	(0.013)	43.50	(1.952)	0.2
Longnose Killifish	*Fundulus similis*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.1	0.004	(0.004)	45.80	(0.000)	0.1
Pinfish	*Lagodon rhomboides*	350	28.9	1.458	(0.300)	14.99	(0.135)	188	16.3	0.783	(0.148)	15.99	(0.249)	-12.6
Pipefish	*Syngnathus sp.*	37	3.1	0.154	(0.023)	69.89	(4.222)	69	6.0	0.288	(0.064)	80.82	(3.140)	2.9
Sheepshead Minnow	*Cyprinodon variegatus*	0	0.0	0.000	(0.000)	0.00	(0.000)	102	8.8	0.425	(0.119)	26.14	(0.534)	8.8
Shrimp Eel	*Ophichthus gomesii*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.1	0.004	(0.004)	62.10	(0.000)	0.1
Southern Flounder	*Paralichthys lethostigma*	28	2.3	0.117	(0.041)	13.20	(0.774)	24	2.1	0.100	(0.031)	20.04	(5.830)	-0.2
Spot Croaker	*Leiostomus xanthurus*	544	44.9	2.267	(0.473)	16.38	(0.097)	259	22.5	1.079	(0.336)	15.50	(0.186)	-22.5
Striped Mullet	*Mugil cephalus*	3	0.2	0.013	(0.009)	23.27	(0.145)	0	0.0	0.000	(0.000)	0.00	(0.000)	-0.2
Total Crustaceans		**24296**						**23974**						
Arrow Shrimp	*Tozeuma carolinense*	2068	8.5	8.617	(2.437)	30.85	(0.254)	2535	10.6	10.563	(2.514)	31.48	(0.261)	2.1
Blue Crab	*Callinectus sapidus*	350	1.4	1.458	(0.367)	7.13	(0.362)	497	2.1	2.071	(0.636)	5.74	(0.287)	0.6
Brown / Pink Shrimp	*Farfantepenaeus sp.*	4	0.0	0.017	(0.010)	33.40	(7.971)	24	0.1	0.100	(0.037)	26.69	(0.962)	0.1
Grass Shrimp	*Palaemonetes sp.*	12278	50.5	51.158	(11.070)	17.35	(0.180)	14180	59.1	59.083	(9.558)	19.82	(0.208)	8.6
Longnose Spider Crab	*Libinia dubia*	5	0.0	0.021	(0.010)	17.08	(4.665)	6	0.0	0.025	(0.009)	33.14	(5.999)	0.0
Mud Crab	*Xanthidae sp.*	61	0.3	0.254	(0.136)	7.23	(0.413)	5	0.0	0.021	(0.017)	5.12	(0.945)	-0.2
Post-Larval Penaeids	*Penaeidae sp.*	85	0.3	0.354	(0.100)	12.06	(0.147)	117	0.5	0.488	(0.130)	19.17	(0.803)	0.1
Snapping Shrimp	*Alpheus heterochaelis*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.0	0.004	(0.004)	18.50	(0.000)	0.0
Unidentified *Hippolytidae*	*Hippolytidae sp.*	9445	38.9	39.354	(8.009)	11.95	(0.156)	6609	27.6	27.538	(5.999)	13.00	(0.147)	-11.3

SPRING														
Total Fish		**1009**						**1782**						
Atlantic Croaker	*Micropogonias undulatus*	1	0.1	0.004	(0.004)	28.20	(0.000)	0	0.0	0.000	(0.000)	0.00	(0.000)	-0.1
Bay Whiff	*Citharichthys spilopterus*	18	1.8	0.075	(0.018)	14.55	(0.959)	12	0.7	0.050	(0.019)	21.42	(1.492)	-1.1
Black Drum	*Pogonias cromis*	0	0.0	0.000	(0.000)	0.00	(0.000)	2	0.1	0.008	(0.006)	4.40	(0.300)	0.1
Blackcheek Tonguefish	*Symphurus plagiusa*	0	0.0	0.000	(0.000)	0.00	(0.000)	0	0.0	0.000	(0.000)	0.00	(0.000)	0.0
Darter Goby	*Gobionellus boleosoma*	319	31.6	1.329	(0.313)	12.06	(0.232)	393	22.1	1.638	(0.500)	14.10	(0.421)	-9.6
Dwarf Seahorse	*Hippocampus zosterae*	17	1.7	0.071	(0.033)	25.99	(0.874)	32	1.8	0.133	(0.036)	16.74	(1.700)	0.1
Gobiosoma sp.	*Gobiosoma sp.*	13	1.3	0.054	(0.017)	19.16	(0.897)	50	2.8	0.208	(0.153)	14.33	(1.226)	1.5
Gulf Menhaden	*Brevoortia patronus*	3	0.3	0.013	(0.009)	7.93	(0.953)	7	0.4	0.029	(0.022)	16.26	(1.801)	0.1
Inland Silverside	*Menidia beryllina*	0	0.0	0.000	(0.000)	0.00	(0.000)	18	1.0	0.075	(0.050)	9.54	(0.293)	1.0
Inshore Lizardfish	*Synodus foetens*	0	0.0	0.000	(0.000)	0.00	(0.000)	2	0.1	0.008	(0.006)	47.75	(4.750)	0.1
Pigfish	*Orthopristis chrysoptera*	0	0.0	0.000	(0.000)	0.00	(0.000)	73	4.1	0.304	(0.102)	13.01	(0.291)	4.1
Pinfish	*Lagodon rhomboides*	559	55.4	2.329	(0.593)	14.82	(0.174)	1013	56.8	4.221	(1.049)	16.54	(0.205)	1.4
Pipefish	*Syngnathus sp.*	37	3.7	0.154	(0.035)	60.26	(4.924)	161	9.0	0.671	(0.130)	40.30	(2.478)	5.4
Silver Perch	*Bairdiella chrysoura*	0	0.0	0.000	(0.000)	0.00	(0.000)	11	0.6	0.046	(0.020)	4.41	(0.290)	0.6
Southern Flounder	*Paralichthys lethostigma*	3	0.3	0.013	(0.013)	16.20	(4.557)	0	0.0	0.000	(0.000)	0.00	(0.000)	-0.3
Spot Croaker	*Leiostomus xanthurus*	39	3.9	0.163	(0.040)	23.61	(1.290)	6	0.3	0.025	(0.014)	15.55	(0.969)	-3.5
Star Drum	*Stellifer lanceolatus*	0	0.0	0.000	(0.000)	0.00	(0.000)	2	0.1	0.008	(0.006)	4.80	(1.600)	0.1
Total Crustaceans		**6593**						**13599**						
Arrow Shrimp	*Tozeuma carolinense*	1124	17.0	4.683	(1.431)	30.61	(0.243)	1402	10.3	5.842	(1.374)	30.98	(0.313)	-6.7
Blue Crab	*Callinectus sapidus*	100	1.5	0.417	(0.094)	7.65	(0.473)	103	0.8	0.429	(0.089)	7.45	(0.525)	-0.8
Brown / Pink Shrimp	*Farfantepenaeus sp.*	24	0.4	0.100	(0.058)	24.53	(1.682)	246	1.8	1.025	(0.280)	30.21	(0.961)	1.4
Grass Shrimp	*Palaemonetes sp.*	1559	23.6	6.496	(1.641)	19.24	(0.182)	4100	30.1	17.083	(3.779)	21.65	(0.240)	6.5
Longnose Spider Crab	*Libinia dubia*	3	0.0	0.013	(0.009)	12.77	(2.074)	2	0.0	0.008	(0.006)	50.95	(10.050)	0.0
Mud Crab	*Xanthidae sp.*	28	0.4	0.117	(0.031)	5.59	(0.601)	116	0.9	0.483	(0.132)	4.86	(0.438)	0.4
Post-Larval Penaeids	*Penaeidae sp.*	1516	23.0	6.317	(1.250)	13.29	(0.124)	1936	14.2	8.067	(2.167)	14.91	(0.256)	-8.8
Unidentified *Hippolytidae*	*Hippolytidae sp.*	2239	34.0	9.329	(3.508)	12.61	(0.190)	5694	41.9	23.725	(4.368)	11.47	(0.241)	7.9

Table 4: The total number of organisms collected (total catch) at impact sites (see Figure 1) seasonally both before and after-opening. The relative abundance (RA) is listed seasonally for fishes and crustaceans for before and after-opening. The change in relative abundance (Δ RA %) was also calculated seasonally for each species and group of nekton by subtracting the before-opening RA (%) from after-opening RA (%). A negative change in relative abundance indicates a decline in relative abundance, while a positive number indicates in increase in relative abundance. Mean densities (m^{-2}) and mean length (mm) for control sites before and after-opening are also shown. Each mean density is calculated from a total of 24 samples collected each season. Mean lengths (total length for fish, total length for shrimp, and carapace width for crabs) were calculated from the number of individuals of a species measured during each season before and after-opening.

		Impact Pre-Opening						**Impact Post-Opening**						
FALL		**Total Catch**	**RA (%)**	**Mean Density (#/m^2)**	**SE**	**Mean Length (mm)**	**SE**	**Total Catch**	**RA (%)**	**Mean Density (#/m^2)**	**SE**	**Mean Length (mm)**	**SE**	**Δ RA %**
Total Fish		**99**						**715**						
Atlantic Croaker	*Micropogonias undulatus*	2	2.0	0.008	(0.008)	9.35	(1.250)	24	3.4	0.100	(0.042)	10.55	(0.554)	1.3
Bay Whiff	*Citharichthys spilopterus*	0	0.0	0.000	(0.000)	0.00	(0.000)	9	1.3	0.038	(0.016)	20.32	(2.854)	1.3
Blackcheek Tonguefish	*Symphurus plagiusa*	0	0.0	0.000	(0.000)	0.00	(0.000)	28	3.9	0.117	(0.045)	25.44	(1.823)	3.9
Blackwing Searobin	*Prionotus rubio*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.1	0.004	(0.004)	23.80	(0.000)	0.1
Darter Goby	*Gobionellus boleosoma*	0	0.0	0.000	(0.000)	0.00	(0.000)	71	9.9	0.296	(0.073)	14.58	(0.759)	9.9
Dwarf Seahorse	*Hippocampus zosterae*	0	0.0	0.000	(0.000)	0.00	(0.000)	3	0.4	0.013	(0.007)	11.03	(2.446)	0.4
Feather Blenny	*Hypsoblennius hentz*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.1	0.004	(0.004)	16.20	(0.000)	0.1
Gobiosoma sp.	*Gobiosoma sp.*	73	73.7	0.304	(0.083)	14.84	(0.564)	313	43.8	1.304	(0.440)	15.94	(0.308)	-30.0
Pinfish	*Lagodon rhomboides*	0	0.0	0.000	(0.000)	0.00	(0.000)	22	3.1	0.092	(0.037)	9.98	(0.253)	3.1
Pipefish	*Syngnathus sp.*	23	23.2	0.096	(0.019)	41.37	(4.811)	134	18.7	0.558	(0.106)	49.36	(2.432)	-4.5
Red Drum	*Sciaenops ocellatus*	0	0.0	0.000	(0.000)	0.00	(0.000)	106	14.8	0.442	(0.102)	9.74	(0.444)	14.8
Sharptail Goby	*Gobionellus oceanicus*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.1	0.004	(0.004)	12.50	(0.000)	0.1
Spotted Seatrout	*Cynoscion nebulosus*	1	1.0	0.004	(0.004)	13.30	(0.000)	2	0.3	0.008	(0.006)	58.70	(35.500)	-0.7
Total Crustaceans		**905**						**6963**						
Arrow Shrimp	*Tozeuma carolinense*	356	39.3	1.483	(0.468)	24.70	(0.390)	593	8.5	2.471	(0.657)	22.50	(0.357)	-30.8
Blue Crab	*Callinectus sapidus*	5	0.6	0.021	(0.008)	9.20	(2.770)	176	2.5	0.733	(0.160)	4.46	(0.296)	2.0
Brown / Pink Shrimp	*Farfantepenaeus sp.*	16	1.8	0.067	(0.020)	34.05	(2.416)	309	4.4	1.288	(0.349)	27.24	(0.752)	2.7
Grass Shrimp	*Palaemonetes sp.*	116	12.8	0.483	(0.200)	16.47	(0.245)	1476	21.2	6.150	(1.853)	14.24	(0.245)	8.4
Mud Crab	*Xanthidae sp.*	0	0.0	0.000	(0.000)	0.00	(0.000)	14	0.2	0.058	(0.046)	2.51	(0.220)	0.2
Post-larval Penaeids	*Penaeidae sp.*	34	3.8	0.142	(0.047)	9.82	(0.386)	1058	15.2	4.408	(0.831)	12.39	(0.230)	11.4
White Shrimp	*Litopenaeus setiferus*	9	1.0	0.038	(0.019)	41.41	(2.111)	5	0.1	0.021	(0.017)	32.28	(2.112)	-0.9
Unidentified *Hippolytidae*	*Hippolytidae sp.*	369	40.8	1.538	(0.315)	8.83	(0.177)	3332	47.9	13.883	(2.246)	9.28	(0.173)	7.1

WINTER														
Total Fish		**106**						**166**						
Atlantic Croaker	*Micropogonias undulatus*	5	4.7	0.021	(0.008)	20.08	(4.538)	39	23.5	0.163	(0.047)	21.86	(0.949)	18.8
Bay Whiff	*Citharichthys spilopterus*	0	0.0	0.000	(0.000)	0.00	(0.000)	15	9.0	0.063	(0.015)	13.11	(0.477)	9.0
Blackcheek Tonguefish	*Symphurus plagiusa*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.6	0.004	(0.004)	25.90	(0.000)	0.6
Darter Goby	*Gobionellus boleosoma*	0	0.0	0.000	(0.000)	0.00	(0.000)	11	6.6	0.046	(0.021)	15.50	(1.871)	6.6
Gobiosoma sp.	*Gobiosoma sp.*	4	3.8	0.017	(0.012)	18.38	(1.357)	16	9.6	0.067	(0.028)	21.11	(1.060)	5.9
Gulf Menhaden	*Brevoortia patronus*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.6	0.004	(0.004)	16.20	(0.000)	0.6
Pinfish	*Lagodon rhomboides*	10	9.4	0.042	(0.012)	14.81	(0.583)	20	12.0	0.083	(0.028)	16.71	(1.121)	2.6
Pipefish	*Syngnathus sp.*	8	7.5	0.033	(0.016)	61.01	(2.614)	8	4.8	0.033	(0.012)	64.81	(4.168)	-2.7
Sheepshead Minnow	*Cyprinodon variegatus*	20	18.9	0.083	(0.052)	28.29	(1.379)	0	0.0	0.000	(0.000)	0.00	(0.000)	-18.9
Shrimp Eel	*Ophichthus gomesii*	1	0.9	0.004	(0.004)	47.80	(0.000)	0	0.0	0.000	(0.000)	0.00	(0.000)	-0.9
Southern Flounder	*Paralichthys lethostigma*	13	12.3	0.054	(0.019)	17.95	(1.780)	14	8.4	0.058	(0.015)	17.51	(1.844)	-3.8
Spot Croaker	*Leiostomus xanthurus*	45	42.5	0.188	(0.051)	14.81	(0.342)	41	24.7	0.171	(0.044)	14.87	(0.456)	-17.8
Total Crustaceans		**239**						**656**						
Arrow Shrimp	*Tozeuma carolinense*	57	23.8	0.238	(0.079)	27.34	(0.487)	80	12.2	0.333	(0.094)	26.20	(0.335)	-11.7
Blue Crab	*Callinectus sapidus*	11	4.6	0.046	(0.021)	8.66	(0.937)	267	40.7	1.113	(0.149)	4.09	(0.202)	36.1
Brown / Pink Shrimp	*Farfantepenaeus sp.*	0	0.0	0.000	(0.000)	0.00	(0.000)	4	0.6	0.017	(0.013)	26.23	(2.331)	0.6
Grass Shrimp	*Palaemonetes sp.*	65	27.2	0.271	(0.098)	17.46	(0.262)	35	5.3	0.146	(0.047)	17.70	(0.513)	-21.9
Mud Crab	*Xanthidae sp.*	0	0.0	0.000	(0.000)	0.00	(0.000)	7	1.1	0.029	(0.013)	8.09	(1.473)	1.1
Post-larval Penaeid	*Penaeidae sp.*	4	1.7	0.017	(0.010)	11.90	(0.699)	78	11.9	0.325	(0.089)	12.95	(0.363)	10.2
Snapping Shrimp	*Alpheus heterochaelis*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.2	0.004	(0.004)	11.70	(0.000)	0.2
Unidentified *Hippolytidae*	*Hippolytidae sp.*	102	42.7	0.425	(0.146)	10.50	(0.192)	184	28.0	0.767	(0.170)	11.68	(0.171)	-14.6
SPRING														
Total Fish		**89**						**299**						
Atlantic Croaker	*Micropogonias undulatus*	1	1.1	0.004	(0.004)	22.70	(0.000)	15	5.0	0.063	(0.046)	17.50	(1.511)	3.9
Bay Whiff	*Citharichthys spilopterus*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.3	0.004	(0.004)	11.60	(0.000)	0.3
Darter Goby	*Gobionellus boleosoma*	0	0.0	0.000	(0.000)	0.00	(0.000)	25	8.4	0.104	(0.036)	13.16	(1.565)	8.4
Dwarf Seahorse	*Hippocampus zosterae*	1	1.1	0.004	(0.004)	22.40	(0.000)	2	0.7	0.008	(0.006)	15.55	(6.550)	-0.5
Gobiosoma sp.	*Gobiosoma sp.*	0	0.0	0.000	(0.000)	0.00	(0.000)	7	2.3	0.029	(0.019)	9.09	(2.328)	2.3
Gulf Menhaden	*Brevoortia patronus*	0	0.0	0.000	(0.000)	0.00	(0.000)	6	2.0	0.025	(0.018)	21.07	(0.406)	2.0
Inland Silverside	*Menidia beryllina*	0	0.0	0.000	(0.000)	0.00	(0.000)	14	4.7	0.058	(0.030)	9.93	(2.240)	4.7
Inshore lizardfish	*Synodus foetens*	0	0.0	0.000	(0.000)	0.00	(0.000)	2	0.7	0.008	(0.006)	40.50	(0.000)	0.7
Pigfish	*Orthopristis chrysoptera*	0	0.0	0.000	(0.000)	0.00	(0.000)	56	18.7	0.233	(0.155)	10.94	(0.293)	18.7
Pinfish	*Lagodon rhomboides*	65	73.0	0.271	(0.164)	12.83	(0.088)	48	16.1	0.200	(0.048)	16.48	(0.453)	-57.0
Pipefish	*Syngnathus sp.*	4	4.5	0.017	(0.010)	26.30	(14.304)	24	8.0	0.100	(0.020)	26.15	(4.547)	3.5
Silver Perch	*Bairdiella chrysoura*	0	0.0	0.000	(0.000)	0.00	(0.000)	21	7.0	0.088	(0.041)	5.28	(0.263)	7.0
Southern Flounder	*Paralichthys lethostigma*	4	4.5	0.017	(0.010)	29.18	(2.098)	0	0.0	0.000	(0.000)	0.00	(0.000)	-4.5
Spot Croaker	*Leiostomus xanthurus*	14	15.7	0.058	(0.024)	28.76	(1.822)	18	6.0	0.075	(0.050)	17.08	(0.674)	-9.7
Spotted Seatrout	*Cynoscion nebulosus*	0	0.0	0.000	(0.000)	0.00	(0.000)	1	0.3	0.004	(0.004)	11.70	(0.000)	0.3
Unidentified *Clupeiform*	*Clupeiformes sp.*	0	0.0	0.000	(0.000)	0.00	(0.000)	59	19.7	0.246	(0.106)	9.42	(0.351)	19.7

Total Crustaceans		**154**						**2268**						
Arrow Shrimp	*Tozeuma carolinense*	29	18.8	0.121	(0.056)	32.51	(0.665)	26	1.1	0.108	(0.030)	33.00	(0.803)	-17.7
Blue Crab	*Callinectus sapidus*	15	9.7	0.063	(0.017)	9.54	(0.507)	12	0.5	0.050	(0.012)	12.18	(1.749)	-9.2
Brown / Pink Shrimp	*Farfantepenaeus sp.*	2	1.3	0.008	(0.008)	28.65	(0.850)	329	14.5	1.371	(0.388)	24.20	(0.571)	13.2
Grass Shrimp	*Palaemonetes sp.*	5	3.2	0.021	(0.012)	18.54	(1.729)	30	1.3	0.125	(0.046)	19.39	(1.085)	-1.9
Mud Crab	*Xanthidae sp.*	1	0.6	0.004	(0.004)	12.10	(0.000)	2	0.1	0.008	(0.008)	8.30	(0.800)	-0.6
Post-larval Penaeid	*Penaeidae sp.*	78	50.6	0.325	(0.086)	13.34	(0.370)	1720	75.8	7.167	(1.629)	13.52	(0.155)	25.2
Unidentified *Hippolytidae*	*Hippolytidae sp.*	24	15.6	0.100	(0.050)	10.39	(0.379)	149	6.6	0.621	(0.142)	10.05	(0.370)	-9.0

Nekton Density and Length Comparisons

Substantial changes were observed in nekton density at the impact sites after Cedar Bayou was opened. The control sites nearer to a long established tidal inlet largely remained similar throughout the study. Overall, impact sites were characterized by significant increases in total organisms, total fish, total crustaceans, and ASD mean densities following the reopening of Cedar Bayou (Figure 2, Table 5 and 6). No significant changes in total organism or total fish mean densities were detected at the control sites before and after-opening. Results for total organism density showed a significant BA x CI interaction ($df = 278$, $F = 20.628$, $p < 0.0001$) (Table 5). The one-way "post-hoc" ANOVAs demonstrated the interaction was due to significantly higher density of total organisms at the impact sites after-opening ($2.21\ m^{-2} \pm 0.346$ to $15.37\ m^{-2} \pm 2.325$, $df = 139$, $F = 93.325$, $p < 0.0001$), while there was no significant difference at control sites between years ($77.42\ m^{-2} \pm 9.227$ to $81.13\ m^{-2} \pm 6.804$, $df = 139$, $F = 3.805$, $p = 0.0531$) (Table 5 and 6, Figure 2).

When examining changes in total fish densities, a significant BA x CI interaction was observed ($df = 278$, $F = 4.624$, $p = 0.0324$). Post-hoc ANOVAs found no significant differences at the control sites after-opening ($3.87\ m^{-2} \pm 0.357$ to $5.34\ m^{-2} \pm 0.567$, $df = 139$, $F = 3.273$, $p = 0.0726$), while total fish density increased at the impact locations after-opening ($0.41\ m^{-2} \pm 0.075$ to $1.64\ m^{-2} \pm 0.251$, $df = 139$, $F = 46.137$, $p < 0.0001$) showing the positive influence of Cedar Bayou on nekton recruitment.

Similar to fish, I found a large response to reopening Cedar Bayou for crustaceans. The main effects ANOVA for total crustacean density showed a highly significant interaction ($df = 278$, $F = 17.644$, $p < 0.0001$) with the post-hoc tests showing a slightly significant change in total crustacean density at the control sites (73.55 m^{-2} ± 9.096 to 75.78 m^{-2} ± 6.632, *df* = 139, F =4.149, $p < 0.0436$). Environmental variability is common (Fogarty et al. 1991) and is most likely the cause of this small change. Changes were much more apparent and significant at the impact sites with very large increase in crustacean density after-opening (1.80 m^{-2} ± 0.322 to 13.73 m^{-2} ± 2.128, *df* = 139, F = 89.926, $p < 0.0001$). The ASD main effects ANOVA did not detect a significant BA x CI interaction; however, the post-hoc ANOVAs showed significant changes did occur at the control (70.17 m^{-2} ± 9.252 to 70.00 m^{-2} ± 6.730, *df* = 139, F = 5.238, $p < 0.0236$) and impact (1.56 m^{-2} ± 0.315 to 8.20 m^{-2} ± 1.747, *df* = 139, F = 18.683, $p < 0.0001$) sites. The mean density, standard error, and sample size used in each ANOVA are shown in Table 6.

To understand more fine-scale changes and determine the impact of Cedar Bayou to estuarine-dependent species, I assessed changes in mean density and length for several species during their peak recruitment season(s) (Table 6 and 7, Figure 3). The main effects ANOVA for Red Drum density detected a significant interaction between the BA and CI factors (*df* = 86, F = 17.275, p = 0.0001). Post-hoc analyses indicate that no significant change in Red Drum density occurred at the control sites between the before and after-opening years (0.54 m^{-2} ± 0.123 to 0.39 m^{-2} ± 0.070, *df* = 43, F = 0.850, p = 0.3617). The impact site Red Drum density was found to increase significantly (0.00 m^{-2} to 0.44 m^{-2} ± 1.102, *df* = 43, F = 36.919, $p < 0.0001$) (Table 6 and 7, Figure 3). I was unable to perform statistical analysis for Red Drum lengths because zero were captured at the impact sites before-opening. After-opening Red Drum mean length was found to be 9.7 mm (Table 6, Figure 3). Mean lengths remained similar at control sites between years (Table 6).

When examining Southern Flounder density, there was not a significant interaction observed between the BA and CI factors (*df* = 86, F = 0.138, p = 0.7112) and there was no significant change in density at either the control (0.12 m^{-2} ± 0.041to 0.10 m^{-2} ± 0.031, *df* = 43, F = 0.850, p = 0.3617) or impact locations (0.05 m^{-2} ± 0.019 to 0.06 m^{-2} ± 0.015, *df* = 43, F = 0.059, p = 0.8098). The mean length main effects ANOVA did not show a significant interaction (*df* = 69, F = 1.380, p = 0.2441). I observed no significant changes in mean length between before and after-opening sample years at the control (13.20 mm ± 0.774 to 20.04 mm ± 5.830, *df* = 47, F = 2.625, p = 0.1119) or impact locations (17.95 mm ± 1.780 to 17.51mm ± 1.844, *df* = 22, F = 0.034, p = 0.8559) (Table 6 and 7, Figure 3).

Atlantic Croaker densities had a significant interaction between the BA and CI factors based on the results of the main effects ANOVA (*df* = 86, F = 17.543, p = 0.0001). I found a significant decrease in density at the control sites (0.06 m^{-2} ± 0.028 to 0.00 m^{-2} ± 0.004, *df* = 43, F = 4.774, p = 0.0344) while impact locations showed increased density (0.02 m^{-2} ± 0.008 to 0.16 m^{-2} ± 0.047, *df* = 43, F = 12.860, p = 0.0009). Atlantic Croaker mean length did not show a significant interaction (*df* = 52, F = 1.618, p = 0.2091); however post-hoc results showed a significant change in mean length at control sites (21.97 mm ± 0.858 to 36.50 mm ± 0.00, *df* = 12, F = 8.253, p = 0.0140) with no change at impact locations (20.08 mm ± 4.538 to 21.86 mm ± 0.949, *df* = 40, F = 0.492, p = 0.4872) (Table 6 and 7, Figure 3). It should be noted that I only collected one Atlantic Croaker after-opening at the control sites, so these results should be interpreted cautiously.

Similar to estuarine-dependent fishes, post-larval penaeid density changed significantly (BA x CI interaction) with the opening of Cedar Bayou (*df* = 278, F = 18.257, p < 0.0001). Post-hoc ANOVAs show that no significant change occurred at the control locations (2.29 m^{-2} ± 0.533 to 3.45 m^{-2} ± 0.822, *df* = 139, F = 2.717, p = 0.1016), while significantly more post-larval penaeids were found at the impact sites after-opening (0.16 m^{-2} ± 0.036 to 3.97 m^{-2} ± 0.688, *df* = 139, F = 85.462, p < 0.0001). I also found a significant length interaction (*df* = 2391, F = 7.270, p = 0.0071) where control sites penaeids were significantly larger after-opening (13.12 mm ± 0.111to 14.67 mm ± 0.212, *df* = 1391, F = 12.940, p < 0.0001), but at impact sites there was no change (13.27 mm ± 0.361 to 13.00 mm ± 0.128, *df* = 1000, F = 1.866, p = 0.1722) (Table 6 and 7, Figure 3).

Blue Crab densities were significantly different after-opening Cedar Bayou (*df* = 278, F = 9.786, BA x CI p = 0.0019). Densities at control sites remained similar (0.72 m^{-2} ± 0.141 to 1.00 m^{-2} ± 0.233, *df* = 139, F = 0.929, p = 0.3368), but there were significantly more Blue Crabs at impact sites (0.04 m^{-2} ± 0.009 to 0.63 m^{-2} ± 0.089, *df* = 139, F = 54.916, p < 0.0001). Blue Crab mean lengths followed a similar pattern with a significant interaction (*df* = 1330, F = 27.899, p < 0.0001) and no significant change in mean length at the control sites (7.15 mm ± 0.259 to 6.87 mm ± 0.262, *df* = 872, F = 3.193, p = 0.0743), but decreased lengths at the impact sites (9.16 mm ± 0.589 to 4.46 mm ± 0.183, *df* = 458, F = 52.384, p < 0.0001) (Table 6 and 7, Figure 3). For more information on juvenile nekton mean density and length data, please see Appendix B.

Table 5: Analysis of variance nested model (for total organisms, fish, crustaceans, and aggregated shrimp (ASD) densities, each over all seasons) with site as a nested random factor within the control-impact (CI) factor. The main effect model is a two-way nested ANOVA ($\alpha = 0.05$) that tests for an interaction between the before-after (BA) and control-impact (CI) factors. The post-hoc ANOVA's ($\alpha = 0.05$) utilize a combined BA:CI factor with four levels to compare before-control (BC) densities to after-control (AC) densities and before-impact (BI) densities to after-impact (AI) densities. ASD represents the summed total of Arrow Shrimp, *Hippolytidae* sp., and *Palaemonetes* species. See Table 6 for mean, standard error, and sample size.

	DENSITY			
	df	*F* -value	*P* -Value	
Total Organisms				
<u>*Main Effect*</u>				
BA	278	4.615	0.0326	
CI	6	185.257	<0.0001	
BA x CI	278	20.628	<0.0001	*
<u>*Post-Hoc*</u>				
BC x AC	139	3.805	0.0531	
BI x AI	139	93.325	<0.0001	*
Total Fish				
<u>*Main Effect*</u>				
BA	278	4.633	0.0322	
CI	6	58.241	0.0003	
BA x CI	278	4.624	0.0324	*
<u>*Post-Hoc*</u>				
BC x AC	139	3.273	0.0726	
BI x AI	139	46.137	<0.0001	*
Total Crustaceans				
<u>*Main Effect*</u>				
BA	278	5.183	0.0236	
CI	6	170.034	<0.0001	
BA x CI	278	17.644	<0.0001	*
<u>*Post-Hoc*</u>				
BC x AC	139	4.149	0.0436	*
BI x AI	139	89.926	<0.0001	*
Aggregated Shrimp Density				
<u>*Main Effect*</u>				
BA	278	7.110	0.0081	
CI	6	110.147	<0.0001	
BA x CI	278	0.319	0.5726	
<u>*Post-Hoc*</u>				
BC x AC	139	5.238	0.0236	*
BI x AI	139	18.683	<0.0001	*

Table 6: The mean density, (m^{-2}), standard error, (SE), and sample size, (n), used in density analyses. Density (n) represents the number of tows taken while length (n) represents the number of individuals measured for each species. Aggregated shrimp density (ASD) was calculated by summing the total of Arrow Shrimp, *Hippolytidae* sp., and *Palaemonetes* species. Total organisms, fish, and crustaceans mean densities were calculated over all seasons while species values were calculated during their primary recruitment seasons (Atlantic Croaker = winter, Red Drum = fall, Southern Flounder = winter, post-larval penaeids = all seasons, Blue Crab = all seasons).

	CONTROL						**IMPACT**					
	Before-opening			After-opening			Before-opening			After-opening		
	Mean	SE	n	Mean	SE	n	Mean	SE	n	Mean	SE	n
DENISTY (m^{-2})												
Total Organism	77.42	9.227	72	81.13	6.805	72	2.21	0.346	72	15.37	2.326	72
Total Fish	3.87	0.357	72	5.34	0.567	72	0.41	0.075	72	1.64	0.251	72
Total Crustaceans	73.55	9.096	72	75.78	6.632	72	1.80	0.322	72	13.73	2.128	72
ASD	70.17	9.252	72	70.00	6.73	72	1.56	0.315	72	8.20	1.747	72
DENISTY (m^{-2})												
Atlantic Croaker	0.06	0.028	24	0.00	0.004	24	0.02	0.008	24	0.16	0.047	24
Red Drum	0.54	0.123	24	0.39	0.070	24	0.00	0.000	24	0.44	0.102	24
Southern Flounder	0.12	0.041	24	0.10	0.031	24	0.05	0.019	24	0.06	0.015	24
Post-Larval Penaeids	2.29	0.533	72	3.45	0.822	72	0.16	0.036	72	3.97	0.688	72
Blue Crab	0.72	0.141	72	1.00	0.233	72	0.04	0.009	72	0.63	0.089	72
LENGTH (mm)												
Atlantic Croaker	21.97	0.858	15	36.50	0.000	1	20.08	4.538	5	21.86	0.949	39
Red Drum	10.55	0.279	130	10.54	0.359	93	0.00	0.000	0	9.74	0.449	106
Southern Flounder	13.20	0.774	28	20.04	5.830	24	17.95	1.780	13	17.51	1.844	14
Post-Larval Penaeids	13.12	0.111	557	14.67	0.212	839	13.27	0.361	78	13.00	0.128	927
Blue Crab	7.15	0.259	418	6.87	0.262	459	9.16	0.589	30	4.46	0.183	433

Figure 2: Overall mean densities (m^{-2}) of total organisms, fish, crustaceans, and Aggregated Shrimp Density (ASD) in control and impact locations over all seasons before and after-opening. ASD is the combined density of Arrow Shrimp (*Tozeuma carolinense*), *Hippolytidae* sp., and *Palaemonetes* sp., which comprised over 85% of the total crustaceans sampled during the course of this study. A main-effects two-way nested ANOVA, (BA*site(CI)), was used to test each group for significance, *p<0.05, ***p<0.001 (see Table 5 and 6).

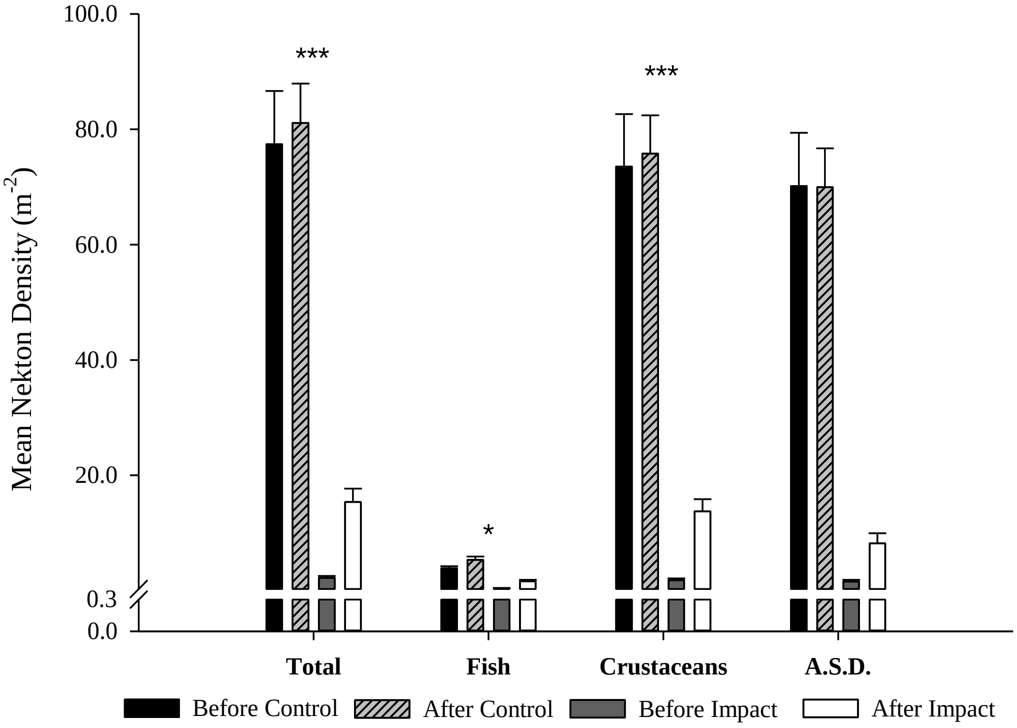

Table 7: Two-way nested ANOVA ($\alpha = 0.05$) model for individual species that tests for an interaction between the before-after (BA) and control-impact (CI) factors. The post-hoc ANOVAs ($\alpha = 0.05$) are each a one-way nested ANOVA that tests for within family differences of the CI factor at both levels of the BA variable. This series of ANOVAs was run for each species of interest for both density (m^{-2}) and length (mm) during their respective peak recruitment seasons (Red Drum -fall, Southern Flounder -winter, Atlantic Croaker - winter, post-larval penaeids - all seasons, and Blue Crab - all seasons). Differences in Red Drum lengths could not be calculated as none were sampled in the year before-opening. See Table 6 for mean, standard error, and sample size.

	DENSITY				**LENGTH**			
	df	*F* -value	*P* -Value		*df*	*F* -value	*P* -Value	
Red Drum								
Main Effect								
BA	86	1.171	0.2823		-	-	-	
CI	6	14.670	0.0087		-	-	-	
BA x CI	86	17.275	0.0001	*	-	-	-	
Post-Hoc								
BC x AC	43	0.850	0.3617		-	-	-	
BI x AI	43	36.919	<0.0001	*	-	-	-	
S. Flounder								
Main Effect								
BA	86	0.122	0.7282		69	2.872	0.0947	
CI	6	0.938	0.3702		6	4.551	0.0769	
BA x CI	86	0.138	0.7112		69	1.380	0.2441	
Post-Hoc								
BC x AC	43	0.850	0.3617		47	2.625	0.1119	
BI x AI	43	0.059	0.8098		22	0.034	0.8559	
Atlantic Croaker								
Main Effect								
BA	86	3.214	0.0765		52	3.040	0.0871	
CI	6	0.613	0.4636		5	0.477	0.5206	
BA x CI	86	17.543	0.0001	*	52	1.618	0.2091	
Post-Hoc								
BC x AC	43	4.774	0.0344	*	12	8.253	0.0140	*
BI x AI	43	12.860	0.0009	*	40	0.492	0.4872	
Post Larval Penaeids								
Main Effect								
BA	278	3.449	0.0643		2391	13.850	0.0002	
CI	6	13.019	0.0113		6	0.184	0.6832	
BA x CI	278	18.257	<0.0001	*	2391	7.270	0.0071	*
Post-Hoc								
BC x AC	139	2.717	0.1016		1391	12.940	<0.0001	*
BI x AI	139	85.462	<0.0001	*	1000	1.866	0.1722	
Blue Crab								
Main Effect								
BA	278	1.332	0.2495		1330	3.632	0.0569	
CI	6	11.757	0.0140		6	3.653	0.1045	
BA x CI	278	9.786	0.0019	*	1330	27.899	<0.0001	*
Post-Hoc								
BC x AC	139	0.929	0.3368		872	3.193	0.0743	
BI x AI	139	54.916	<0.0001	*	458	52.384	<0.0001	*

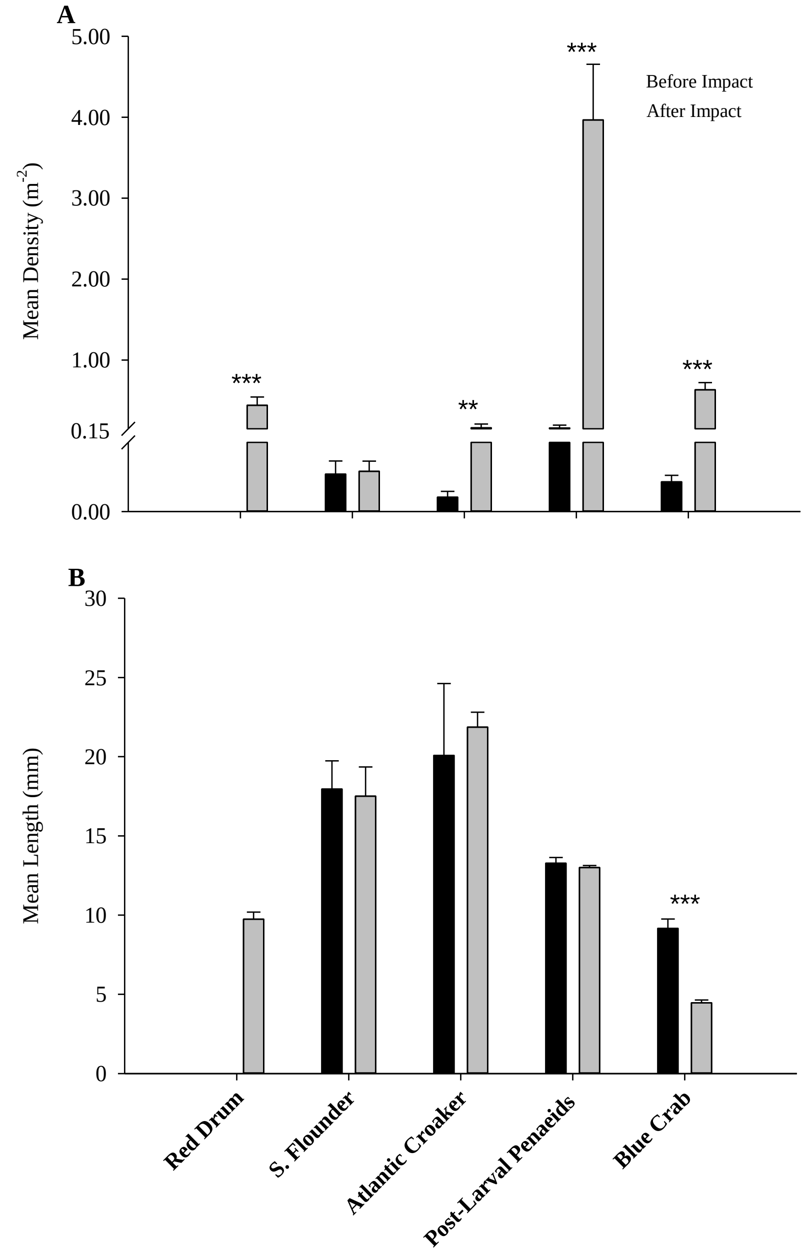

Figure 3: Mean densities (A) and mean lengths (B) of selected fish and crustacean species before and after-opening for impact locations during their respective peak recruitment season(s) (see Table 6). Mean densities and lengths were calculated from all individuals sampled within the specified recruitment season. Post-hoc ANOVA (BI*AI) results are indicated above the bars; ** $p<0.01$, ***$p<0.001$.

Community Analysis

Community analysis showed distinct changes in community assemblage at the impact locations after Cedar Bayou was reopened. PERMANOVA results indicate a moderate interaction might have occurred between the BA and CI factors at a community level (998 permutations, *df* = 1, pseudo-F = 2.196, p = 0.07). Pair-wise tests demonstrated that while the control site community did not change before and after-opening, (393 permutations, *df* = 10, t = 0.932, p = 0.461), the impact site community changed drastically (418 permutations, *df* = 10, t = 2.288, p = 0.003). Differences between the control and impact sites existed before (411 permutations, *df* = 10, t = 3.166, p = 0.002) and after opening (402 permutations, *df* = 10, t = 2.034, p = 0.003). For the PERMANOVA table see Appendix C. A closer examination of the MDS ordination provided evidence that the after-opening impact site community assemblages shifted towards the healthy control site community (Figure 4).

Given that changes to the impact site community were so great, additional analyses were performed to determine more fine-scale changes. The RELATE routine verified that no correlation existed between the abundance matrices for before and after-opening impact communities (R = 0.281, p = 0.100). Differences in before and after-opening nekton densities at the impact sites were also demonstrated with the Bray-Curtis cluster analysis and MDS ordination. The Bray-Curtis cluster revealed four significant clusters at the 63% similarity level, represented by two before-opening groups and two after-opening groups (Figure 5). The MDS ordination verified four distinct clusters with clear separation of before and after-opening samples at 63% similarity (Figure 5).

Increased densities of post-larval penaeid shrimp (estuarine-dependent), *Hippolytidae* sp., and Blue Crab (estuarine-dependent) were observed at the impact sites after opening of the inlet (Table 8). The two-way crossed SIMPER analysis showed these species contributed the most to the dissimilarity between the before and after-opening communities at the impact sites across all seasons. These same organisms contributed greatly to within group similarity (Table 8). Overall, numerous other estuarine-dependent species including Red Drum, Pinfish, and Atlantic Croaker contributed to within group similarity at the impact sites after-opening Cedar Bayou (Table 8).

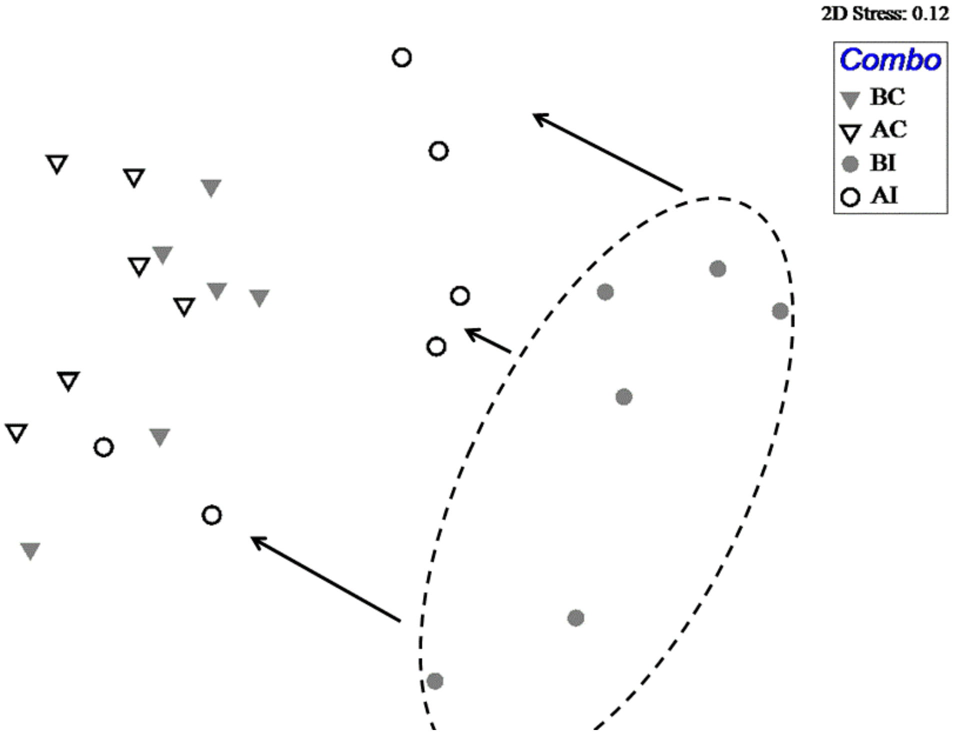

Figure 4: Multidimensional scaling (MDS) ordination of control and impact community assemblages before and after-opening (BC = before-control, AC = after-control, BI = before impact, AI = after-impact). This plot visually depicts the impact community assemblage shifting towards the healthy control site community after-opening. The contour is for visual emphasis only and does not indicate similarity percentages.

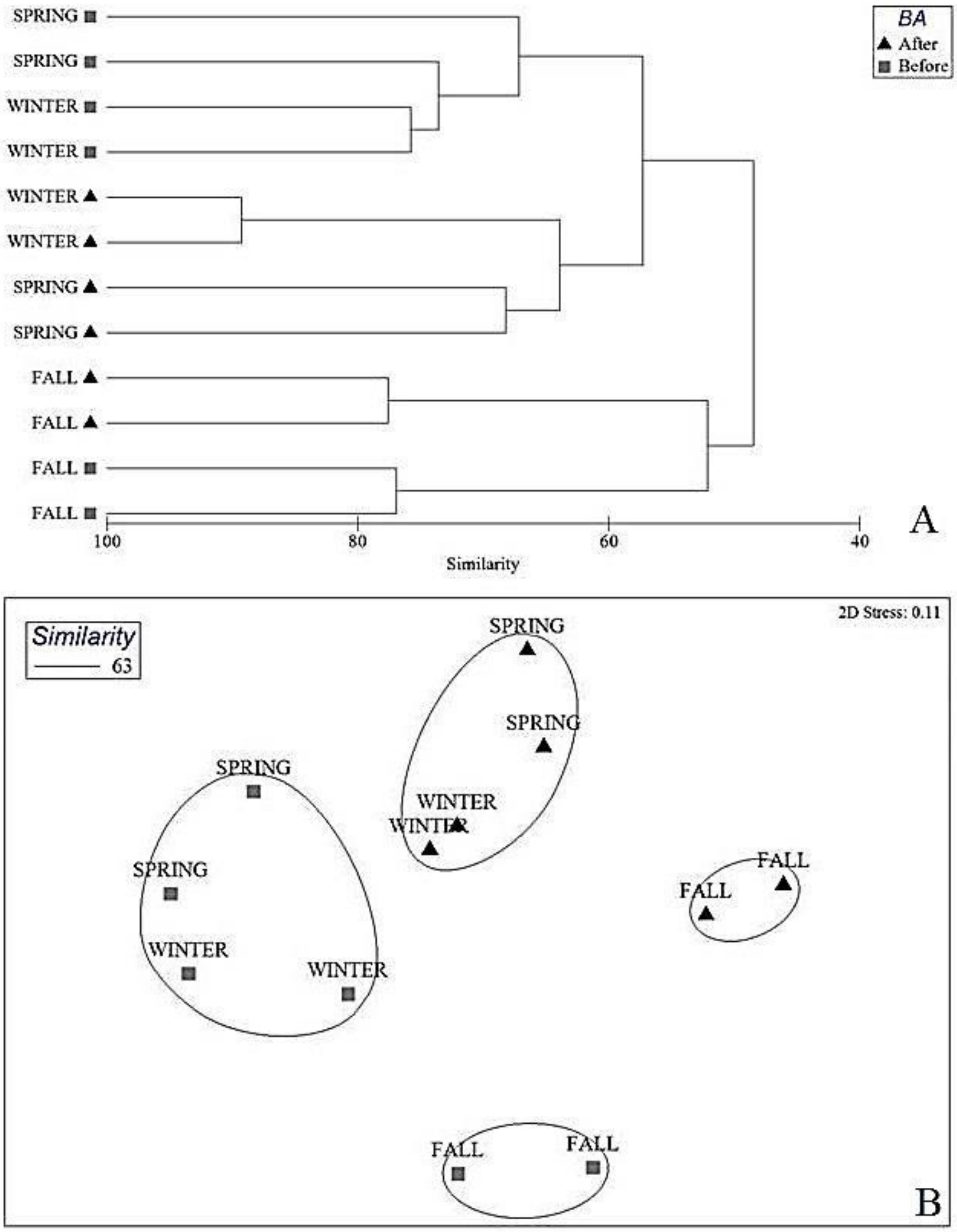

Figure 5: Bray-Curtis cluster analysis (A) and MDS ordination (B) with Bray-Curtis cluster analysis superimposed using 63% similarity of nekton density from before and after opening samples from impact sites only over all seasons. Densities were averaged for CI locations by date for a total of 12 samples.

Table 8: Two-way crossed SIMPER summaries (before and after-opening across all seasons) for impact sites showing species that contributed >1% to the between group dissimilarities. Mean densities (m^{-2}) and similarity data were calculated from impact locations over all seasons for both before and after-opening (n=144). Similarity data were fourth root transformed. A dash (-) indicates species that contributed <1% to the average similarity or dissimilarity.

		Before-opening		After-opening		Before- & After-Opening
Species		Mean Density	Similarity (%)	Mean Density	Similarity (%)	Dissimilarity (%)
Penaeidae sp.		0.16	18.77	3.97	21.39	12.88
Unidentified *Hippolytidae*		0.69	26.56	5.09	17.12	9.83
Callinectus sapidus	Blue Crab	0.04	4.64	0.63	12.05	9.41
Farfantepenaeus sp.		0.03	2.19	0.89	9.69	8.62
Palaemonetes sp.	Grass Shrimp	0.26	7.74	2.14	5.26	7.33
Tozeuma carolinense	Arrow Shrimp	0.61	17.08	0.97	8.32	7.18
Syngnathus sp.	Pipefish	0.05	5.01	0.23	4.93	5.65
Lagodon rhomboides	Pinfish	0.10	3.59	0.13	3.36	5.32
Gobionellus boleosoma	Darter Goby	0.00	0.00	0.15	2.65	4.48
Gobiosoma sp.		0.11	5.19	0.47	2.36	4.07
Micropogonias undulatus	Atlantic Croaker	0.01	0.41	0.11	2.34	3.98
Eucinostomus argenteus	Spotfin Mojarra	0.08	6.90	0.08	3.16	3.75
Sciaenops ocellatus	Red Drum	0.00	-	0.15	3.10	2.73
Paralichthys lethostigma	Southern Flounder	0.02	1.36	0.02	1.08	2.28
Citharichthys spilopterus	Bay Whiff	0.00	-	0.03	1.27	2.20
Mud Crabs (*Xanthidae*)		0.00	-	0.03	0.22	1.27
Orthopristis chrysoptera	Pigfish	0.00	-	0.08	0.33	1.20
Unidentified *Clupeiform*		0.00	-	0.08	0.25	1.19
Bairdiella chrysoura	American Silver Perch	0.00	-	0.03	0.34	1.15
Symphurus plagiusa	Blackcheek Tonguefish	0.00	-	0.04	0.45	1.08

Discussion

The goal of this study was to assess the effects of re-opening a natural tidal inlet on estuarine nekton. Dramatic changes to total nekton, fish, and crustacean densities were found to occur at the impact sites after opening, showing that the opening of Cedar Bayou significantly impacted the community in a positive manner. For example, I observed elevated recruitment and density for numerous estuarine-dependent species including Red Drum, Blue Crab, Atlantic Croaker, and post-larval penaeid shrimp. The increases in estuarine-dependent species were large enough to induce a community shift at the impact sites after-opening and demonstrated the positive effect that tidal inlets can have on nekton density and community assemblage.

Physical Parameters

Overall physical water parameters remained very similar. The reopening of tidal inlets can cause changes in physical water parameters at both a local and regional level (Reese et al. 2008; Milbrandt et al. 2012). These changes have the potential to impact nekton assemblages and community structure (Hoff and Ibara 1977; Reese et al. 2008); however, the differences in water quality parameters observed here were not necessarily attributable to opening Cedar Bayou and variance fluctuations were within a normal range that would not have influenced nekton habitat use patterns. For example, I found significant salinity changes at both the control and impact sites across numerous seasons, indicating that these changes are most likely due to environmental variability and not directly impacted by Cedar Bayou. Water temperature and salinity generally remained similar within season at the control and impact sites. Thus, the differences observed here are likely due to normal fluctuations within estuaries and are unlikely to have any biological effect.

Nekton Density and Abundance

Reopening tidal inlets is often a costly and time consuming process, making restoration efforts fairly rare. For this reason it is important to document the ecological effects of inlet restorations not only to justify the high cost, but also to justify the concept as a viable management and conservation tool. Moreover, many species that support valuable fisheries along the Texas coast are dependent on access to estuarine nursery habitat, typically seagrass beds, for their population persistence.

Exploring the relative abundance of all species sampled provided a broad perspective of how nekton responded to Cedar Bayou being reopened in relation to control areas located near a well-established tidal inlet. Differences in nekton abundance and densities at the impact sites were observed, while control sites had fewer changes and were more stable throughout study. The most abundant species at the control sites did not change after Cedar Bayou was reopened; although, there were slight shifts in their abundances. These species are very common in healthy Texas estuaries (Nelson 1992; Reese et al. 2008) and these changes are most likely attributable to normal abundance fluctuations. For example, the most abundant fish species at the winter before-opening control sites were Spot, Pinfish, and Darter Gobies, respectively. After-opening I found that Darter Gobies were the most abundant followed by Spot and Pinfish. Although the order of relative abundance changed, the same few species comprised the largest portion of individuals sampled with only minor changes in relative abundance of other abundant species at the control sites as well.

The effect of the opening Cedar Bayou was best characterized by the numerous within-season changes to nekton abundance at impact sites between the before and after-opening years. For example, during the winter before-opening season, the fish community at impact sites was dominated by Spot, Sheepshead Minnow, and Southern Flounder. After-opening the most abundant species changed to Spot, Pinfish, and Atlantic Croaker. Both Pinfish and Atlantic Croaker are estuarine-dependent species that recruit in the winter (Patillo et al. 1997; Reese et al. 2008) and were present in very low numbers prior to opening. Additionally, some species that were not documented to occur at the impact sites before-opening became very abundant after-opening. For example, the increase in Blue Crab abundance at the impact sites during the fall season suggests that opening Cedar Bayou allowed estuarine-dependent species to move more effectively from their offshore spawning grounds to inshore nursery habitat in Mesquite Bay showing that inlets play a large role in determining nekton populations and assemblages (Milbrandt et al. 2012).

Total organism density greatly increased at the impact sites after Cedar Bayou was open. In contrast, total organism density at control sites remained relatively stable through time suggesting a greater influx of estuarine-dependent species to impact locations in Mesquite Bay. An increase in fish density provides additional evidence that estuarine-dependent nekton, particularly fish, are migrating through Cedar Bayou and settling at the first seagrass beds they encounter (Bell and Westoby 1986). Reese et al. (2008) demonstrated similar patterns for fish densities; however, they observed a significant decrease in total organism abundance after the reopening of Packery Channel, a tidal inlet south of Cedar Bayou on the Texas coast. Unlike the Cedar Bayou study, opening Packery Channel caused a dramatic decrease in seagrass cover in the sampling area, likely leading to a drastic decrease in some of the most common species (Reese et al. 2008) post-opening. Few other studies have found differences in fish density after reopening a tidal inlet. For example, inlet restoration studies on Rollover Pass, TX (Reid 1957), and Blind Pass, FL (Milbrandt et al. 2012), concluded that no significant difference in fish density occurred and attributed the similar abundances to the close proximity of other inlets which already allowed for recruitment to these areas. Thus, the impact of opening tidal inlets like Cedar Bayou and Packery Channel that are more isolated may be heightened because estuarine-dependent nekton are able to recruit to nursery habitats that were previously inaccessible (Reese et al. 2008).

Total crustacean densities increased at both control and impact sites after-opening. This pattern was clearly driven by the high abundance of the ASD species which comprised over 85% of total crustaceans, while masking the response of other crustacean species that occurred in lower abundance. Thus, separate analyses were performed by separating the ASD to elucidate patterns for other species of interest. As suspected, ASD seemed to drive crustacean changes at both the control and impact locations, although the response at the impact sites was dramatically more apparent. The species represented in ASD are estuarine residents that are present in high densities in seagrass beds throughout the year and exhibit peak abundances during different seasons based on environmental conditions. These ASD species, particularly *Palaemonetes* sp., provide a critical prey base for numerous organisms including Red Drum (Wood 1967;

Morgan 1980). Seeing such a large increase in ASD shows an increase of prey in Mesquite Bay since Cedar Bayou was opened.

The impact of Cedar Bayou can be most clearly seen when examining Red Drum densities before and after-opening. Red Drum were not found at impact sites before-opening. After opening I found densities at impact sites that resembled the control sites, which remained similar over the 2 years of sampling. This clearly demonstrates significant increases during peak recruitment season(s) after the inlet was opened. These responses were very similar to Reese et al.'s (2008) finding that juvenile Red Drum have difficulty dispersing and accessing seagrass habitat that is not near an open inlet. Bushon (2006) concluded that juvenile nekton densities, including Red Drum, decrease the farther away a location is from an open inlet. Other *sciaenids* responded similarly.

Atlantic Croaker support an economically important fishery and are also a key prey species for sportfish such as Spotted Seatrout (*Cynoscion nebulosus*), Red Drum, and Southern Flounder (Bonzek et al. 2009). After opening Cedar Bayou, much higher densities of Atlantic Croaker were found at impact sites during their peak winter recruitment. Conversely, control site densities decreased very slightly over the course of this study, which is most likely attributed to annual variations in recruitment (Fogarty et al. 1991). However, our data suggest that the increased density found at the impact sites was larger than normal variation, and is a function of juveniles having access to Mesquite Bay via Cedar Bayou. Thus, the reopening of Cedar Bayou increased densities of Atlantic Croaker in Mesquite Bay which will likely contribute to supporting larger populations of the economically valuable sportfish that prey on them.

Newly-settled and juvenile Southern Flounder are generally found in very low densities throughout Texas estuaries, so our findings of no change in their density after-opening at impact sites is not surprising. Although several studies have shown that juvenile Southern Flounder densities are highest in vegetated areas nearest to open inlets (Stokes 1977; Nañez-James et al. 2009; Froeschke et al. 2013 A), it appears that the reopening of Cedar Bayou may have had little impact on juvenile Southern Flounder densities. My results are consistent with the findings of Bushon (2006), Reese et al. (2008), and Milbrandt (2012), who also observed that the reopening of tidal inlets did not cause any changes in Southern Flounder populations. Nañez-James et al. (2009) showed that young flounder have the potential to reach isolated areas even when nearby inlets are closed and their low abundance even near open tidal inlets may have prevented the detection of any significant impacts, indicating that access to tidal inlets may not be the most limiting factor on Southern Flounder recruitment. Given that Southern Flounder populations support a large recreational and commercial fishery and have been in decline for the past 25 years (Stunz et al. 2000) and show very slow recovery, more research is needed to elucidate recruitment dynamics for this species.

Post-larval penaeid shrimp are extremely important to the Texas economy and ecosystem. Penaeid shrimp are the main target of the Texas shrimping industry, which generates $159 million annually (NMFS 2014). They showed a very large increase at the impact sites after-opening, while no change was seen at the control sites suggesting that post-larval penaeid shrimp were able to access nursery habitats in Mesquite Bay from the Gulf of Mexico via Cedar Bayou. Because post-larval penaeid shrimp now have better access to seagrass beds and the resources they provide, it could translate into increased productivity and recruitment success. Although we are not able to estimate this increased productivity directly, this concept is important as penaeid shrimp are a main prey item of Spotted Seatrout, Red Drum, Atlantic Croaker, and Southern Flounder, (Minello and Zimmerman 1983; Minello et al. 1989) and are the key forage base supporting the multi-billion dollar recreational sportfish fishery within the state (NMFS 2014). Thus, any increase in their productivity has potentially large ecological and economic benefits to the Mesquite Bay region.

Blue Crab support an important commercial fishery within Texas waters (NMFS 2012), and their importance as a prey species for other organisms makes them a central component of most estuary ecosystems in North America (Scharf et al. 2000). A major increase in Blue Crab density was observed at the impact locations after the opening of Cedar Bayou, while control site density remained the same. Numerous studies have documented the ability of Blue Crab to disperse over wide areas, (Patillo et al. 1997; Bushon 2006; Reese et al. 2008) and many studies have shown that the highest densities of juvenile Blue Crab occur nearest to open tidal inlets (Heck and Thoman 1984; Heck et al. 2001; Bushon 2006). These results clearly suggest that reopening Cedar Bayou allowed juvenile blue crabs better access to Mesquite Bay habitats. Increasing Blue Crab recruitment and access to nursery habitat in Mesquite Bay is important for numerous reasons with the most important being that the endangered Whooping Crane winter survival rates correlate directly with Cedar Bayou’s Blue Crab availability (Pugesek et al. 2008). Increased Blue Crab recruitment into Mesquite Bay’s seagrass nurseries could translate into higher survival rates of juveniles, thus providing more crabs contributing to the large size classes in population. This increase would provide an increased forage base for sensitive Whooping Crane populations wintering near Cedar Bayou.

The average length of many nektonic species remained the same at the impact and control sites after flow was restored to Cedar Bayou. Previous studies that examined the effects of reopening tidal inlets demonstrated that mean size of estuarine-dependent nekton may decrease at impact sites once the inlet is reopened (Bushon 2006; Reese et al. 2008). This indicates recruitment via the new inlet rather than an extended migration from another inlet farther away (allowing for growth as they are moving). Blue Crab were smaller at the impact sites after-opening. Post-larval penaeid shrimp mean lengths also changed significantly at the control sites, although the mean difference was an increase of only ~ 1.5 mm and the biological significance of this change is questionable. Given the high growth rate of many juvenile recruits (Rooker et al. 1997) and the variability in exact recruitment dates (Rooker et al. 1998), size may not always be a good indicator of where recruitment is derived. This is especially true when numerous cohorts of recruits simultaneously occur and when averaging lengths outside of peak recruitment. Thus, I was unable to broadly detect significant changes to the size of many species in relation to Cedar Bayou’s reopening. Changes in density patterns were a better indicator of recruitment potential via the new tidal inlet, and indicated a substantial increase in juvenile nekton after the inlet was opened.

Community Analysis

Community assemblage shifted at the impact sites while the control site communities remained unchanged before and after-opening showing that the flowing inlet substantially influenced on structuring these populations. I found strong seasonal effects on community structure as seen in other studies (Reese et al. 2008) with winter/spring samples assemblage groups clustered together and fall groups being unique. It is well-known that seasonal pulses of juvenile estuarine-dependent nekton can cause community assemblages to shift and change seasonally, which is most likely the cause of seasonal groupings (Reese et al. 2008). Additionally, there was also a very distinct difference between the samples taken before and the samples taken after reopening Cedar Bayou, regardless of season, which indicates a community shift before and after-opening. The driving force behind the impact site community shift were increased numbers of juvenile post-larval penaeids, Blue Crab, and *Farfantepenaeus* species. In addition, estuarine-dependent species, such as the American Silver Perch (*Bairdiella chrysoura*) and Red Drum, that were absent in before-opening samples contributed to the similarity of the community assemblage after-opening. Other estuarine-dependent species such as post-larval penaeids, Blue Crab, and *Farfantepenaeus* species, were found in higher abundance after-opening and contributed to the within-community similarity (Reese et al. 2008). Because most of the species that were responsible for the community shift are estuarine-dependent and rely on open inlets to deliver them to seagrass nursery habitat, it is clear that reopening Cedar Bayou had a positive influence on estuarine-dependent nekton in Mesquite Bay.

In conclusion, the opening of Cedar Bayou caused positive changes in nekton density, size, and community structure in the seagrass habitats of Mesquite Bay. Evidence from this study clearly shows that reopening this natural tidal inlet provided a new means of ingress for estuarine-dependent species, which attributed to community shifts after-opening. Increases in penaeid shrimp and Blue Crab, both estuarine-dependent, were the major drivers of this change. Penaeid shrimp and Blue Crab provide a large forage base for numerous sportfish species including Red Drum, Spotted Seatrout (*Cynoscion nebulosus*), and Southern Flounder. In addition, increased Blue Crab abundance may have the potential to support the over wintering Whooping Crane population. Although size patterns were less clear, my overall findings show the potential for increased species productivity as a result of reopening Cedar Bayou. It will take years to appreciate and understand the full effect of restoring Cedar Bayou on ecologically and economically important species such as Red Drum and Blue Crab; however, the inlet’s positive influence on the Mesquite Bay ecosystem was observed almost immediately after it was reopened and we anticipate these positive effects to continue.

CHAPTER 2

HISTORICAL EFFECTS OF CEDAR BAYOU ON MESQUITE BAY'S NEKTON POPULATIONS: A MODELING APPROACH

Introduction

With the progression of sustainable fisheries management, the importance of tidal inlets to estuarine ecosystems has become an increasingly important topic. While several efforts have been made document the effects of reopening tidal inlets (Reese et al. 2008; Milbrandt et al. 2012), very few efforts have been made to document their importance in historical ecosystems. This is due mainly to the lack of comprehensive fishery-independent data. Understanding the historical significance of tidal inlets in providing access to nursery areas and essential fish habitat (EFH) is needed to set realistic goals for contemporary inlet restoration efforts.

The Cedar Bayou study is very unique in that a comprehensive, historical fishery-independent dataset exists for the Mesquite Bay region (Nañez-James 2009). The Texas Parks and Wildlife Department (TPWD) has collected extensive seine, gillnet, and physical parameter data along the Texas coast since 1978 (TPWD 2003). These data provide an abundance of fish and crustacean population information through time and allow for a direct comparison of historical organism abundance and physical conditions to the flow regime (open or closed) of Cedar Bayou. By using statistical modeling and looking at historical trends, it is possible to predict how reopening the Cedar Bayou inlet will impact certain estuarine species. Additionally, it is possible to understand what physical parameters most influence individual species populations during years of open or closed flow.

Predictive models are valuable tools that generate expected population responses of species to future events such as inlet restoration efforts. The high cost and effort of inlet restoration often prohibits projects from being successful. However, being able to predict restoration outcomes and their economic impact would undoubtedly help future restoration efforts and help identify which inlet would provide the greatest environmental and economic returns if reopened. Additionally, the accuracy of the predicted model could be tested by empirical studies to allow for further refinement and help statisticians adapt it for other inlet restorations in the future. Thus, the overall objective of this study was to build a model of estuarine species population trends using the long term TPWD dataset to determine how Cedar Bayou's flow regime impacts species density. Specifically, this study focused on:

3. Assessing long term population trends using the TPWD dataset for juvenile (bag seine) and adult (gill net) fish abundance during times of historical flow and closure of the bayou.

H_{A3}: Cedar Bayou flow regimes influence historic species CPUE.

Methods

Background

This study sought to establish general long-term abundance trends for juvenile (bag seine data) and adult (gill net data) Red Drum (*Sciaenops ocellatus*), Blue Crab (*Callinectes sapidus*), Atlantic Croaker (*Micropogonias undulatus*), Brown Shrimp (*Farfantepenaeus aztecus*), and Southern Flounder (*Paralichthys lethostigma*) in

Mesquite Bay as they related to historical environmental data and the historical flow status of Cedar Bayou. The modeled historical catch-per-unit-effort (CPUE) responses to Cedar Bayou's flow status were compared to empirical density data collected one year before and one year after-opening Cedar Bayou. Boosted Regression Trees (BRT) are an ideal tool to test these hypotheses.

Boosted Regression Tree analyses (as described in Froeschke and Drymon 2013) were used to analyze a long term data set collected by the TPWD between the late 1980 and 2012. Boosted Regression Trees are designed to improve single models by combining the processes of boosting and regression tree modeling to produce numerous models for prediction (Elith et al. 2008). Tree based models partition response variables into subsets that respond similarly to predictor variables (Elith et al. 2008; Froeschke et al. 2013 B). Boosting fits models to training data in order to minimize the loss of predicative performance of a model (Elith et al. 2008). In short, BRTs are able to accept different types of predictor variables while accommodating missing values through the use of surrogates. In addition, they resist the effects of outliers and automatically fit interactions between predictors since the response of one input variable depends on previous predictor values (Froeschke et al. 2013 B; Elith et al., 2006, 2008; Leathwick et al., 2006, 2008).

Modeling Approach

Spatially-explicit estimates of abundance of estuarine species were developed using species distribution models to estimate impacts of an additional tidal inlet on recruitment patterns of estuarine nekton. Models of predicted abundance were developed using BRT (Elith et al. 2008; Froeschke et al. 2010). Boosted Regression Trees use a model averaging (ensemble) method that allows for both explanation and prediction (Elith et al. 2008). However, BRTs have only recently been applied to ecological questions (Friedman 2001, Leathwick et al. 2006, Elith et al. 2008, Froeschke et al. 2010, Froeschke and Froeschke 2011). Given the ability of BRTs to model interactions and automatically select important variables, as well as robustness to outliers and missing data, BRT models are growing in popularity. Each individual model consists of a simple regression tree based on a series of binary splits constructed from the predictor variables (Hastie et al. 2001), accommodates continuous or categorical predictors, missing values, and is not affected by transformation or outliers. This technique can also fit complex non-linear relationships and often has superior predictive performance to other techniques such as generalized linear and additive models that are often used to develop standardized indices of abundance (Lo et al. 1992). Relative importance of explanatory variables can be estimated by averaging the number of times a variable is selected for splitting and the squared improvement resulting from these splits (Friedman 2001, Friedman and Muelman 2003). Values are scaled to 100 with higher numbers indicating a stronger influence on the response variable. The ability to model interactions is controlled by a tree complexity *(tc)* parameter where the value specifies the number of nodes on each tree and subsequently the ability to model interactions (Leathwick et al. 2006). This approach was extended using a delta-lognormal approach (Lo et al. 1992), where the proportion of positive samples and the catch rates on positive sets were combined to develop a single estimate of abundance (Cass-Calay and Schmidt 2009). This modeling approach is flexible and used frequently when considering zero-inflated data that are common in ecology (Froeschke et al. 2013 C; Drexler and Ainsworth 2013).

Specifically,

$$l_y = c_y p_y \qquad (1)$$

where c_y is the estimate of abundance for positive catches only for sample *y*, and p_y is the estimate of mean probability for sample *y*. Both c_y and p_y were estimated using BRT. For all species considered, data used to estimate abundance for positive catches *c* and probability of occurrence *p* were assumed to have a lognormal distribution and a binomial distribution, respectively, and modeled using the following equations:

$$ln(c) = \beta X + \varepsilon \qquad (2)$$

and

$$p = \frac{e^{X\beta+\varepsilon}}{1+e^{X\beta+\varepsilon}} \qquad (3)$$

respectively, where *c* is a vector of the positive catch data, *p* is a vector of the presence/absence data, *X* is the design matrix for main effects, β is the parameter vector for main effects, and E is a vector of independent normally distributed errors with expectation zero and variance σ^2 (Ingram et al 2010).

For each species, the objective was to estimate the probablilty of catching one or more nekton species of interest in a sample as well as the expected number in samples where one or more animals were caught. To estimate the probability of occurrence, a binomial BRT was fit to these data using a tree complexity of five with a learning rate equal to 0.001 to minimize residual deviance without overfitting. The positive samples were log transformed and a BRT was fit to these data using a tree complexity of five with a learning rate equal to 0.001 with a Gaussian error distribution. After the full model was fit, a model simplification routine ("gbm.simplify") was used that is similar to a backward selection routine in regression (Elith et al. 2008). Variables were removed until significant increases in residual deviance were found between the full and reduced models. This routine was employed for both the binomial and log-normal models and permits different explanatory variables to be included in each sub-model for each species. A delta-value for each positive sample was computed as the product of the probability of occurrence times un-logged CPUE value from the positive BRT model using functions from J.T. Froeschke (Drexler and Ainsworth 2013).

Model Validation

Model validation was done for both sub-models. Residuals were extracted and geo-referenced to investigate spatial patterns (i.e., locations or regions were model fits may be poor) and plotted in histograms to examine overall model fits. This approach was conducted seperately for both sub-models (binomial and log-normal). For the binomial sub-models, a reciever-operator-characteristic curve (ROC) was fit as an indicator of model performance. Values for ROC estimate the degree to which fitted values discriminate between observed presences and absences and can be interpreted as the probability that a presence for a species drawn at random will have a higher fitted probability than an absence drawn at random (Parisen and Moritz 2009). The area under the ROC curve can be integrated and interpreted as an Area Under the Curve (AUC) value that has a range range from 0.5 to 1. Using this metric, a value of 1 indicates perfect discrimination of probabilities between presence and absence samples and a value of 0.5 indicates that model discrimination is no better than random. While models with AUC values greater than 0.6 are considered useful (Parisien and Moritz 2009), values greater than 0.8 are considered very good, and above greater than 0.9 excellent (Lane et al. 2009). For the log-normal model, residuals were extracted and plotted in the same manner as the binomial model, however, only positive catches were included. A QQ-plot was also used to examine model fits.

Spatial Predictions

Spatially explicit delta-BRT models were developed by relating probability of abundance, and predicted occurrence in postive samples to environmental conditions (e.g., salinity, depth, temperature, DO). Temporal variables (i.e., month, year) of sampling date were also considered. The environmental variables were interpolated into spatial grids (i.e., rasters) by interpolating each variable per month, per year throughout the time series using ordinary kriging via the automap package in R. This provided a robust historical dataset to make model predictions to consider inlet effects. Depth, was not considered temporally dynamic, thus all samples in the study region were considered and interpolated using inverse-distance weighting. Inverse-distance weighting is accurate with a large number of samples and much more computationally efficient than ordinary kriging for large sample sizes. In aggregate, a suite of environmental conditions were developed each month and year from 1980 to 2012.The fitted models were then predicted to these grids (Elith et al. 2008). The grids from each sub-model were multiplied (i.e., delta approach) to develop spatially explicit predictions of abundance for each month, year combination in the time series. From here, the mean abundance (per cell), per month was calculated for display purposes. Finally, the distance to the nearest tidal inlet was also calculated for each sample using the cost-distance function in a GIS. This was done in three ways. First, a historical series of when Texas inlets were open and closed was developed of when Texas inlets were open and closed (G.W. Stunz, personal communication). This distance matrix was used to predict to from the fitted delta models. Next, distance matrices were calculated assuming Cedar Bayou was either open or closed. This approach was done both with and without the Cedar Bayou inlet and the predicted impact of the inlet opening was calculated by subtracting the predicted abundance of the open versus closed grids. This value was tabulated as a percent change and then plotted as a histogram for each species.

Results

The BRT analysis indicated that many historic estuarine-dependent species populations had higher CPUE values when Cedar Bayou was flowing. In addition, the models also found that the distance from an open inlet played a large role in historical estuarine-dependent nekton distribution, meaning that there are more nekton in habitats closer to open tidal inlets. Additionally, the results of the BRT models support my findings from chapter 1 of this study.

Red Drum

Analysis indicated that Red Drum CPUE was 18.5 % higher during years that Cedar Bayou was open (Figure 6). The BRT model also demonstrated that during years that Cedar Bayou was closed the distance from an open inlet influenced Red Drum populations the most (24 % relative influence (RI)). When Cedar Bayou was open, water temperature became the most influential factor on CPUE (22 % RI), with distance from an open inlet being the second most important variable (20 % RI). Regardless of whether Cedar Bayou was flowing, month, year, and salinity heavily influenced Red Drum CPUE (Figure 7).

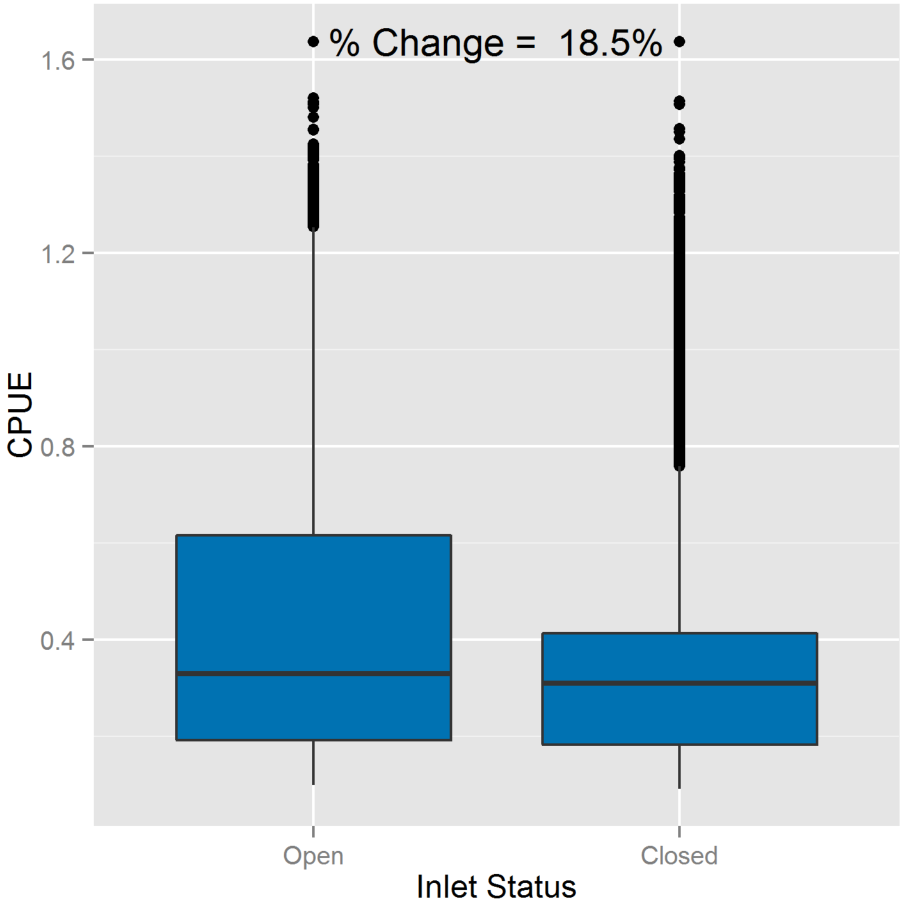

Figure 6: Predicted change of mean Red Drum catch-per-unit-effort (CPUE) based on the flow status (open or closed) of Cedar Bayou between 1978 and 2003. Black horizontal bars inside the blue box indicate the median, with the portion of the blue box above the median representing the 75th percentile and the portion below the median representing the 25th percentile. The black circles represent outliers, with the black vertical lines indicating lowest and highest observations.

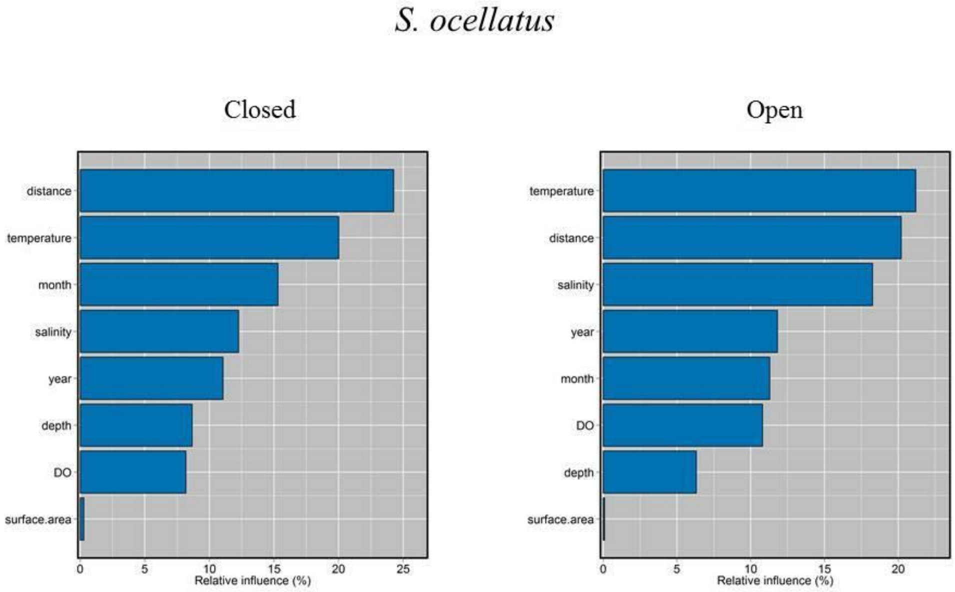

Figure 7: Physical parameters that have the highest percent relative influence on Red Drum populations during periods of open and closed flow. Distance from an open inlet was measured in km, DO represents the dissolved oxygen level (%), depth is the average depth of the bay (m), surface area is the surface area of the bay (km^2), temperature (^{o}C), salinity (ppt), and month and year are the temporal variables.

<u>Blue Crab</u>

During years that Cedar Bayou remained open, the BRT model showed that Blue Crab CPUE was 17.7% higher than in years when the inlet was closed (Figure 8). Similarly to Red Drum, distance from an open inlet was the most determining factor in Blue Crab CPUE when the inlet was closed with a RI of 31% (Figure 9). Dissolved oxygen levels became the most influential factor during years the inlet was open (29 % RI), with distance from an open inlet becoming second (24% RI). Temperature and salinity were important factors regardless of the inlet’s flow (Figure 9).

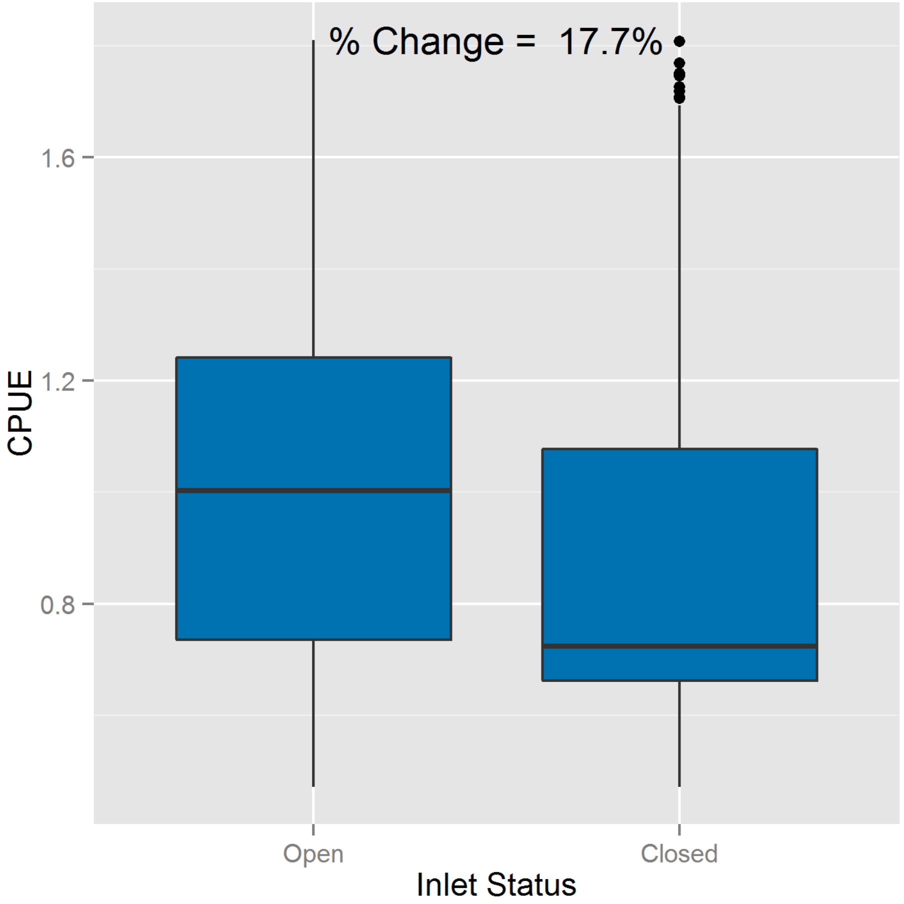

Figure 8: Predicted change of mean Blue Crab catch-per-unit-effort (CPUE) based on the flow status of Cedar Bayou between 1978 and 2003. Black horizontal bars inside the blue box indicate the median, with the portion of the blue box above the median representing the 75th percentile and the portion below the median representing the 25th percentile. The black circles represent outliers, with the black vertical lines indicating lowest and highest observations.

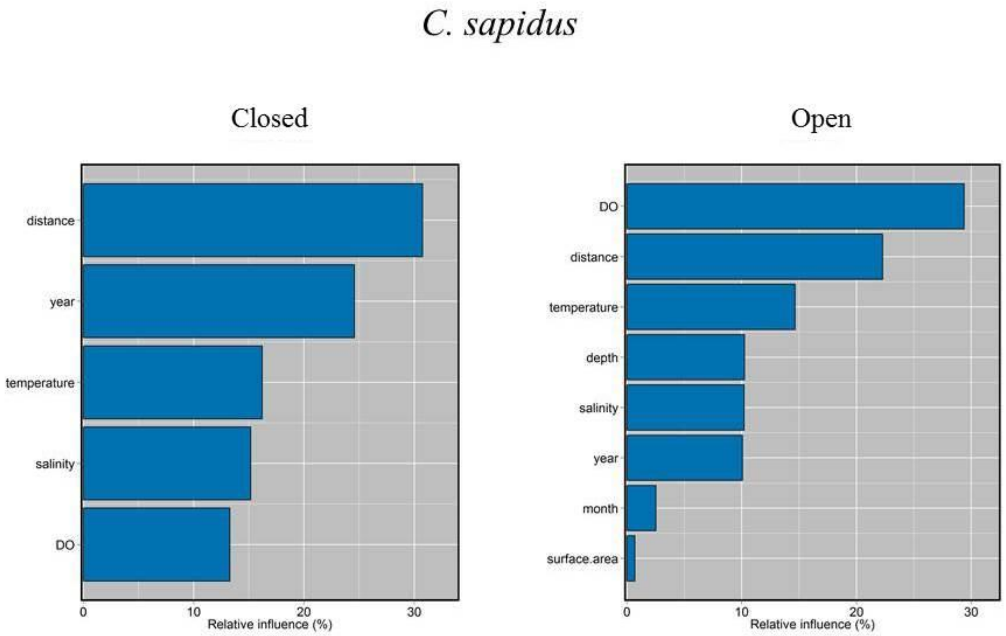

Figure 9: Physical parameters that have the highest percent relative influence on Blue Crab populations during periods of open and closed flow. Distance from an open inlet was measured in km, DO represents the dissolved oxygen level (%), depth is the average depth of the bay (m), surface area is the surface area of the bay (km^2), temperature (°C), salinity (ppt), and month and year are the temporal variables.

Species of Interest:

Other species of interest displayed less distinct relationships with Cedar Bayou's flow status. Southern Flounder CPUE was modeled to increase 19.6 % during years that Cedar Bayou was open (Appendix D). The most influential variables differed greatly between periods of closed and open flow. During years when Cedar Bayou was closed, dissolved oxygen and distance from an open inlet were the most important variables. However, when Cedar Bayou was open distance from an open inlet remained important, and temperature became the most important variable (Appendix D).

Atlantic Croaker CPUE was shown to increase 9.6 % during years that Cedar Bayou was open. Salinity and distance from an open inlet (respectively) were shown to be the most important factors effecting Atlantic Croaker CPUE, regardless of Cedar bayou flow status. Year and temperature were also important factors despite changes to the inlet's flow (Appendix D).

Brown Shrimp (*Farfantepenaeus aztecus*) CPUE was shown to decrease by 6.7 % during years that Cedar Bayou was open. Month (> 30 % RI) and year (> 10 %) were shown to be the determining factors regardless of Cedar bayou's flow status. During years when Cedar Bayou was closed, distance from an open inlet was shown to be the third most important (12 % RI) factor, while it decreased in importance (< 10 %) during years of open flow (Appendix D).

Discussion

The goal of this study was to determine if historical nekton CPUE was influenced by the flow status of Cedar Bayou using BRT analysis. The BRT modeling approach was very useful and indicated that numerous estuarine dependent species had higher CPUE values during years that Cedar Bayou was open. These results were largely supported by the empirical juvenile nekton density sampling that was conducted one year before- and after-opening. More specifically, estuarine-dependent species such as Red Drum were higher after-opening Cedar Bayou, just has historical CPUE was higher during years the inlet was open. Thus, these data show that the flow of Cedar Bayou has always played an influential role in Mesquite Bay's nekton densities.

Establishing how abiotic factors influenced historical baselines for individual species is difficult but can provide invaluable insight into how ecosystems functioned in the past. Collaborating with = statistical experts, long-term (BRT) predictive statistical modeling was used to examine fish population dynamics when Cedar Bayous was open or closed throughout its recorded history. Specifically, Boosted Regression Tree models were created for several species of interest in Mesquite Bay during years when Cedar Bayou was open and closed. These models indicated that between 1980 and 2012 the CPUE for many estuarine-dependent species, including Red Drum, Blue Crab, and Atlantic Croaker, was much higher during periods when Cedar Bayou was open. These models also indicated other abiotic factors that most influenced CPUE during periods of closure and open flow. These factors have a great potential to influence other management decisions. Field sampling conducted for the empirical density study aligned well with the BRT model results. Although total validation of the BRT models will require further field sampling, the concept of being able to predict a tidal inlet's influence on an ecosystem could provide managers with a critical tool, and may help justify the need for future inlet restorations

Boosted Regression Tree Models

This study revealed that Cedar Bayou's historic flow status had a large influence on Red Drum and Blue Crab, as there were large increases in CPUE when the inlet was open. Likewise, distance from an open inlet was a very important factor that influenced Red Drum and Blue Crab CPUE during years that Cedar Bayou was closed, and it was less important during years when Cedar Bayou was open. These relationships were expected as estuarine-dependent species, like Red Drum and Blue Crab, require open inlets to access estuary habitats (Bushon 2006; Reese et al. 2008). Furthermore, data from the empirical test one year before and one year after Cedar Bayou was dredged indicated very large increase in both Red Drum and Blue Crab densities at Mesquite Bay impact sites after Cedar Bayou was reopened in 2015. Thus, historical and contemporary data show higher CPUE/density for numerous estuarine species when Cedar Bayou is open.

Unlike Red Drum, Southern Flounder showed a poor response to years when Cedar Bayou was open, even though they have similar life history patterns. This is similar to my observed measurements of their densities after-opening, which did not differ from the before-opening samples. The dynamics of the Southern Flounder fishery in Texas, along with limited data, have long made population assessments difficult (Nañez-James 2006). Equally perplexing is that Froeschke et al. (2013B) modeled Southern Flounder CPUE along the Texas coast using a very similar BRT method and found that distance from an open inlet was the most important factor influencing juvenile Southern Flounder populations. The results of the empirical density study, when coupled with the modeling relationship of distance from an open inlet becoming more important when Cedar Bayou is open, indicate that more data are needed to better assess Southern Flounder abundance and distribution in the Aransas Bay complex. There are likely other factors, such as fishing pressure, and generally low abundance, which may be influencing recruitment of this species regardless of inlet availability (Nañez-James 2006). Understanding how these factors influence Southern Flounder recruitment is crucial to better manage this species.

Atlantic Croaker are one of the principle species in the fishing economy and sportfish food chain in Texas bays. The models indicated that CPUE increased during years that Cedar Bayou was open. Salinity and distance from an open inlet (respectively) were shown to have the most relative influence regardless of Cedar Bayou's state of flow (Appendix D). The strong influence of distance from an open inlet is supported by the findings of Levin et al. (1999), who concluded that initial larval supply to estuary systems played a substantial role in Atlantic Croaker populations. Inlets like Cedar Bayou are crucial to larval supply and thus would play a large role in Atlantic Croaker populations on a year to year basis. Surprisingly, DO was not highly ranked as an influential factor, which is dissimilar to Froeschke and Stunz (2012) findings. Their comprehensive study concluded that DO greatly impacts Atlantic Croaker behavior and distribution. This may indicate that DO is not a restrictive variable within the Mesquite Bay system, but further research on this relationship may be warranted.

The Brown Shrimp model showed a decrease in CPUE during years when Cedar Bayou was open (Appendix D). This stands in stark contrast to the large increases in *Farfantepenaeus* sp. and other post-larval penaeid shrimp observed in the juvenile density study. The modeling results may be from a gear bias as gillnets and seine nets are not optimal for collecting *Farfantepenaeus* species. Gillnets and seines used for the long term dataset are most likely avoided by adult *Farfantepenaeus* sp. and have mesh sizes too large to capture juveniles. Epibenthic sleds that were used in the nekton study have a much smaller mesh size and have shown to be the most effective and accurate way of sampling juvenile nekton in estuarine habitats (Stunz et al. 2002), including *Farfantepenaeus* sp. (Robillard et al. 2008; Baker and Minello 2011), which would explain why the density study was able to detect increases in *Farfantepenaeus* species. Future crustacean modeling approaches using TPWD long term dataset should consider potential gear bias for these species.

In most instances, BRT models based on historical data reflected the results from the density study. This indicates that the BRT models were able to accurately describe general increasing and decreasing trends for numerous estuarine dependent species based on Cedar Bayou's flow status. Gear bias in the historical dataset also needs to be considered for certain species, such as Brown Shrimp; however, other species like Southern Flounder are in need of further study to simply determine what biotic and abiotic factors influence their spawning, recruitment, and distribution before any accurate models can be developed. Overall, the BRT models were able to demonstrate that historical estuarine-dependent species had higher CPUE values during years when Cedar Bayou was open. This information is valuable for several reasons. First, it shows that Cedar Bayou (and to some degree all tidal inlets) historically played a large role in determining nekton populations within estuarine systems. This information will be useful in justifying the need for tidal inlet restoration projects in the future. Second, these models provide historical CPUE baselines for managers to pursue when restoring Cedar Bayou and other tidal inlets in the future. These baselines are critical for proper restoration and are often not available. Finally, the models themselves are valuable as a tool for establishing baselines for other projects in which historical data may be available. While the initial ability of the BRT models to predict nekton abundances looks promising, fully validating the model using multiple years of empirical data will require additional years of after-opening sampling. With continued development, these BRT models could become a very important tool for other estuary restoration projects.

CHAPTER 3

THE IMPORTANCE OF NEWLY-OPENED INLETS AS SPAWNING MIGRATION PATHWAYS FOR ADULT RED DRUM (*SCIAENOPS OCELLATUS*)

Introduction

Given that barrier islands separate most Texas estuaries and bays from the Gulf of Mexico, tidal inlets are imperative for adults of some species to migrate from estuaries to offshore spawning grounds in order to reproduce and complete their life cycles. Natural and anthropogenic factors have led to numerous inlet closures along the Texas coastline, often to the detriment to local ecosystems and tourism-based economies. Cedar Bayou is one of these areas, providing the only direct link between Mesquite Bay's seagrass nurseries and the Gulf of Mexico. The inlet was intentionally closed in 1979 to prevent contaminants from the Ixtoc I oil spill from reaching the Texas mainland.

Tidal inlets play a vital role for adult estuarine-dependent species that spend their sub-adults years in estuaries but must migrate to offshore spawning grounds once mature. The importance of tidal inlets is especially apparent in Texas estuaries given that inlets are the only passage through the barrier islands that separate the majority of estuaries from the Gulf of Mexico. Closure of any inlet along the Texas coast could increase the distance individuals must migrate to spawning grounds, which is why it is important to understand how reopening inlets may influence spawning migrations for estuarine-dependent species.

Acoustic telemetry has become an increasingly more reliable and affordable research tool for tracking large-scale movements of fish. Passive acoustic telemetry systems (PATS) have gradually become popular as they require minimal equipment maintenance while providing constant data recording capabilities (Heupel et al. 2006). Most PATS use an array of stationary receivers that detect acoustic signals from fish implanted with uniquely coded transmitters (Reese Robillard et al. 2015). By using combinations of various array layouts, coverage of multiple animals can be established to elucidate fine to landscape-scale movements with relatively few receivers. In addition, study animals do not have to be recaptured to elucidate movement data. Acoustic data can be used in a variety of ways including "presence/absence" applications, which can be adjusted to determine the timing of movement events such as a migration through Cedar Bayou (Heupel et al. 2006).

Red Drum (*Sciaenops ocellatus*) are an iconic species that have a classic estuarine-dependent life cycle. They are one of the most economically important species in the Gulf of Mexico (NMFS 2014), are very well-studied, and are an ideal species to examine estuarine-dependent species migrational use of tidal inlets. Moreover, findings from movement studies are somewhat mixed. For example, some studies indicate adult movement is limited (Collins et al. 2002; Dresser and Kneib 2007, Reyier et al. 2011), while other research found that much broader movements occur and are subject to age (Bachelor et al. 2008). Thus, there is a need to expand scientific knowledge of this species, including spawning movement patterns through inlets.

In Texas waters, larval abundance studies suggest that spawning most frequently occurs near open tidal inlets between October and January. Typically, schools of mature Red Drum form in estuaries and migrate through tidal inlets where they join adult stocks to spawn in nearshore areas, often near tidal inlets. These large migrations are a seasonal landmark along the Gulf coast and support a huge recreational fishery (Pearson 1929; Holt and Arnold 1982; Holt et. al 1985; Holt et al. 1989; Holt 2008). With a natural distribution stretching from Massachusetts to northern Mexico, Red Drum are known to persist in a variety of habitats (Hoese and Moore 1998). This has led to speculation regarding the preferred spawning locations of Red Drum, and there are numerous reports indicating spawning behavior can occur over a wide range of conditions (Carr and Smith 1977; Holt et al. 1989; Matlock 1990; Johnson and Funicelli 1991; Reyier et al. 2011). Some research has suggested that adult Red Drum display high spawning site fidelity and even natal homing in certain portions of their range (Collins et al. 2002; Patterson et al. 2004; Rooker et al. 2010). Thus, understanding if Red Drum would migrate through a restored inlet that was fully connected to the Gulf of Mexico for the first time in decades has important management implications.

Mesquite Bay is the southernmost component of the larger San Antonio Bay estuary complex (Armstrong 1987; Britton and Morton 1989). It is bordered to the north by San Antonio Bay proper and to the southwest by the Aransas Bay complex (Figure 1). The bay’s only direct connection to the Gulf of Mexico is Cedar Bayou, an ephemeral tidal inlet. Given its position on the landward side of the Matagorda and San Jose barrier islands, Mesquite Bay is very isolated from the waters of the Gulf of Mexico when Cedar Bayou is closed. However, the inlet was closed intentionally in 1979 to prevent petroleum from the Ixtoc oil spill from entering the Texas bay system. Cedar Bayou was open and flowing to the Gulf of Mexico for intermittent periods after the 1979 closure, (Bermudez et al. 2005, Ward 2010), suggesting that the inlet could have provided viable spawning conditions for episodic periods of time. However, most of the openings occurring over a very brief period and went undocumented, making it difficult to know exactly when the inlet was open or closed. Sounds associated with spawning Red Drum were detected near Cedar Bayou between 1998 and 2002, although the full extent of the spawning area was never determined (Holt 2002). It remains uncertain if that spawning aggregation was displaying natal homing and was spawning near a closed inlet, or if the inlet happened to be open and provided an opportunistic migration/spawning location. The reopening of Cedar Bayou provides a unique opportunity to determine if Red Drum adults would opportunistically utilize a newly dredged inlet for spawning migration.

There is a clear need to understand Cedar Bayou's viability as a migration route for newly matured fish to access offshore spawning grounds. Thus, the goal of this study was to determine Red Drum movement patterns in Cedar Bayou. Specifically, the objective of this study is to:

4. Use acoustic telemetry to determine if adult Red Drum are using Cedar Bayou as a migration route between Mesquite Bay and spawning grounds in the Gulf of Mexico.

H_{A4}: Fish fitted with acoustic receivers do migrate through Cedar Bayou during established spawning periods and conditions.

Methods

Study Area

The San Antonio Bay estuary complex encompasses Mesquite Bay (Armstrong 1987; Britton and Morton 1989). It is bordered to the north by San Antonio Bay proper and to the southwest by the Aransas Bay complex. Mesquite Bay is very isolated from the Gulf of Mexico when Cedar Bayou is closed, the nearest inlet being Aransas pass which is approximately 32 km to the south. The region is rich with essential fish habitats including numerous seagrass (primarily *Halodule wrightii*) beds, marshes (*Spartina alterniflora*) and oyster reefs (*Crassostrea virginica*). These habitats are essential in supporting numerous estuarine-dependent species including Red Drum.

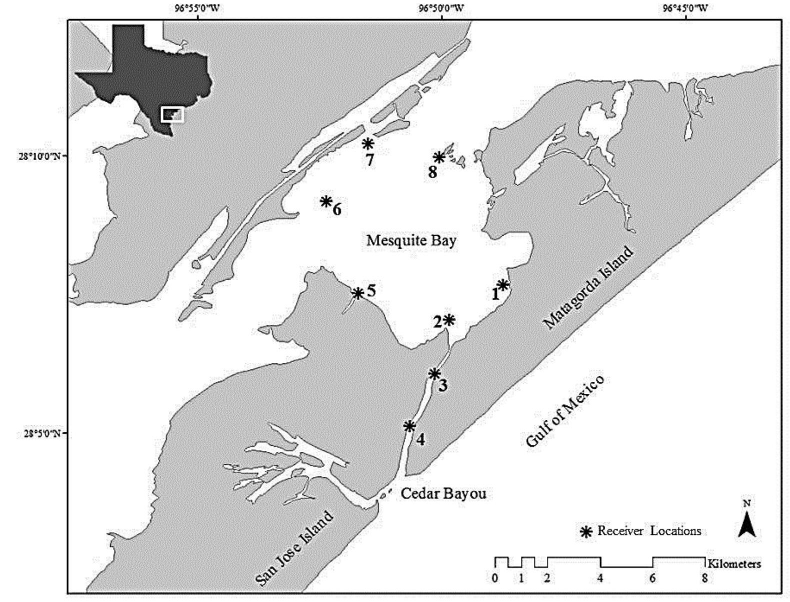

Figure 10: Vemco VR2W receiver locations within the Mesquite Bay/Cedar Bayou complex. Each receiver was placed near "bottlenecks" or near the inlet in order to track fish movements in and around Cedar Bayou. Four receivers were mounted to pilings while all other receivers utilized a removable pole mounting system.

Acoustic Equipment

Acoustic transmitters (Vemco V13-1x) were implanted in eleven Red Drum within the Mesquite Bay complex between April and July 2014, prior to opening the inlet to examine pre-opening movement patterns. To track their movement patterns an array of eight Vemco VR2W receivers was deployed in the Mesquite Bay/Cedar Bayou region in May 2014 and provided continuous coverage through June 2015. Receiver locations were specifically placed in Cedar Bayou, the perimeter of Mesquite Bay, and suspected "bottlenecks" to capture as much movement information as possible (Payne 2011) (Figure 10). The Mesquite Bay receiver array was an addition to the Texas Acoustic Array Network ("TEXAAN"), a large-scale hydrophone network that encompasses the Aransas/Corpus Christi Bay systems and the Laguna Madre, including other tidal inlets. While not specifically part of the Mesquite Bay/Cedar Bayou fine-scale movement project, the TEXAAN allowed for the potential to detect large-scale movements if they occurred. Based on other shallow water acoustic array designs, each receiver was positioned just off the bottom to ensure sound detection even during the lowest tide events (Heupel and Hueter 2001; Payne 2011; Reyier et al. 2011). I used Vemco V13-1x coded acoustic transmitters, which were similar to the Vemco V16-5H that were successfully implanted into Red Drum in a previous study (Reyier et al. 2011). Each acoustic transmitter weighed 6 g in water (11 g in air) and was 36-mm in length. The transmitters were programmed to randomly send a uniquely coded acoustic "ping" every 60 to 180 seconds at a frequency of 69.0 kHz. Randomizing ping timing minimized the probability of signal blockage when numerous tagged animals were simultaneously within the detection range a single receiver (Payne 2011). The estimated battery life of each transmitter was 658 days. Receivers in the array were cleaned and data downloaded (Vemco VUE 1.6.5) every 4 to 6 months of deployment. The detection range for each receiver is dependent on environmental conditions, such as turbidity, wave energy, water depth, and other background noise (Heupel et al. 2008). Under ideal conditions, each receiver is capable of detecting 69.0 kHz transmitters within a 1000 m radius (Payne 2011; Reyier et al. 2011). To test the actual range for receivers in Mesquite Bay, a sentinel transmitter (sending a ping every ten seconds) was deployed on a weighted, vertical mainline 0.6 m from the bottom. The sentinel transmitter was submerged for five minutes (for a total of 30 potential pings per time interval) at 0, 100, 200, 300, 400, and 500 m from three randomly selected stationary receivers (receivers 1, 5, and 8). This process was then repeated along randomly selected cardinal compass directions where practical (i.e., no land interference or shallow water) for two different transects for each receiver. These transects included signal obstructions such as oyster reef, varying depths, and sand or seagrass bottom. Thus, these estimates are likely conservative range estimates. A Vemco VR-100 hydrophone unit was used to ensure there were 30 pings from the sentinel transmitter every five minutes (Topping and Szedlmayer 2011). The receivers were then downloaded to examine detections at each distance. The number of detections for each receiver at each distance was divided by the total number of pings at the 0 m station (used as a control). The mean detection percentage of the three receivers was plotted at each distance and an overlaid sigmoidal regression line was calculated to determine detection rates at any distance (Kessel et al. 2013) (Figure 11).

Surgical Implantation of Transmitters

This study sought to elucidate tidal inlet migration behaviors of late stage sub-adult and sexually mature Red Drum during the fall spawning season. Female Red Drum in Texas reach sexual maturity by 750 mm in total length (TL), while males reach sexual maturity at smaller sizes than females (Matlock 1985 A & B; Murphy and Taylor 1990; Ross et al. 1995). The mean size of fish used in this study was 581 mm TL ± 18.6 and ranged from 521 to 708 mm TL. Thus, I assumed any Red Drum caught within the Texas slot limit (508 – 711 mm) would mature by the conclusion of this study and have the opportunity to migrate through Cedar Bayou. Additionally, previous studies have found that surgically implanted acoustic transmitters should not exceed 2% of the animal’s total body weight, (Winter 1992; Jepsen et al. 2002; Bradshaw 2006), indicating that Red Drum used in this study should be no less than 300 g or approximately 306 mm (TL) (Matlock 1985 A & B). Thus, all fish used in this study were well below the 2% body-weight recommendations for transmitter implantation.

The surgical implantation of the transmitters combined procedural elements originally developed by Reyier et al. (2011) and Reese Robillard et al. (2015). Red Drum were collected in Mesquite Bay using hook-and-line and artificial lures to minimize deep-hooking. Post collection, fish were measured (mm TL), and transferred to an oxygenated holding tank. Fish were monitored for approximately 20 minutes to ensure normal behavior post-capture. All fish were hooked in/near mouth, and no fish exhibited any type of behavior that would have been indicative of fishing mortality. Fish weight was not measured to reduce holding time and physical stress on the animal prior to surgery, but weight was determined using length-weight keys. Red Drum were not anesthetized for surgery to minimize stress and behavioral changes due to prolonged handling and anesthesia recovery time (IACUC 23-12, Texas A&M University-Corpus Christi) (Reese Robillard et al. 2015). Additionally, Rose et al.'s (2014) recent review of fish pain pathways and procedures suggested that fish do experience any additional pain or discomfort when anesthesia is not used. Furthermore, there are no FDA approved anesthetics for use on sportfish that are consumed by anglers that do not require extended withdrawal periods. Next, fish were placed dorso-ventrally in a cradle that allowed the gills to remain in oxygenated seawater (Reese Robillard et al. 2015). Using a scalpel, scales were removed from the incision site and an approximate 25-mm incision was made parallel to the ventral midline approximately 3 cm anterior to the anus (Reyier et al. 2011). Transmitters were disinfected using a 12.9% solution of benzalkonium chloride and rinsed in sterile water prior to insertion into the peritoneal cavity (Mulcahy 2003; Reese Robillard et al. 2015). The incision was closed using two Vicryl absorbable sutures held with a surgeon knot (Vicryl, 2-0 reverse cutting PSL, Ethicon, Inc., Somerville, New Jersey) (Cooke et al 2004). Distinctly numbered dart tags were inserted lateral of the dorsal fin for individual identification as well as a phone number for recapture information (Reyier et al. 2011; Reese Robillard et al. 2015). Post-surgery, fish were monitored for 20 minutes. All fish behaved normally post-surgery, and were released immediately at the location of capture. All fish swam away robustly and their movements were monitored from May 17, 2014 – June 6, 2015 for this study, and will continue to be monitored for the entirety of the estimated 658 day transmitter life (until August 2016).

Residency Index Calculation

Residency indices were calculated by dividing the number of days detected by the number of days at liberty. A residency index value of 1.00 indicates that an individual was detected at least once every day of the study. Chin et al. (2013) recommended that two actual detections were made within a 24 hour period before being classified as a "day detected." Given that some of the fish in this study moved through the Cedar Bayou array in less than 24 hours, only one detection within a 24 period was used to qualify as a day detected in this study.

Results

Range Testing

As expected, mean detection rate in the Mesquite Bay acoustic array decreased as distance from the receiver increased (Figure 11). Based on a 100% detection rate at 0m, the mean detection percentage was 72.2% ± 9.2 at 100 m, 42.5% ± 16.4 at 200 m, 10.7% ± 6.5 at 300 m, 12.3% ± 7.6 at 400 m, 2.6% ± 1.7 at 500 m. Range tests in the Mesquite Bay array indicate a 50% detection rate at approximately 170 m. The furthest distance that detections were recorded was 500 m. These detection ranges are typical for estuarine systems (Payne 2011), and each fish was detected an average of 1525 times.

Residency Index

Eight of eleven fish were detected in the array. The mean number of detections per fish was 1525 ± 930, although the number of detections per fish ranged from 0 to

10,228. Each fish that we detected showed a moderate level of movement, with fish moving among 3 receivers on average throughout the course of the study (Table 9). Red Drum activity varied greatly between individuals. Fish 1 was found to have the highest residency index (0.474) while fishes 4, 7, and 11 were never detected after they were released (Table 9). Receiver 1 recorded the highest number of detections (16,052 detections) while receivers 7 and 8 recorded no detections (Figure 12, Table 10). Receivers 2, 3, and 4 (located in Cedar Bayou) only provided 4.19 % (703 detections) of the total detections (Table 10).

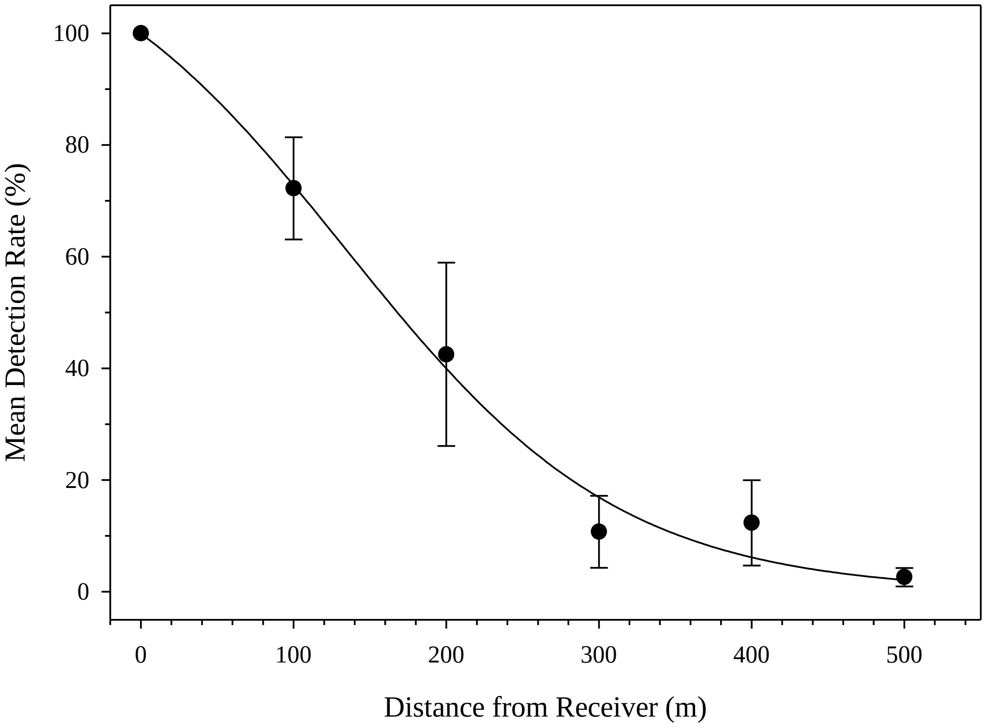

Figure 11: Mean detection rate (%) for three receivers in the Mesquite Bay array. A sentinel transmitter was moved at 100 m intervals out to 500 m from the receiver. The transmitter was submerged for 5 minutes at each interval and transmitted a ping every ten seconds (for a total of 30 pings at each distance interval). This process was conducted twice for each of the three receivers. A sigmoidal curve was fit to the data indicating a 50% detection rate at 170 m (Adj. $R^2 = 0.9819$).

Table 9: Summary data for the Mesquite Bay Red Drum acoustic study. Total detections represent the total number of times an individual fish was detected over the array. The stations visited column represents the number of individual stations that an individual was detected by. Days at liberty were the number of days between an individual's release and the last data download, which was conducted on June 6, 2015. The number of days detected represents the number of individual days a fish was detected by any receiver in the array. Residency index is simply the number of days detected divided by the number of days at liberty. A residency index value of 1.00 indicates that an individual was detected at least once every day of the study.

Fish #	**Release Date**	**TL (mm)**	**Total Detections**	**Stations Visited**	**Days at Liberty**	**# Days Detected**	**Residency Index**
1	05/17/14	551	1724	5	386	183	0.474
2	05/17/14	526	10228	4	386	175	0.453
3	05/17/14	611	69	2	386	6	0.016
4	06/11/14	521	0	0	361	0	0.000
5	06/11/14	525	27	3	361	2	0.006
6	06/11/14	525	7	4	361	6	0.017
7	07/23/14	656	0	0	319	0	0.000
8	07/23/14	708	72	2	319	5	0.016
9	08/15/14	618	3408	3	296	71	0.240
10	08/15/14	565	1243	5	296	33	0.111
11	08/15/14	580	0	0	296	0	0.000
	Mean =	**581**	**1525**	**3**	**342**	**44**	**0.121**

Table 10: Total number of detections and percentage of total detections for each receiver between May 17, 2014 and June 6, 2015. Receivers 2, 3, and 4 (located in Cedar Bayou) only made up 4.19 % of the total detections.

Receiver	**Detections**	**% Total**
1	16052	95.67
2	96	0.57
3	129	0.77
4	478	2.85
5	17	0.10
6	6	0.04
7	0	0.00
8	0	0.00
Total	**16778**	**100**

Cedar Bayou Detections

Only one detection of Red Drum occurred in Cedar Bayou prior to it being opened on September 25, 2014. After-opening, Cedar Bayou receivers (receivers 2, 3, and 4) made numerous detections during a short period of time between the fall and winter of 2014-15 (Figure 12), suggesting that these fish were more actively using the inlet post-opening. Of the 703 total detections made within Cedar Bayou, 95.6 % were made between October 18, 2014 and November 28, 2014, representing a total of five individual fish (Figure 12 and 13). Four of these five fish (fish 5, 6, 9, and 10) were last detected by receiver 4, the receiver located closest to the Gulf of Mexico (approximately 200 m north of the Gulf, Figure 10). Detections from the Cedar Bayou receivers were plotted against surface water temperature data for Mesquite Bay between October 1, 2014 and December 1, 2014 (Figure 13). Water temperature data were obtained from the Mission Aransas National Estuarine Research Reserve's (MANERR) remote data station in Mesquite Bay (accessed via the Texas Coastal Ocean Observation Network, Conrad Blucher Institute) and mean surface water temperature was found to be 18.1 ^{o}C between October 18, 2014 and November 28, 2014 (Figure 13).

Detections on other receivers throughout Mesquite Bay varied. Fish presence near Receiver 1 was consistent throughout the year (Figure 12). Receiver 5 was visited for short periods by fish 1, 6, and 10 in December and January. Receiver 6 made only six detections during the study period, which occurred on August 28, 2014 by fish 3(Figure 12). Receivers 7 and 8 did not make any detections throughout the study period. Review of these receiver units proved that they were working as multiple pings were recorded.

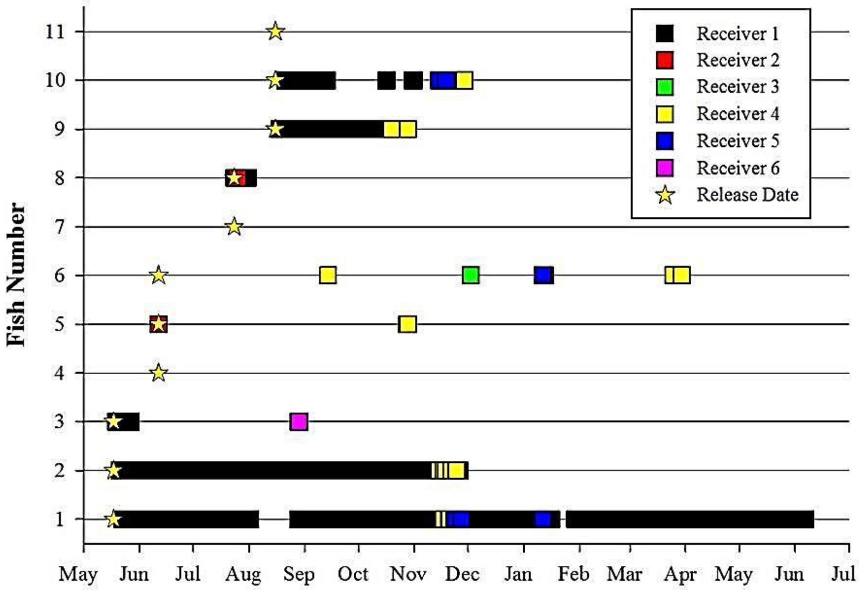

Figure 12: Total acoustic detections for each fish over the entire study period (May 19, 2014 through June 6, 2015). The majority of detections for the Cedar Bayou receivers (receivers 2, 3, and 4) seemed to occur within a relatively short period of time during fall 2015. Receivers 7 and 8 did not record any detections during this study and are thus not represented on this graph.

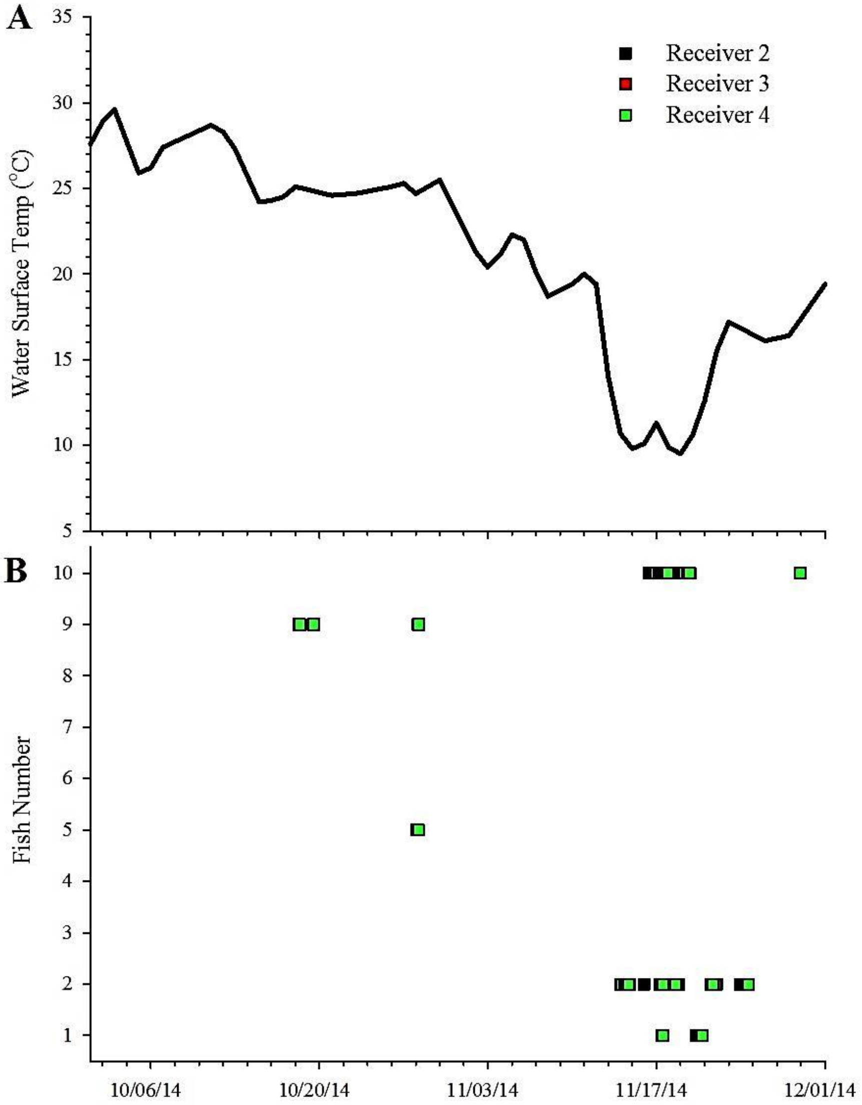

Figure 13: Water surface temperature (°C) (A) and acoustic detections within Cedar Bayou (B) (stations 2, 3, and 4 only) between October 1, 2014 - December 1, 2014. Water surface temperature data were obtained from the Mission Aransas National Estuarine Research Reserve's (MANERR) remote data station in Mesquite Bay (accessed via the Texas Coastal Ocean Observation Network, Conrad Blucher Institute). Of the 703 total detections in Cedar Bayou, 672 detections (96%) were recorded between October 18 and November 28, 2014. The water surface temperature averaged 18.1 °C between October 18, 2014 and November 28, 2014.

Discussion

The goal of this acoustic study was to examine how adult fish might use a recently opened tidal inlet. Red Drum, an idealized estuarine-dependent species, is well known for migrating from estuarine nurseries upon reaching sexual maturity in to join adult stocks. Specifically, my hypothesis was that mature Red Drum would migrate through Cedar Bayou during established spawning periods after the inlet was opened, and we used acoustic tracking test this hypothesis. A large portion of detections made in Cedar Bayou occurred during months that are well established as the Red Drum spawning and migration season. Most detections within Cedar Bayou occurred immediately following a drop in water surface temperature, which is known to elicit spawning behavior for Red Drum. Thus, given that extensive use of Cedar Bayou only occurred during the established spawning season post-opening, and that high abundances of early juvenile Red Drum were found in Mesquite Bay (see chapter 1), I concluded that Red Drum used Cedar Bayou as a migration and spawning corridor.

Survival and Detection of Red Drum Post-Surgery

Acoustic transmitter implantation has become a very viable method for obtaining spatial data more effectively. These data, along with other studies, indicate that Red Drum handle the surgical process very well (Reyier et al 2011; Moulton *in review*). Of the eleven Red Drum that were implanted in the Mesquite Bay study seven fish were recorded throughout the study. An additional fish was also detected for the first two weeks after being implanted, indicating that it survived the initial effects of the surgery.

These results show that Red Drum survive acoustic implantations and are a good model estuarine-dependent species to use to test inlet biodynamics.

Range Testing & Residency Index

The receivers had a reasonable detection range capable of identifying 50% of transmissions at 170 m. Coverage was established at key points along Mesquite Bay's perimeter and allowed for almost complete coverage across the width of Cedar Bayou. Most importantly, I was able to consistently detect eight of eleven fish which exhibited high residencies within the Mesquite Bay system and displayed spawning migration-like movements in Cedar Bayou post-opening. These data indicate that adult Red Drum most likely use reopened inlets for spawning migrations which may increase recruitment.

There have been numerous attempts to quantify Red Drum movement patterns with varying results. The results of this study were very similar to those of Reyier et al. (2011) who found that adult Red Drum in Mosquito Lagoon, Florida, remained relatively stationary to where they were captured and released during the summer months. In the Mesquite Bay study, seven of the eleven fish were captured and released nearest to receiver 1. This receiver recorded the most detections and the most individual fish throughout the study period. Two of these fish were found to have a residency index of over 0.450, indicating high site fidelity. Given the relatively small detection range of the receivers in Mesquite Bay, residency indexes are likely conservative. In addition, the overall mean residency index of 0.121 is lower than expected due to the three fish that were never detected. Given the large recreational fishery for Red Drum in this area, it is possible that these three fish did not naturally leave the study area but were removed from the system by anglers before they could be detected. These results indicate that adult Red Drum in Mesquite Bay remain relatively stationary over large periods of time, particularly during the summer months.

Impact of Cedar Bayou on Spawning Migration

Understanding how opening Cedar Bayou influenced Red Drum spawning migration patterns in the Mesquite Bay complex is important to ensuring stable recruitment in the region. Red Drum remained fairly close to the areas they were captured and released during the summer months and were only detected in Cedar Bayou once prior to October 2014. However, detections in Cedar Bayou increased during the well-established spawning months of October and November (Holt 2002; Patterson et al. 2004; Rooker et al. 2010). These pulses of activity coincide with punctuated decreases in surface water temperature that are established spawning triggers for this species, as the aquaculture industry has used decreasing light and temperatures to initiate Red Drum spawning in captivity (McEachron et al. 1993).

Four of the five fish detected in Cedar Bayou during the spawning season were last detected nearest to the Gulf of Mexico. Many Red Drum will permanently join the offshore population after their first spawning event, so if these fish did spawn it is possible that they remained offshore thus explaining why no detections were made after the spawning season. Reyier et al. (2011) found that the greatest number of detections and movement out of the study area occurred in September and October. This movement out of the study area was attributed in part to the Red Drum migrating offshore to spawn. Reyier et al.'s (2011) results are similar to the findings here. Thus, my results strongly suggest that Red Drum are using Cedar Bayou as a migration route to offshore spawning grounds.

Evaluating Cedar Bayou's value as a migration pathway from estuaries to offshore spawning grounds was also important when determining the impact of the dredging effort. For this reason I conducted a separate, but integral, acoustic telemetry study that examined adult Red Drum within the Mesquite Bay – Cedar Bayou complex. Only one implanted fish was detected in Cedar Bayou prior to reopening. Even after dredging was complete, most Red Drum remained near the locations in Mesquite Bay where they were originally implanted and released. It was only during the well-established spawning months of October and November that a defined pulse of activity was detected in Cedar Bayou. This activity also occurred directly following decreases in seasonal fall water temperature patterns - a factor that is known to trigger Red Drum spawning behavior. Likewise, the majority of study fish using Cedar Bayou during the spawning season were last detected nearest the Gulf of Mexico, suggesting a spawning migration to permanently join adult stocks offshore and explaining why these fish were never detected in the array again. Collectively, this evidence strongly suggests that Cedar Bayou provided a spawning migration route for adult Red Drum less than a month after it was reopened. This escapement of larger fish into the spawning populations could potentially increase recruitment and increase the overall population of Red Drum in the Mesquite Bay complex.

BROADER IMPACTS & CONCLUSION

Reopening tidal inlets is often a costly and time consuming process, making restoration efforts fairly rare. For this reason it is important to document the ecological effects of inlet restorations not only to justify the high cost, but also to justify the concept as a viable management and conservation tool. Moreover, recreational and commercial fisheries are major economic drivers on the Texas coast. Many species that support these fisheries are dependent on access to estuarine nursery habitat, typically seagrass beds (*Halodule wrightii*), for their population persistence. These species include juvenile Red Drum (*Sciaenops ocellatus*), Atlantic Croaker (*Micropogonias undulatus*), Southern Flounder (*Paralichthys lethostigma*), Spot Croaker (*Leiostomus xanthurus*), Blue Crab (*Callinectes sapidus*), and Penaeid shrimp species (e.g., *Farfantepenaeus aztecus*, *F. duorarum*, and *Litopenaeus setiferus*). Given that barrier islands separate most Texas estuaries and bays from the Gulf of Mexico, tidal inlets are imperative for transporting juvenile nekton (fish, shrimp, and crabs) from offshore spawning grounds into estuary nurseries where they can develop. Inlets also allow adults of the same species to migrate from estuaries to offshore spawning grounds in order to reproduce and complete their life cycles. Natural and anthropogenic factors have led to numerous inlet closures along the Texas coastline, often to the detriment to local ecosystems and tourism-based economies. Cedar Bayou is one of these areas, providing the only direct link between Mesquite Bay’s seagrass nurseries and the Gulf of Mexico. The inlet was intentionally closed in 1979 to prevent contaminants from the Ixtoc I oil spill from reaching the Texas mainland. Despite several small dredging attempts the inlet experienced drastically reduced flow and even closure until the 2014 dredging operation funded by Aransas County, Texas and the Coastal Conservation Association –Texas. Thus, the goals of this study were to: 1). Quantify what changes occurred to juvenile nekton densities in Mesquite Bay after Cedar Bayou was reopened 2). Document changes in nekton communities as a forage base, 3). Define how historical nekton populations were influenced by Cedar Bayou's flow status, and 4). Determine if adult Red Drum used Cedar Bayou as a migration route to access the Gulf of Mexico.

Nekton Density and Community Structure

I collected juvenile nekton samples from seagrass beds (*Halodule wrightii*) at control and impact sites one year before and one year after Cedar Bayou was opened. The control sites were considered healthy and far enough away from Cedar Bayou to not be influenced by the reopening. The impact sites were located in Mesquite Bay near the inlet. All organisms collected were identified to the lowest taxon and assessed for total length. I found that total organism, total fish, and total crustacean densities increased significantly in Mesquite Bay after Cedar Bayou was opened, while remaining largely the same at the control sites. Individual species responded much the same way. Red Drum, Atlantic Croaker, Blue Crab, and post-larval penaeid shrimp all increased significantly in density at the impact sites, while control site densities for all of these species, (except Atlantic Croaker which actually decreased at the control sites), remained the same. Based on these results, it is clear that reopening Cedar Bayou increased juvenile nekton densities in Mesquite Bay.

The community structure at the impact sites was also found to change dramatically after-opening. Increases in penaeid shrimp and Blue Crab, both estuarine-dependent, were the major drivers of this change. Penaeid shrimp and Blue Crab provide a large forage base for numerous sportfish species including Red Drum, Spotted Seatrout (*Cynoscion nebulosus*), and Southern Flounder. In addition, increased Blue Crab abundance may have the potential to support the over wintering Whooping Crane population. Given this evidence, I have concluded that opening Cedar Bayou has led to more available prey species, and there is potential for increased productivity in Mesquite Bay as a result of reopening Cedar Bayou.

Historical Effects on Nekton Populations

Establishing how abiotic factors influenced historical baselines for individual species is difficult but can provide invaluable insight into how ecosystems functioned in the past. Collaborating with statistical experts, long-term predictive statistical modeling was used to examine fish population dynamics when Cedar Bayou was either open or closed throughout its recorded history. Specifically, Boosted Regression Tree (BRT) models were created for several species of interest in Mesquite Bay during years when Cedar Bayou was open or closed. These models indicated that between 1980 and 2012 the catch-per-unit-effort (CPUE) for many estuarine-dependent species including Red Drum, Blue Crab, and Atlantic Croaker, was much higher during periods when Cedar Bayou was open. These models also indicate the other abiotic factors that most influenced historical CPUE. Understanding these factors has a great potential to influence management decisions. Field sampling conducted for the empirical density study aligned with the BRT model results. Although total validation of the BRT models will require further field sampling, the concept of being able to predict a tidal inlet's influence on an ecosystem could provide managers with a critical tool, and may help justify the need for future inlet restorations.

Cedar Bayou as a Spawning Migration Pathway

Evaluating Cedar Bayou's value as a migration pathway from estuaries to offshore spawning grounds was also important when determining the impact of the dredging effort. For this reason I conducted a separate, but integral, acoustic telemetry study that examined adult Red Drum within the Mesquite Bay – Cedar Bayou complex. Eleven adult Red Drum were implanted with uniquely coded acoustic transmitters which were detected using a stationary acoustic receiver system for approximately one year (four months before-opening, 8 months after-opening). Only one implanted fish was detected in Cedar Bayou prior to reopening. Even after dredging was complete, most Red Drum remained near the locations in Mesquite Bay where they were originally implanted and released. It was only during the well-established spawning months of October and November that a defined pulse of activity was detected in Cedar Bayou. This activity also occurred directly following decreases in seasonal fall water temperature patterns - a factor that is known to trigger Red Drum spawning behavior. Likewise, the majority of study fish using Cedar Bayou during the spawning season were last detected nearest the Gulf of Mexico, suggesting a spawning migration to permanently join adult stocks offshore and explaining why these fish were never detected in the array again. Collectively, this evidence strongly suggests that Cedar

Bayou provided a spawning migration route for adult Red Drum less than a month after it was reopened. In addition, this escapement of larger fish into the spawning populations could potentially increase recruitment and increase the overall population of Red Drum in the Mesquite Bay complex.

Summary

These three related studies show that opening Cedar Bayou and reconnecting Mesquite Bay to the Gulf of Mexico led to an increased presence of numerous species that are vital to Texas' ecology and economy. The inlet is providing robust connectivity between productive estuary areas and the open Gulf of Mexico and should reinstitute natural processes vital to the ecological stability of the Aransas, Mesquite, and San Antonio Bay regions.

Future Studies:

The results show very positive benefits of opening Cedar Bayou. As with most studies, many questions were uncovered and need be answered in future studies. Certainly, an additional year of juvenile nekton sampling would help solidify if changes to nekton relative abundance, mean density, and mean length will be consistent through time as nekton populations are inherently variable. These data would also help determine if the community has continued changing at the impact sites or if it has stabilized. Moreover, the acoustic transmitters and the acoustic array will continue to be monitored and maintained for the full 658 day transmitter life (August 2016). Having an additional year of acoustic data will allow for annual movement comparisons and may shed additional light on spawning movements of Red Drum through Cedar Bayou.

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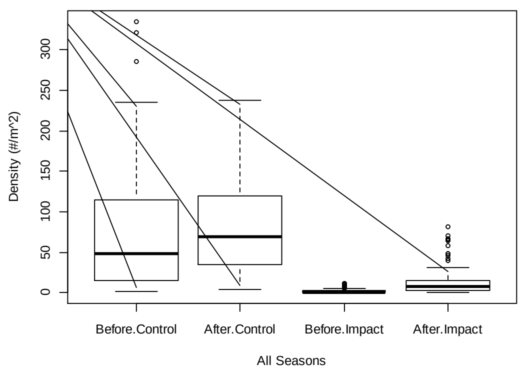
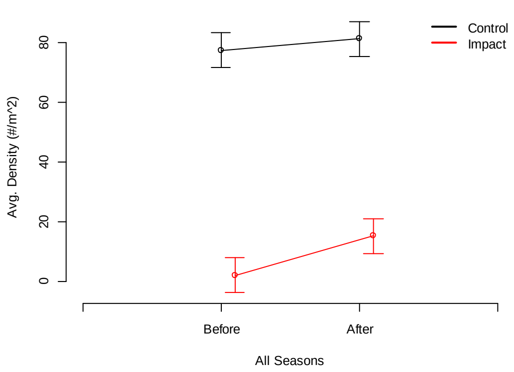
APPENDIX A

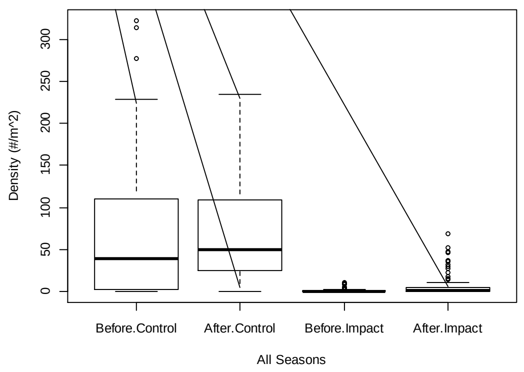
	CONTROL					IMPACT				
	DF	**SS**	**MS**	**F**	**P**	**DF**	**SS**	**MS**	**F**	**P**
FALL										
Water Temperature (^{o}C)										
Between Groups	1	5.581	5.581	0.24	0.632	1	21.669	21.669	1.07	0.318
Residual	14	325.221	23.23			14	283.399	20.243		
Total	15	330.802				15	305.068			
Salinty										
Between Groups	1	18.598	18.598	6.893	0.02	1	19.981	19.981	8.093	0.013
Residual	14	37.771	2.698			14	34.566	2.469		
Total	15	56.369				15	54.547			
Dissolved Oxygen (mg L^{-1})										
Between Groups	1	7.77	7.77	2.154	0.164	1	7.812	7.812	12.613	0.003
Residual	14	50.501	3.607			14	8.671	0.619		
Total	15	58.271				15	16.483			
WINTER										
Water Temperature (^{o}C)										
Between Groups	1	16.504	16.504	0.957	0.344	1	0.888	0.888	0.0612	0.808
Residual	14	241.352	17.239			14	203.052	14.504		
Total	15	257.856				15	203.94			
Salinty										
Between Groups	1	4.774	4.774	7.586	0.016	1	3.322	3.322	6.668	0.022
Residual	14	8.811	0.629			14	6.974	0.498		
Total	15	13.585				15	10.295			
Dissolved Oxygen (mg L^{-1})										
Between Groups	1	4.548	4.548	2.215	0.159	1	0.0049	0.0049	0.00771	0.931
Residual	14	28.743	2.053			14	8.903	0.636		
Total	15	33.291				15	8.908			
SPRING										
Water Temperature (^{o}C)										
Between Groups	1	4.951	4.951	2.679	0.124	1	145.866	145.866	52.376	<0.001
Residual	14	25.871	1.848			14	38.99	2.785		
Total	15	30.821				15	184.856			
Salinty										
Between Groups	1	1.626	1.626	1.199	0.292	1	6.015	6.015	4.691	0.048
Residual	14	18.979	1.356			14	17.952	1.282		
Total	15	20.605				15	23.966			
Dissolved Oxygen (mg L^{-1})										
Between Groups	1	59.136	59.136	52.158	<0.001	1	0.077	0.077	0.722	0.41
Residual	14	15.873	1.134			14	1.493	0.107		
Total	15	75.009				15	1.57			

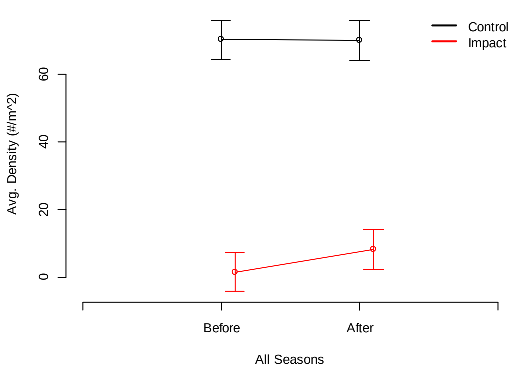
Appendix A: Physical water parameter ANOVA table. ANOVA information for mean physical parameters (water temperature (^{o}C), dissolved oxygen (mg L^{-1}), and salinity) for control and impact sites before-opening (October 2013 – April 2014) and after-opening (October 2014 – April 2015). An one-way ANOVA ($\alpha = 0.05$) was conducted to determine changes at the control and impact sites before and after reopening. See Table 2 for mean values and standard error (SE).

APPENDIX B


Total Organism Density


Avg. Total Organism Density


Agg. Shrimp Density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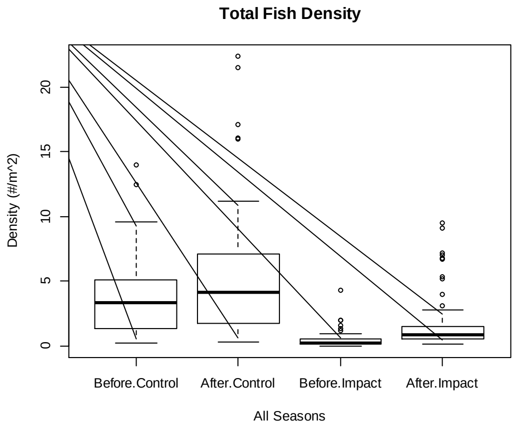
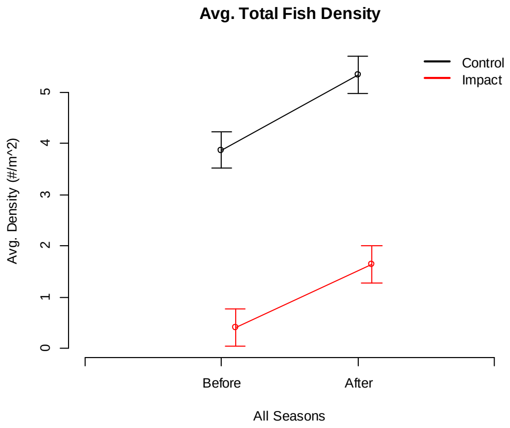
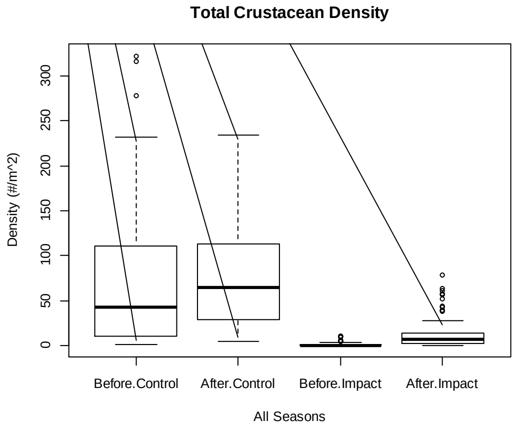
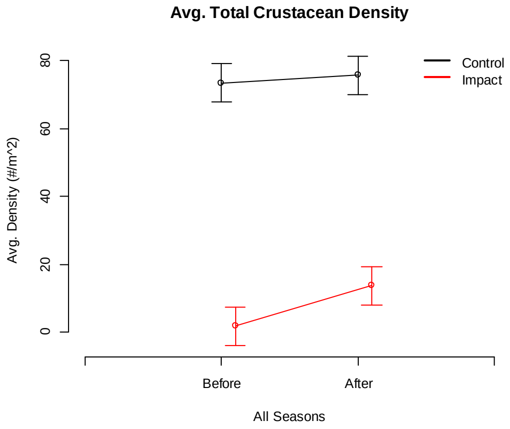
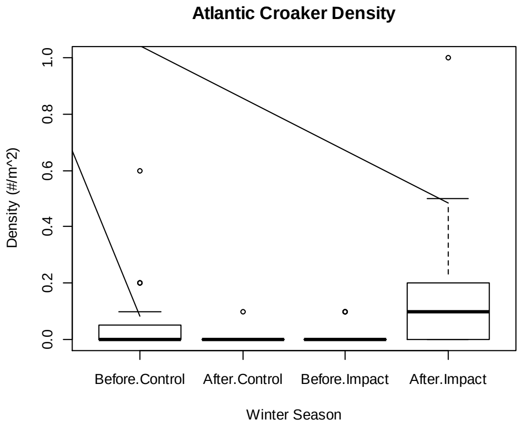

Total Fish Density

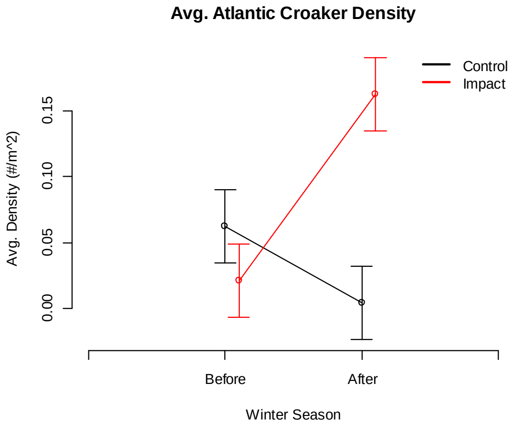

Avg. Total Fish Density


Total Crustacean Density

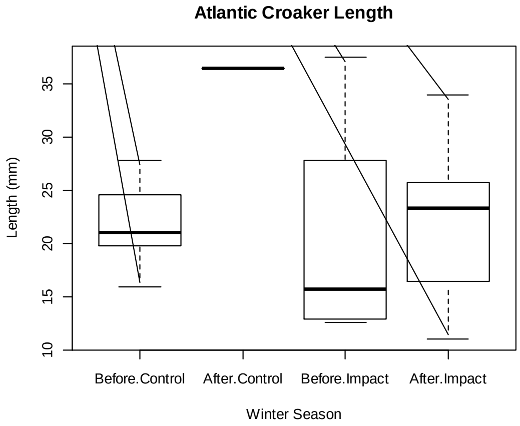

Avg. Total Crustacean Density



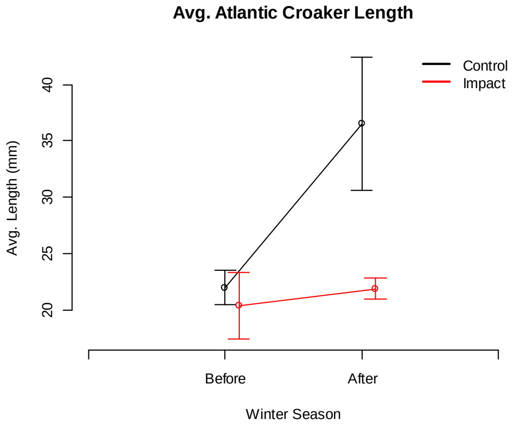
Atlantic Croaker Density



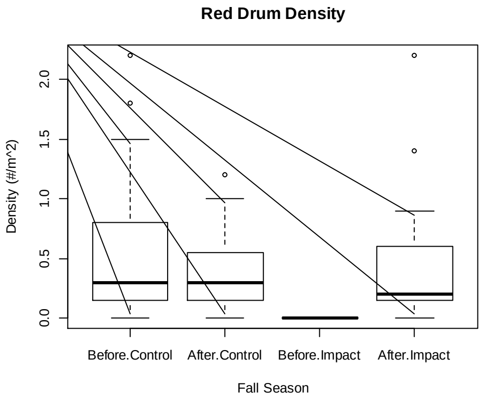
Avg. Atlantic Croaker Density

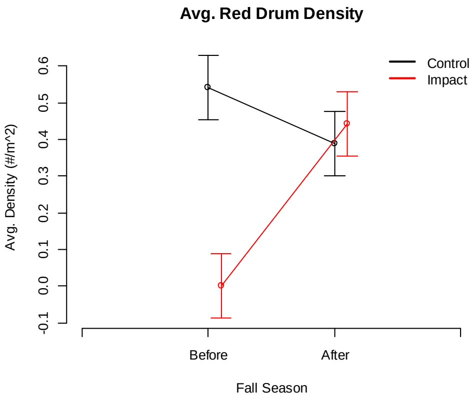


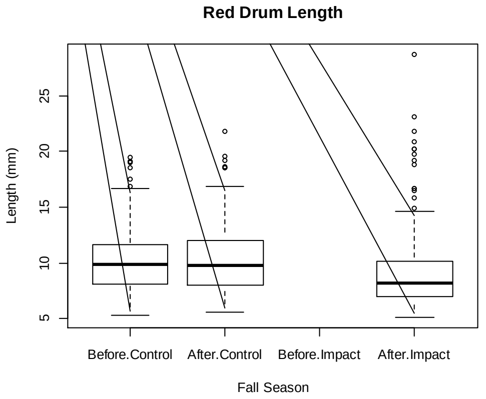
Atlantic Croaker Length

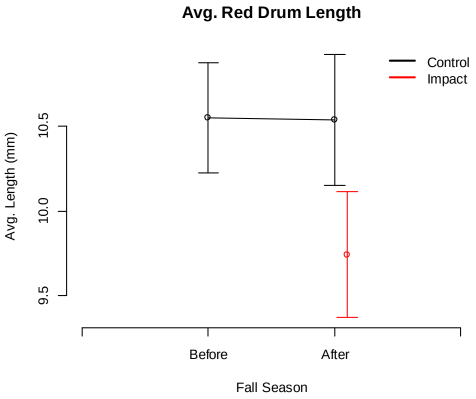


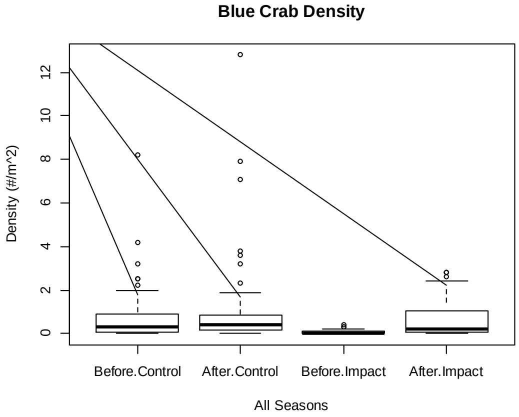
Avg. Atlantic Croaker Length

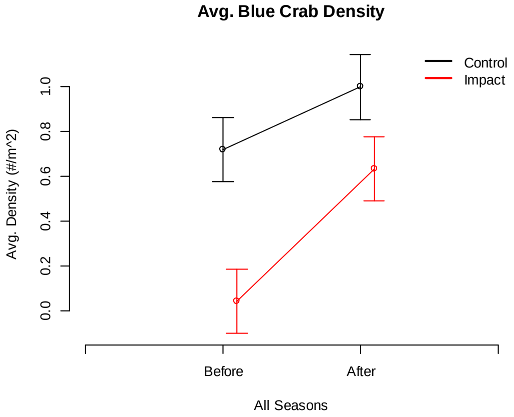

Red Drum Density

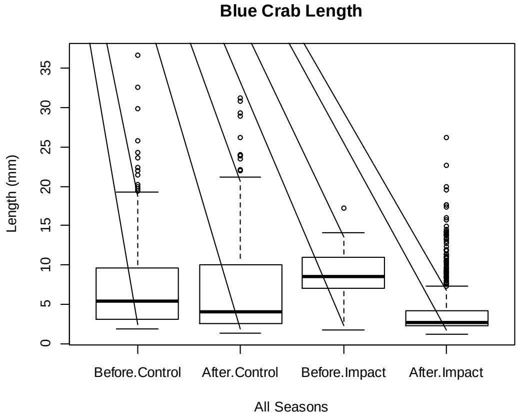

Avg. Red Drum Density

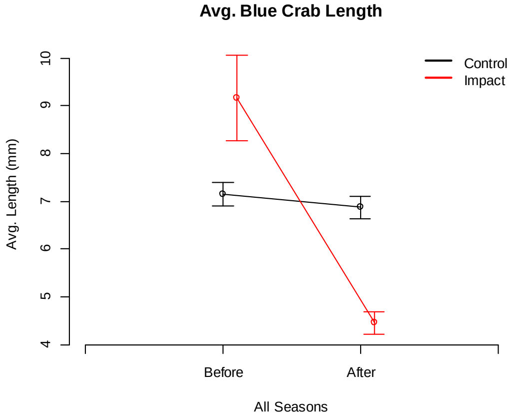

Red Drum Length

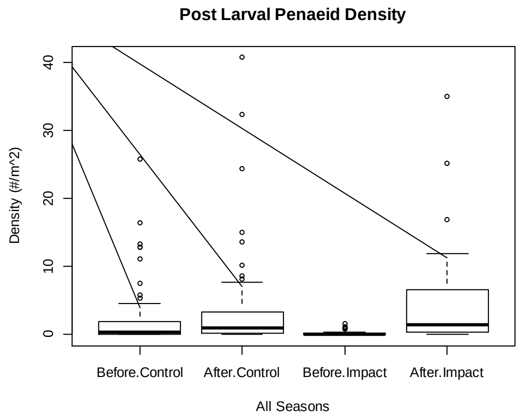

Avg. Red Drum Length

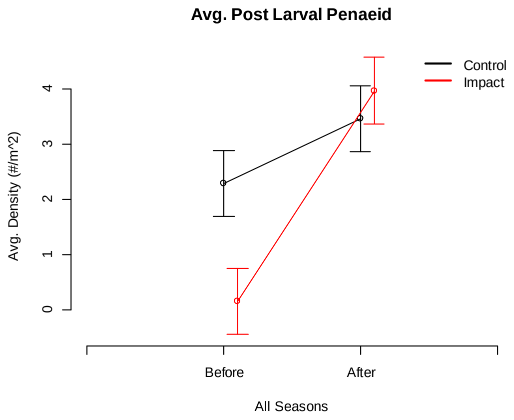

Blue Crab Density
Density (#/m^2)
0
2
4
6
8
10
12
Before.Control
After.Control
Before.Impact
After.Impact
All Seasons

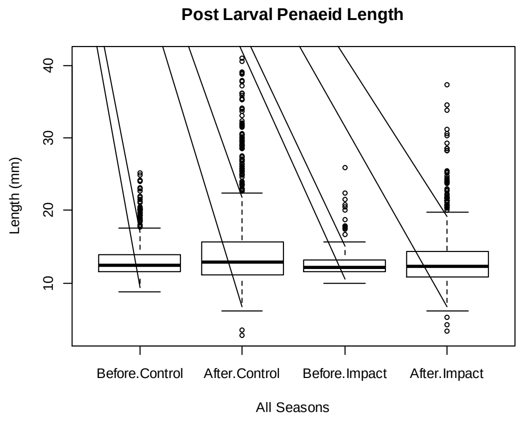

Avg. Blue Crab Density
Avg. Density (#/m^2)
0.0
0.2
0.4
0.6
0.8
1.0
Control
Impact
Before
After
All Seasons

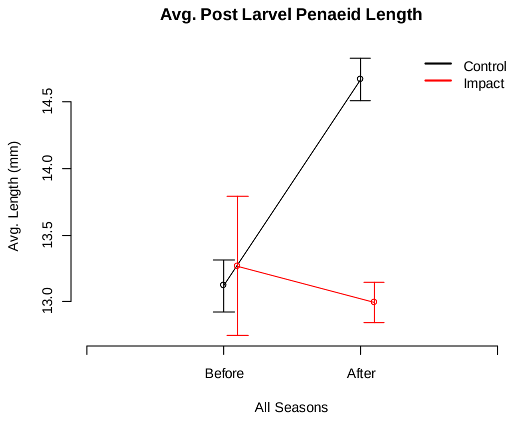

Blue Crab Length
Length (mm)
0
5
10
15
20
25
30
35
Before.Control
After.Control
Before.Impact
After.Impact
All Seasons

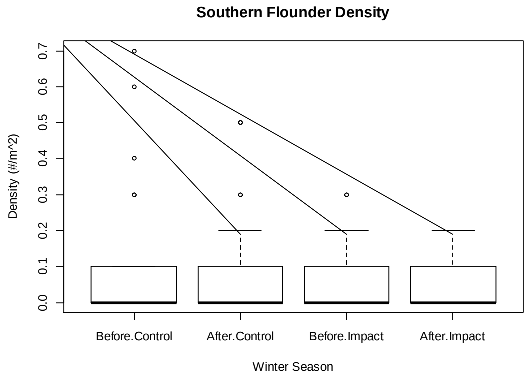

Avg. Blue Crab Length
Avg. Length (mm)
4
5
6
7
8
9
10
Control
Impact
Before
After
All Seasons

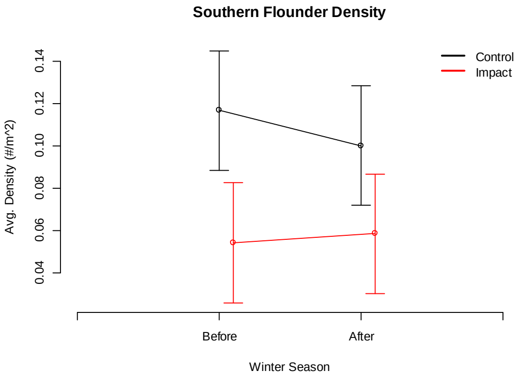

Post Larval Penaeid Density
Density (#/m^2)
0
10
20
30
40
Before.Control
After.Control
Before.Impact
After.Impact
All Seasons

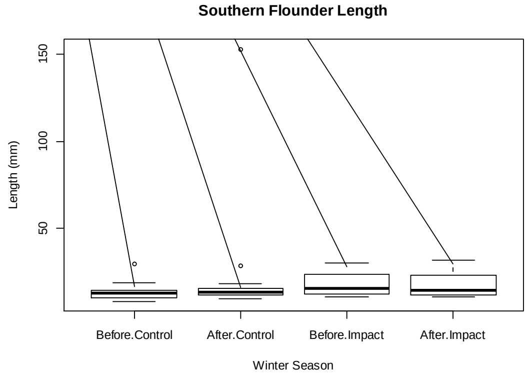

Avg. Post Larval Penaeid
Avg. Density (#/m^2)
0
1
2
3
4
Control
Impact
Before
After
All Seasons

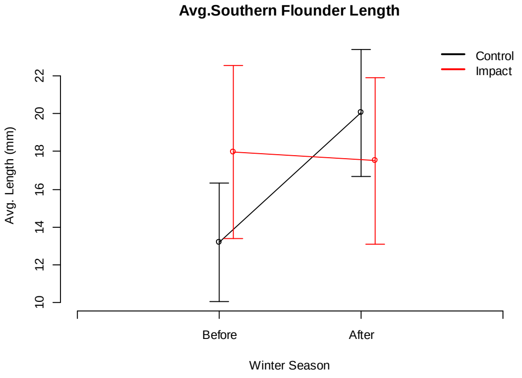

Post Larval Penaeid Length
Length (mm)
10
20
30
40
Before.Control
After.Control
Before.Impact
After.Impact
All Seasons


Avg. Post Larvel Penaeid Length
Avg. Length (mm)
13.0
13.5
14.0
14.5
Control
Impact
Before
After
All Seasons


Southern Flounder Density


Southern Flounder Density


Southern Flounder Length

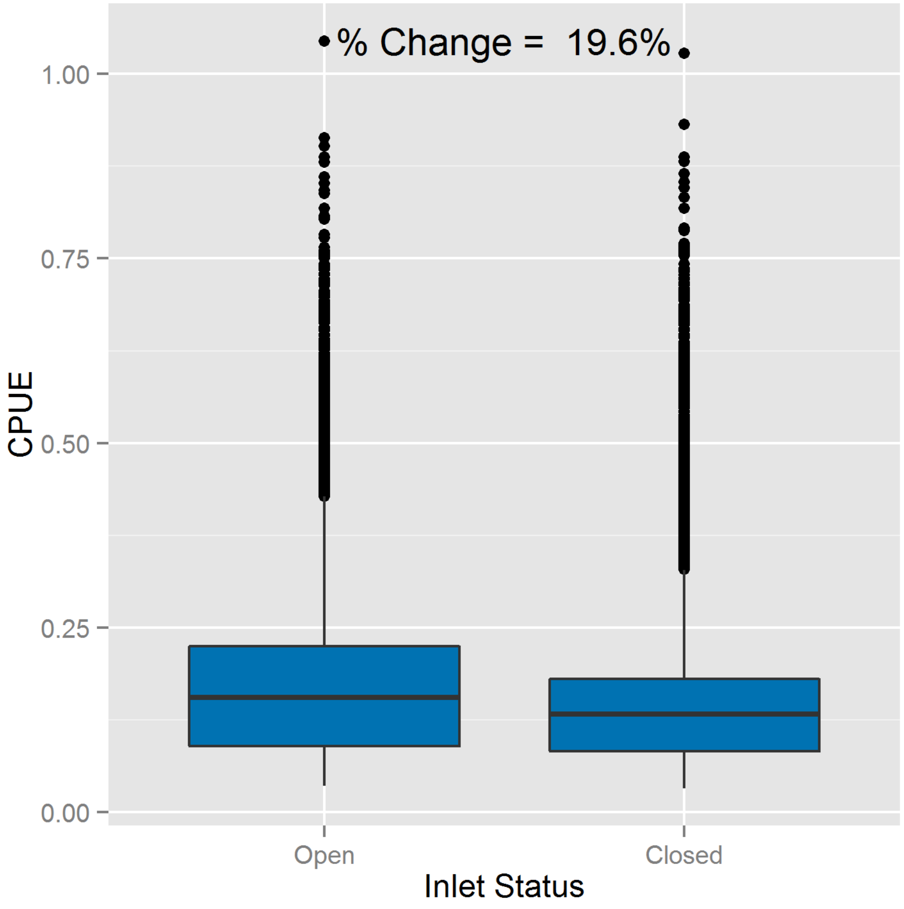

Avg.Southern Flounder Length

APPENDIX C

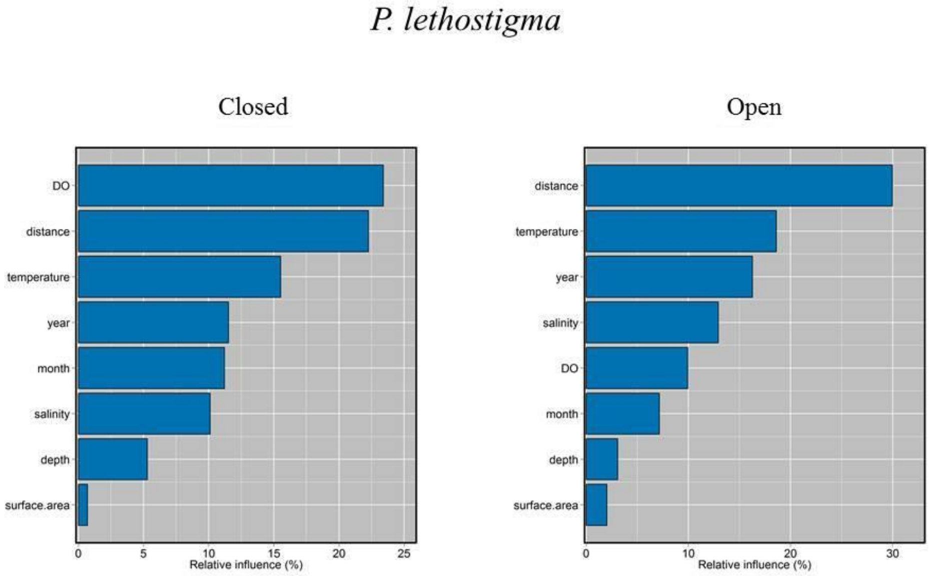
PERMANOVA						
Source	**df**	**SS**	**MS**	**Pseudo-F**	**P(perm)**	**Unique Perms.**
BA	1	2261.5	2261.5	4.782	0.005	998
Le	1	5724.2	5724.2	12.103	0.001	999
BAxLe	1	1038.7	1038.7	2.196	0.07	998
Res	20	9459	472.95			
Total	23	18483				

Appendix C: Community assemblage PERMANOVA table. This analysis tested for an interaction between the before-after (BA) and control-impact (CI) factors. A mildly significant interaction was found.

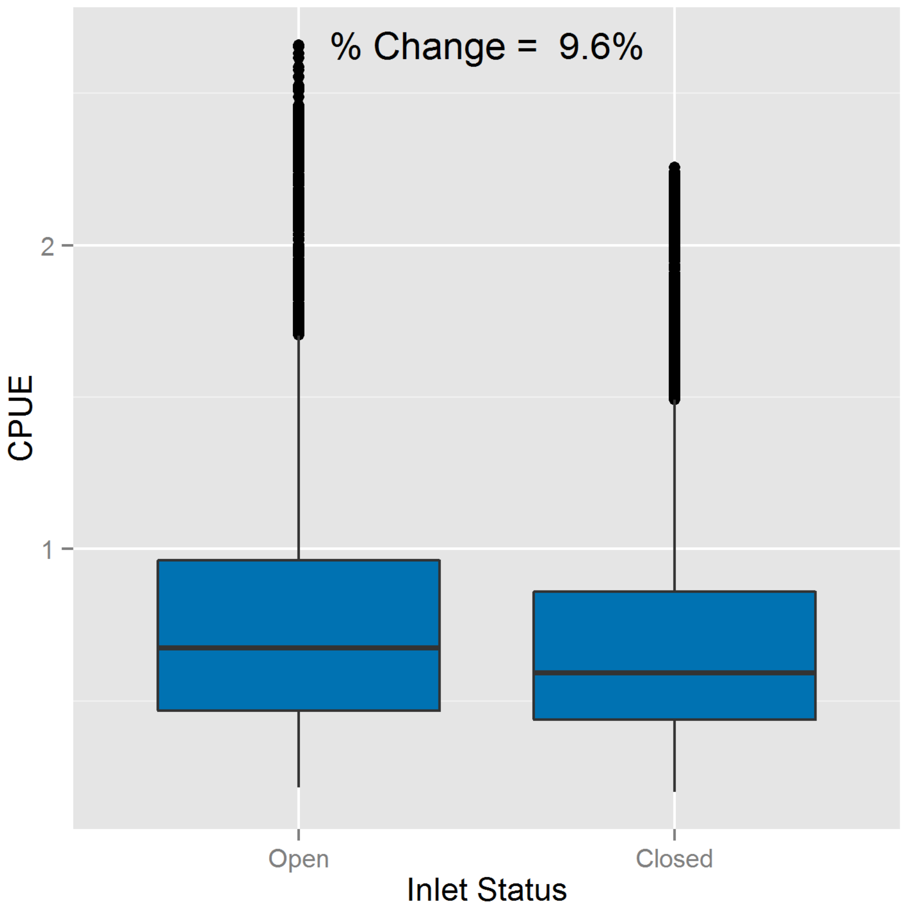
APPENDIX D



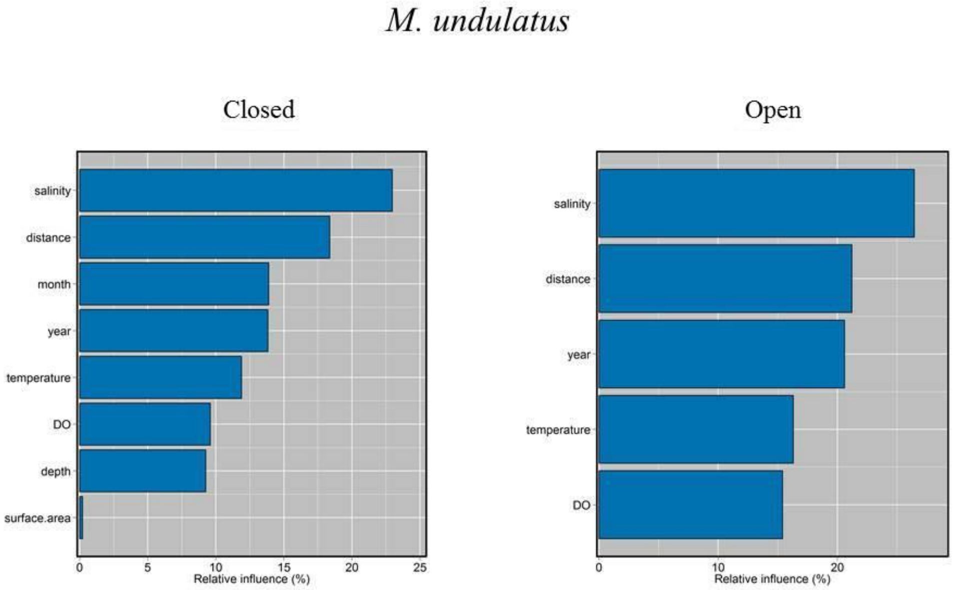
Predicted change of mean *P. lethostigma* CPUE based on the flow status of Cedar Bayou between 1978 and 2003.



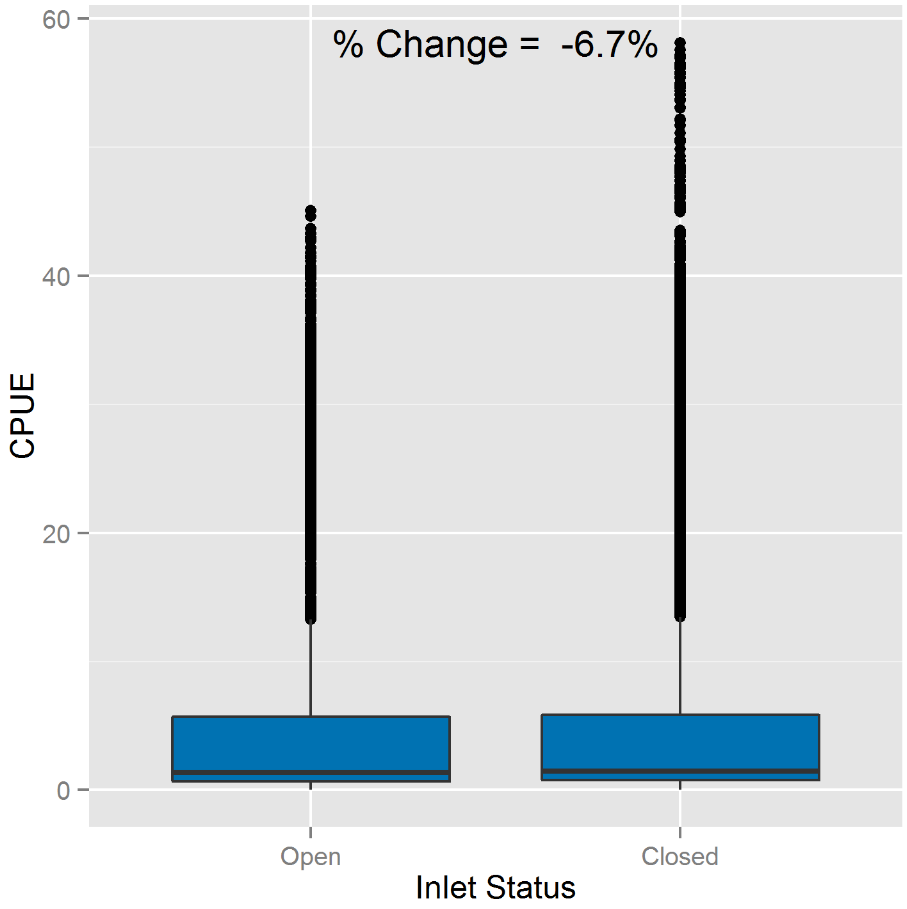
Physical parameters that have the most relative influence on *P. lethostigma* populations during periods of open and closed flow. Physical parameters that have the highest percent relative influence on *P. lethostigma* populations during periods of open and closed flow. Distance from an open inlet was measured in km, DO represents the dissolved oxygen level (%), depth is the average depth of the bay (m), surface area is the surface area of the bay (km^2), temperature (^{o}C), salinity (ppt), and month and year are the temporal variables.



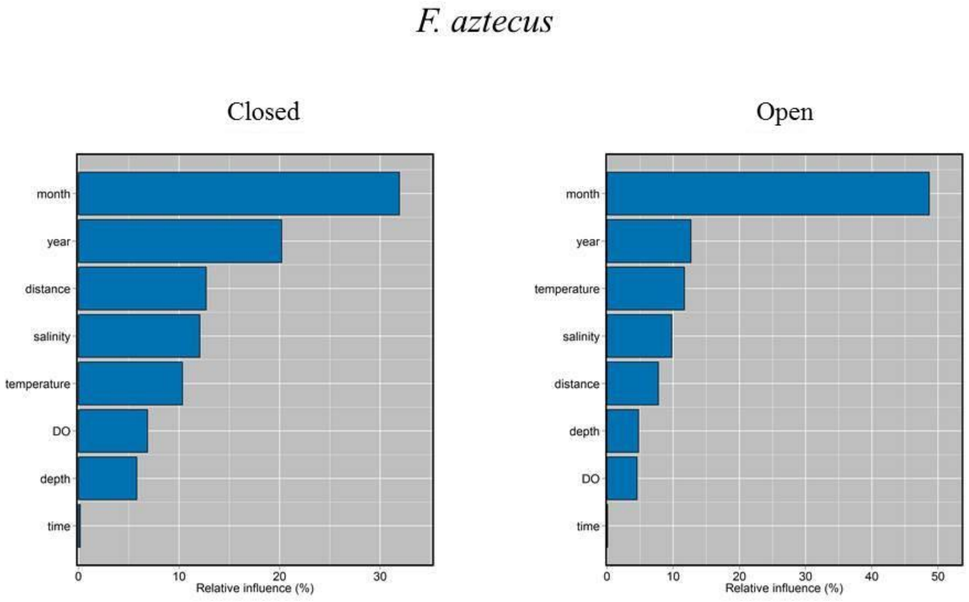
Predicted change of mean *M. undulatus* CPUE based on the flow status of Cedar Bayou between 1978 and 2003.



Physical parameters that have the most relative influence on *M. undulatus* populations during periods of open and closed flow. Physical parameters that have the highest percent relative influence on *M. undulatus* populations during periods of open and closed flow. Distance from an open inlet was measured in km, DO represents the dissolved oxygen level (%), depth is the average depth of the bay (m), surface area is the surface area of the bay (km^2), temperature (^{o}C), salinity (ppt), and month and year are the temporal variables.



Predicted change of mean Brown Shrimp CPUE based on the flow status of Cedar Bayou between 1978 and 2003.



Physical parameters that have the most relative influence on Brown Shrimp populations during periods of open and closed flow. Physical parameters that have the highest percent relative influence on Brown Shrimp populations during periods of open and closed flow. Distance from an open inlet was measured in km, DO represents the dissolved oxygen level (%), depth is the average depth of the bay (m), surface area is the surface area of the bay (km^2), temperature (^oC), salinity (ppt), and month and year are the temporal variables.