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Fish Assemblage Dynamics and Red Drum Habitat Selection in Bayou St. John and Associated Urban Waterways located within the City of New Orleans, Louisiana

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Fish Assemblage Dynamics and Red Drum Habitat Selection in Bayou St. John and Associated Urban Waterways located within the City of New Orleans, Louisiana

A Thesis

Submitted to the Graduate Faculty of the
University of New Orleans
in partial fulfillment of the
requirements for the degree of

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in
Earth and Environmental Science

by

Patrick Smith

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Dedication

I dedicate this manuscript to my wife, Greer Smith. I could not have done this without her love and support. Thank you for your support and understanding.

Acknowledgments

Many people helped me throughout this process. First, I would like to thank my Advisor, Dr. Martin T. O’Connell, for accepting me into his lab and all of the hard work since. I would also like to thank my committee members, Dr. R. Glenn Thomas and Dr. Michael D. Kaller, for their support and guidance along the way. I am very appreciative of Sunny Brogan for all her help since I took over this project. Many people within the Nekton Research Lab have helped along the way with field and analytical work including: Chris Schieble, Will Stein, Jenny Wolfe, Chris Davis, Jon McKenzie, Dr. Tom Lorenz, Angela Williamson, Arnaud Kerisit, Celeste Espinido, Shane Abeare, Miadah Bader, and Iain Kelly. Several volunteers have helped throughout this process; I would like to especially thank Anthony Armato for his hard work and endless enthusiasm. I would also like to thank my family for their support and understanding during this process.

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Abstract

Bayou St. John (BSJ) and City Park Lakes and Lagoons (CPLL) are urban waterways in New Orleans, Louisiana. I studied habitat selection of red drum in BSJ, and fish assemblage change in BSJ and CPLL over 40 years. Temperature was found to be the best predictor of red drum habitat selection in Bayou St. John, whereas salinity and change in depth also were found to be good predictors for certain sites. Potential prey item abundance did not appear to influence habitat selection. Using data from 1971 – 2010, shoreline habitats in CPLL were affected by Hurricane Katrina, but have since recovered and shoreline habitats in BSJ were found to have decreased diversity. Pelagic habitats in both areas were found stable across 40 years. Since 2006, shoreline assemblages were similar for CPLL and BSJ with a decrease in fishes from Order: Cyprinodontiformes and an increase in other fishes seen across years.

Key words: red drum, *Sciaenops ocellatus*, urban fisheries, habitat selection, telemetry

Chapter 1

Introduction

Red drum life history and ecology

Red drum (*Sciaenops ocellatus*) are an estuarine dependent fish species of the Family Sciaenidae (Matlock, 1987). They are one of the largest members of this family and have a broad salinity tolerance (Matlock, 1987; Thomas, 1991; McEachron, 1998; Bachelor, 2009). Red drum occur from Massachusetts to Northern Mexico (Matlock, 1987). Juveniles are most often found in low salinity estuaries, while adults can occur at least 119 km offshore in the Gulf of Mexico (Matlock, 1987; Bachelor, 2009). Although they can be raised from larvae to sexually mature adults in fresh water (Thomas, 1991; McEachron, 1998), estuarine environments are required for larval recruitment and juvenile survival if the species is to live and reproduce in natural habitats (O'Connell, 2005).

Juvenile red drum settle in estuarine, nearshore sub-tidal and intertidal habitats (Pearson, 1928; Bass and Avault Jr., 1975; Matlock, 1987; Beckman *et al*., 1988a; Beckman *et al.*, 1988b; Adams and Tremain, 2000; Scharf, 2000; Scharf and Schlight, 2000; Stuntz *et al.*, 2002; Brown *et al.*, 2004; Dresser and Kneib, 2007; Bachelor, 2008). During this stage, red drum have been shown to exhibit high site fidelity and usually occupy small home ranges (Matlock, 1987; Adams and Tremain, 2000; Dresser and Kneib, 2007). Juvenile red drum movement patterns have been shown to be influenced by tides and solar periodicity, along with both biotic and abiotic factors (Dresser and Kneib, 2007; Bacheler *et al*., 2009). It has been postulated that due to their relatively predictable behavior and small home range, local overfishing of juvenile red drum can and has occurred (Dresser and Kneib, 2007; Bacheler *et al.*, 2008; Bacheler *et al.*, 2010).

Once sexually mature most red drum move offshore and form large schools (Boothby and Avault, 1971; Beckman *et al*., 1988b; Hein and Shepard, 1993; Wilson and Nieland, 1994; Gold and Turner, 2002; Porch *et al*., 2002; Brown *et al*., 2004). Unlike most fish species, there is no clear age or size at which they become sexually mature or when they migrate to deeper subtidal waters (Beckman *et al*., 1988b, Wilson and Nieland, 1994; Wilson and Nieland, 2001). In a Louisiana study, all male red drum examined were found to be sexually mature by age five, over 850 mm SL and over 5.5 kg; all females were found to be sexually mature by age six, over 850 mm SL and over 6.5 kg (Wilson and Nieland, 1994). These mature fish broadcast spawn near tidal passes adjacent to appropriate juvenile habitat (Matlock, 1987; Brown *et al*., 2004). However, in at least one location in northeastern Florida, red drum successfully spawn and may complete their entire lifecycle in a shallow, microtidal estuary with no tidal pass (Johnson and Funicelli, 1991). The behavior of spawning adults includes nudging of females by males while the males make a “drumming” sound (Guest and Lasswell, 1978). A laboratory study also saw increased activity of spawning adults at night (Guest and Lasswell, 1978). After fertilization, eggs float until they reach a salinity of 20 (Peters and McMichael Jr., 1987; Brown *et al*., 2004). At this specific salinity, the eggs are no longer buoyant and some larvae settle into appropriate nearshore estuarine habitat as described above (Brown *et al*., 2004).

Numerous studies exist on the diet of red drum from all age classes (Boothby and Avault Jr., 1971; Bass and Avault Jr., 1975; Overstreet and Heard, 1978; Matlock, 1987; Peters and McMichael Jr., 1987; Llanso *et al*., 1998; Guillory and Prejean, 1999; Scharf and Schlight, 2000). Larval fish feed predominantly on plankton (Bass and Avault Jr., 1975). Juveniles and subadults feed on a variety of food items, with penaeid shrimp, portunid crabs, and teleost fishes being the most important across all studies (Bass and Avault Jr., 1975; Peters and McMichael Jr., 1987; Llanso *et al*., 1998; Guillory and Prejean, 1999; Scharf and Schlight, 2000). Polychaete worms are also listed as prey items for juvenile and subadult red drum (Overstreet and Heard, 1978; Peters and McMichael Jr., 1987; Llanso *et al*., 1998). Both polychaetes and mud crabs (Family Xanthidae) were found to be a part of red drum diet when occurring in habitats with an un-vegetated substrate (Bass and Avault Jr., 1975; Peters and McMichael Jr., 1987). The diet of adult red drum is similar to that of juvenile and subadult fish. The main difference noted is that larger prey items of penaeid shrimp, portunid crabs, and teleost fishes are taken by adults (Boothby and Avualt Jr., 1971; Overstreet and Heard, 1978; Guillory and Prejean, 1999). In a review of literature on red drum diet, it was determined that blue crabs (*Callinectes sapidus*) were the most important prey item for red drum (Guillory and Prejean, 1999). Foraging studies also indicate a high level of plasticity in the diets of red drum from specific areas and certain times of the year (Matlock, 1987; Llanso *et al*., 1998). That is, it has been shown that red drum do not select for specific prey items. This strategy has been suggested by a study on red drum diet in a saltwater impoundment (Llanso *et al*., 1998).

Red drum were once an important commercial fish species in Louisiana and remain an important game fish for the State as well as the rest of its range (Boothby and Avault, 1971; Bass and Avault, 1975; Wakeman and Ramsey, 1985; Hein and Shepard, 1986; Beckman *et al.*, 1988a; Beckman *et al.*, 1988b; Wilson and Nieland, 1994; Exec. Order No. 13449, 2007). After an assumed decrease of red drum in the mid-1980s, commercial harvest was banned in 1990 for the entire Northern Gulf of Mexico (Wilson and Nieland, 1994; Scharf, 2000). According to the Federal Recreational Fishing Regulations, it is currently illegal to harvest or possess any red drum in the federal waters of the United States. In 2007, an executive order was written stating the importance of conserving red drum in the United States based upon sound science (Exec. Order No. 13449, 2007).

Due to their popularity as a sport fish and assumed stock declines, some states such as Texas have implemented widespread aquaculture and stocking practices. Fingerlings have been stocked throughout Texas starting since the mid-1980s (McEachron *et al.*, 1998). However, any benefits of stockings have been difficult to document (Scharf, 2000). Currently, no large public aquaculture and stocking program such as this exists in the state of Louisiana. In addition to stock enhancement programs' limited success other reasons suggest that a stocking program may not be successful in Louisiana's unimpounded marshes. The dominant broken marsh habitat in Louisiana is probably not suitable for stocking success because these habitats are more complex (Chesney *et al.*, 2000).

Site background

Many anthropogenic impacts have affected Bayou St. John (BSJ) over the past few centuries, since the founding of New Orleans (Ward, 1982). The Bayou has been dredged, dammed, pumped, cemented, channelized, shortened, lengthened, widened, narrowed, and disconnected from and reconnected to various natural and artificial waterways (Ward, 1982; Brogan, 2010). Currently, there is a series of pumps, culverts, sluice valves, butterfly valves, storm water drains, and diversions that control water flow in, out, and throughout the Bayou (Lake Pontchartrain Basin Foundation, 2006; Burk-Kleinpeter, Inc., 2011). The sector gate, located at the mouth at Lake Pontchartrain, contains three valves, two measuring 91.44 cm in diameter and one at 60.96 cm used to manage BSJ's water level. An old flood control structure exists south of this and it is regulated by three Pratt Butterfly valves: one is rusted shut, one is rusted open, and the third is rusted partially open (Lake Pontchartrain Basin Foundation, 2006). A 60.96 cm differential valve located at the extreme southern end and a 76.2 cm culvert at I-610 are used for drainage. Much of BSJ has cement banks to aide in the prevention of erosion and thus much of the original submersed aquatic vegetation and riparian plant life are reduced.

Elevated levels of toxins occur in BSJ sediments and water, with lead (Pb) and polycyclic aromatic hydrocarbons (PAHs) having the most common high values across samples (Mowat and Bundy, 2001; Wang *et al*., 2004). Higher concentrations of PAHs were found in the southern portion of BSJ and this may be attributed to heavier automobile traffic in this region or increased sediment input from Lake Pontchartrain in the north (Wang *et al.*, 2004). Heavy metals such as lead and arsenic were found to be above U.S. Environmental Protection Agency standards in water samples taken post-Katrina (Pardue *et al.*, 2005). Periodic magnitudes in fecal coliform counts also occur in BSJ (McCorquodale, 2004).

Recent initiatives have been put in place to help improve this severely altered and degraded waterway (Lake Pontchartrain Basin Foundation, 2006; Brogan, 2010; Burk-Kleinpeter, Inc., 2011; Schroeder, 2011; Pezold, 2012). Collaboration between the Orleans Levee District and Burk-Kleinpeter, Inc. (BKI) has generated a plan for water level management (Burk-Kleinpeter, Inc., 2011). This plan suggests a more ecological approach to water management be taken, with the major goal being increased fishery productivity. The Faubourg St. John Neighborhood Association has partnered with the Louisiana State University Agricultural Center and Bayou Land Resource Conservation and Development to plant native emergent grasses for habitat restoration (Pezold, 2012). The Bayou St. John Action Plan suggests the stocking of appropriate wild and hatchery-raised fishes and crabs as a method to aide in the recovery of BSJ recreational fisheries (Lake Pontchartrain Basin Foundation, 2006).

To help meet this recommendation, the Louisiana Department of Wildlife and Fisheries (LDWF) stocked largemouth bass (*Micropterus salmoides*) in 2006. In addition, the effectiveness of a red drum stocking program for BSJ is currently being studied through a joint project between LDWF and the Nekton Research Laboratory (NRL) at the University of New Orleans.

As part of the red drum stocking program, approximately 75 wild-caught red drum were stocked in Bayou St. John from 2006 to 2008 to determine the suitability of BSJ for a red drum stock enhancement program. Fish were fitted with acoustic telemetry equipment and were tracked for two years, both manually and remotely. It was found that tagged individuals were found more often in the northern habitats of BSJ. No fish was ever detected South of Interstate 610 (I610) while being manually tracked. Red drum were detected much less frequently south of I610 than north of I610 during remote tracking. No significant differences in water quality parameters were found between the northern and southern sections of BSJ. Significant differences were found for width and depth, with the northern section being deeper and wider. (Brogan, 2010)

Current Study

My research was a continuation of the previous study, building on its findings. Specifically, I considered the influence of potential prey item abundances and whether changes in various water quality parameters affected red drum habitat selection. In other studies, red drum have been shown to select habitats based upon both of these criteria (Dresser and Kneib, 2007; Bacheler *et al.*, 2009). Whereas red drum in BSJ appeared to select northern sections; I examined possible differences in biotic and abiotic factors within the Bayou that might explain this selection. More specifically, the goals of this project were to answer the following questions:

1. Is there a relationship between the occurrence and composition of potential prey items and red drum habitat selection in BSJ?
2. How do changes in abiotic variables affect red drum habitat selection in BSJ?

Materials and Methods

Site Description

Located in the north-central portion of the City of New Orleans, Louisiana (Fig. 1), BSJ is an urban waterway. It is approximately 6.5 km long and for most of its length has a north-south orientation. The width of the bayou varies from 45 m to 200 m (Martinez *et al*., 2008; Brogan, 2010). Depths range from 1.3 to 3.5 m, with the northern section (north of I610) being significantly deeper and wider than the southern section (Martinez *et al*., 2008; Brogan, 2010). The northern extremity is partially connected to Lake Pontchartrain, an oligohaline embayment, by a sector gate (781343 m E, 3325059 m N; Zone 15 R; UTM). The most southern point ends at the corner of Jefferson Davis Parkway and Lafitte Street (780677 m E, 3319389 m N; Zone 15 R; UTM). Its connection with Lake Pontchartrain provides BSJ with brackish water (salinity ranges from 1.5 to 8). The water level is maintained by sluice valves on a sector gate near the BSJ and Lake Pontchartrain confluence. Current management of surface water height is set at approximately -0.24 m NAVD88 (BKI, 2011).

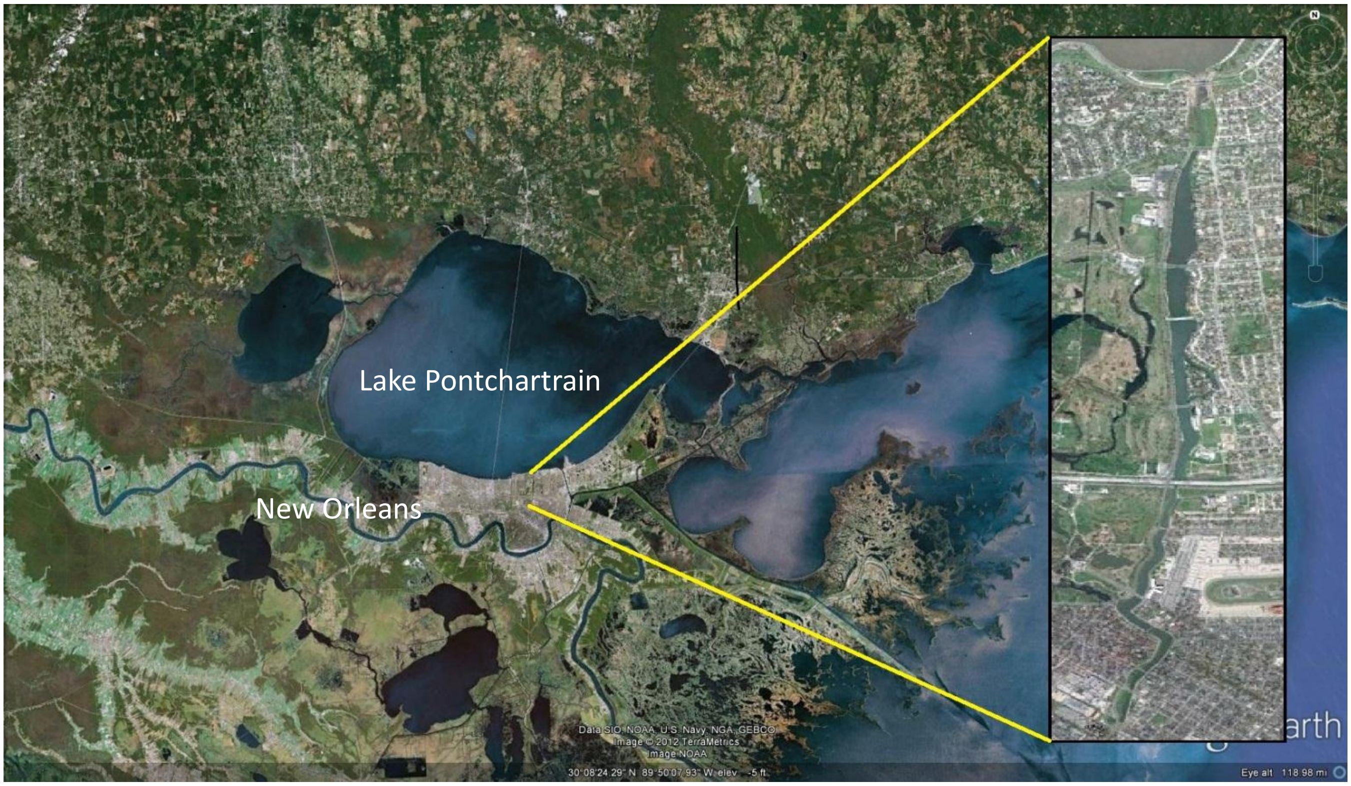

Figure 1: Image of Bayou St. John (excerpt) and its relation with Lake Pontchartrain and New Orleans, LA. Notice its location within the urbanized area surrounding New Orleans. The Eastern portion of City Park Lakes and Lagoons can also be seen West of the Bayou. Image adapted from Google Earth©.

Abundance of Potential Prey

I conducted a remote tracking study to determine if the occurrence of red drum in BSJ was related to the occurrence and composition of potential prey items. Much of these efforts were a continuation of previous tracking research and I used much of the same equipment outlined in Brogan (2010). Of the original 19 fish tagged and tracked by Brogan (2010) in 2009, six were still being detected every month from September through December 2010. I used these six fish as my focal organisms. These red drum were surgically implanted with VEMCO V13-1L-69 KHz transmitters (Length = 52-96 mm, Diameter 13 mm, weight = 9-16 g) which have batteries expected to last well beyond the time of my research (August – December 2011, depending on activation and deployment). Transmitter specific hydroacoustic signals were detected at three VEMCO VR2W-coded acoustic receivers deployed in BSJ. These receivers were moored in the same position for the duration of this study at three sites: Robert E. Lee Boulevard (REL), North End Island (NEI), and Interstate 610 (I610; Fig. 2). After checking for assumption violations, I conducted an analysis of variance (ANOVA) on the number of pings per day per site across the study to determine if the number of pings per day were significantly different among these three sites (α = 0.05). If significant differences were found, I performed Tukey HSD post-hoc analyses to test for pair-wise differences (α = 0.05).

Figure 2: Image of Bayou St. John and epibenthic survey (minnow traps), shoreline survey (beach seine), receiver location, and continuous water quality station from this study. The blue ovals represent the area covered during the epibenthic survey, the yellow ovals represent shoreline fish survey sites, the red ovals represent the location of the receivers, and the orange oval represents the continuous water quality monitoring site. Image adapted from Google Earth©.

To sample benthic epifauna (i.e., potential prey), I used galvanized steel Gee minnow traps (228.6 mm X 444.5 mm) with a 6.35 mm mesh and 25.4 mm opening. Three minnow traps were placed arbitrarily in eight sections along the length of BSJ monthly from May 2010 through May 2011 (Fig. 2). November's samples were not included in any analysis because all of the traps from I610 were missing when retrieval was attempted. Samples from September through December 2010 were analyzed and compared to the average number of daily pings. Random selection of sampling sites was considered, but it is believed a high probability of public interference outweighs the benefit of random sampling. To avoid public interference, minnow traps were placed near the center of BSJ in an attempt to avoid shore-bound human interaction. Due to low diversity, these samples were not analyzed as an assemblage. These data were analyzed using Wilcoxon non-parametric tests using R statistical software ($\alpha = 0.05$) to test for among-site differences in the abundance of organisms.

Data outside the time period when receivers were deployed were also analyzed because the low sample size may affect results ($n = 9$). Analyzing all samples from May 2010 through 2011, greatly increases sample size ($n = 36$ versus $n = 9$). I tested for significant differences in abundances of organisms collected from the entire survey using the same Wilcoxon non-parametric tests ($\alpha = 0.05$).

From September 2010 through December 2010, NRL personnel sampled three sites monthly using a 5 m beach seine. Two of the three seining sites, REL and Mirabeau Boulevard co-occurred with moored receivers and minnow trap samples. One site located near Dumaine Street did not occur within the area of a moored receiver. These samples were geared towards assemblage analyses by using three standardized seine hauls per site each month with data recorded for each haul. A Bray-Curtis similarity matrix was created using these assemblage data. Analysis of similarity (ANOSIM) was then performed using Primer 5 software (Clarke, 1993; $\alpha = 0.05$). The statistic used in ANOSIM, Global R, has values ranging from -1 to 1. Values approaching 0 indicate similar among groups and within group variation, values approaching 1 indicate higher variation among the groups than within the groups, and values approaching -1 indicate higher variation from within group (Clarke, 1993). Any significant differences across sites were further analyzed using similarity percentage analysis (SIMPER;

Clarke, 1993). This analysis lists species that contribute most to any dissimilarity displayed in the pairwise ANOSIM tests.

The species that drive any changes in either of the sampling surveys mentioned above were compared to the ample list of red drum prey items in the literature. If species that drive the change in assemblages were considered potentially be prey item(s) for red drum, it was compared, by inspection, to the daily number of pings near the sampling site. Without any data on the diet of red drum in BSJ, potential prey items were only referenced with other studies.

Water Quality Modeling

I analyzed continuous water quality data in a way to better understand red drum's response to change in abiotic conditions. From 1 September through 31 December 2010, a remote monitoring continuous water quality station collected specific conductivity, dissolved oxygen as percent and concentration, salinity, and depth every fifteen minutes in BSJ (Fig. 2). Data are directly linked to a database web server (YSI - Remote Monitoring and Control System, 2010). The calibration of each station was maintained by The Louisiana State University Agricultural Center and is currently maintained by the Louisiana Department of Wildlife and Fisheries. Daily averages from 1 September 2010 through 31 December 2010 were calculated from this station (Fig. 2). These values are not meant to represent the average daily values for BSJ's entirety. The change in these daily values is used to estimate the change across the Bayou.

Analysis of these continuous variables was a multi-step process. The first step was to determine appropriate tests by analyzing each of the predictor variables. Since specific conductivity and salinity are different expressions of the same measurement, only one of them is appropriate for analysis. Salinity was chosen because its values are the most common in the literature and have been found to influence red drum behavior (Dresser and Kneib, 2007; Bacheler *et al*., 2009). Dissolved oxygen was represented as both a concentration and as a percentage. Percent dissolved oxygen is a factor of water temperature, so only dissolved oxygen as a measurement was used (mg/L). Temperature and depth did not have any mathematical dependencies with other variables, so they were both used. After deciding which variables to analyze, each variable was compared pairwise in linear regression models to determine any between-predictor variable pairwise collinearities. Each was significantly correlated with every other variable in pairwise testing ($p < 0.001$; Fig. 3). Additionally, since sampling was not random with respect to each variable, a test robust to both collinearity and interdependence of variables needed to be selected.

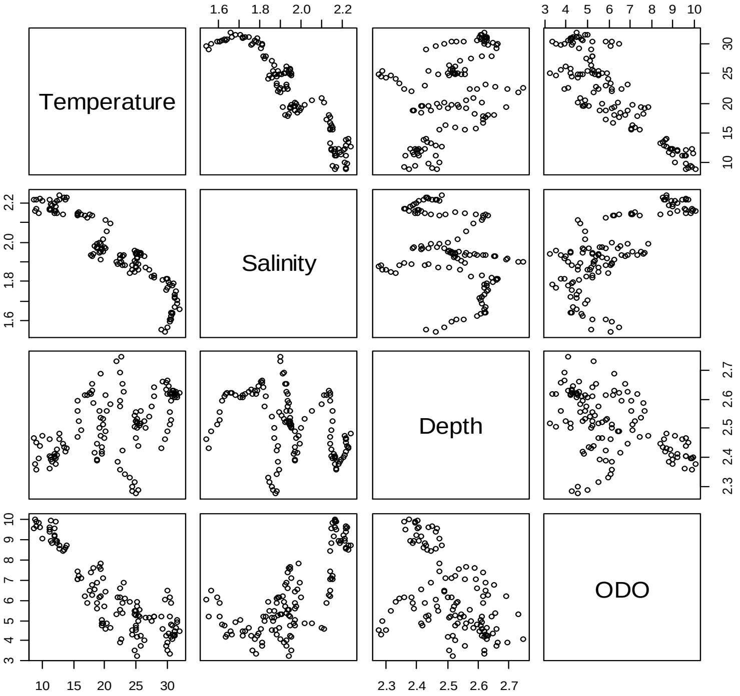

Figure 3: Figure depicting all the pairwise relationships between predictor variables used in the models. “ODO” is an abbreviation for Dissolved Oxygen as measured by the instrument using an optical probe. Note the strong straight line linear relationships between these variables. Each predictor variable was analyzed pairwise using linear regression models to test for collinearity. All were found to have significant linear relationships ($p < 0.001$).

I chose Generalized Estimating Equations (GEEs) as the most appropriate statistical tool (Liang and Zeger, 1986). These tests are robust to correlated predictor variables as well as spatial auto-correlation, and observational correlations. Multiple GEEs, each testing a different response variable and the same predictor variables, were analyzed. The four response variables were the number of pings per day from REL, NEI, I610 and the total number of pings per day for all sites (Total). The predictor variables were salinity, dissolved oxygen concentration, depth, and temperature. Step-wise model reductions with an exchangeable correlation structure were performed using the GEEpack for R statistical software (Hojsgaard *et al.*, 2005). Interactions were not included in these models because all variables were highly correlated.

After determining appropriate analyses and predictor variables, two different approaches of selecting predictor variable and response variable relationships were used. The first mentioned here was mathematically driven, and is referred to as “Mathematical Models” or “Mathematical GEEs”. These used multiple regressions to determine the relationship between each predictor variable – response variable relationship. This approach was unbiased in that logical or practical relationships between variables were not considered. The second approach I considered was driven by logical and practical relationships, and is referred to as “Practical Models” or “Practical GEEs”. These models were based relationships that seemed likely to occur in nature. Once the relationship between each variable was established using each approach, a GEE step-wise model reduction comparing each response variable (REL, NEI, I610, Total) against all four predictor variables (salinity, dissolved oxygen concentration, depth, and temperature) was developed.

Mathematical Models

Before analyzing all of the variables in one model, the relationship between each predictor and explanatory variable was determined. In order to do this, I fitted several regression models between each predictor variable and explanatory variable in a stepwise manner. First, linear models were tested and then higher order polynomials were added until the addition of one did not significantly increase the amount of variation explained by the predictor variable for the response variable ($\alpha = 0.05$). The highest order polynomial fit that significantly increased the amount of variance explained in the regression model was chosen to represent the relationship between predictor and explanatory variable.

Practical Models

Each of the variables used in the Practical GEEs were based on what makes the most ecological sense. Between mean daily temperature and the number of pings per day, a second-order relationship seems likely, suggesting that data including temperatures below, above, and optimal for red drum activity. It is likely that this occurred in our study period at our site based upon a review of red drum’s natural range (Massachusetts to Northern Mexico; Matlock, 1987) and aquaculture experiments (Thomas, 1991). Salinity has been shown to influence red drum habitat selection, with either low or high values being selected (Bacheler, 2009). Therefore, a straight line linear relationship between salinity and the mean number of pings per day was chosen. Similarly, depth has been shown to be a good predictor of red drum habitat selections, with different habitats being selected at low and high values (Dresser and Kneib, 2007). Like salinity, a straight line linear relationship was chosen for depth. Dissolved oxygen was not considered in any of these models. This was because the most logical response to dissolved oxygen would be avoidance of an area based on low dissolved oxygen levels. Since there was only one station where dissolved oxygen was recorded, it was not included in the Practical Models. Additionally, an aquaculture study found juvenile red drum to be tolerant of low dissolved concentrations (< 3.0 mg/L; Thomas, 1991).

Results

Prey Abundances

The number of pings detected at each site was found to be significantly different (ANOVA, F = 186.1, $p < 0.001$). Tukey's post hoc analysis was performed and found that each pairwise test between sites was significantly different (REL~NEI: $p < 0.001$; REL~I610: $p < 0.001$; NEI~I610: $p < 0.001$). Higher mean daily pings were found for REL followed by NEI, I610 had the lowest mean daily pings (REL = 273.6311, s.d. = 173.3973; NEI = 94.6056, s.d. = 86.84569; I.610 = 1.42623, s.d. = 5.52368; Fig. 4).

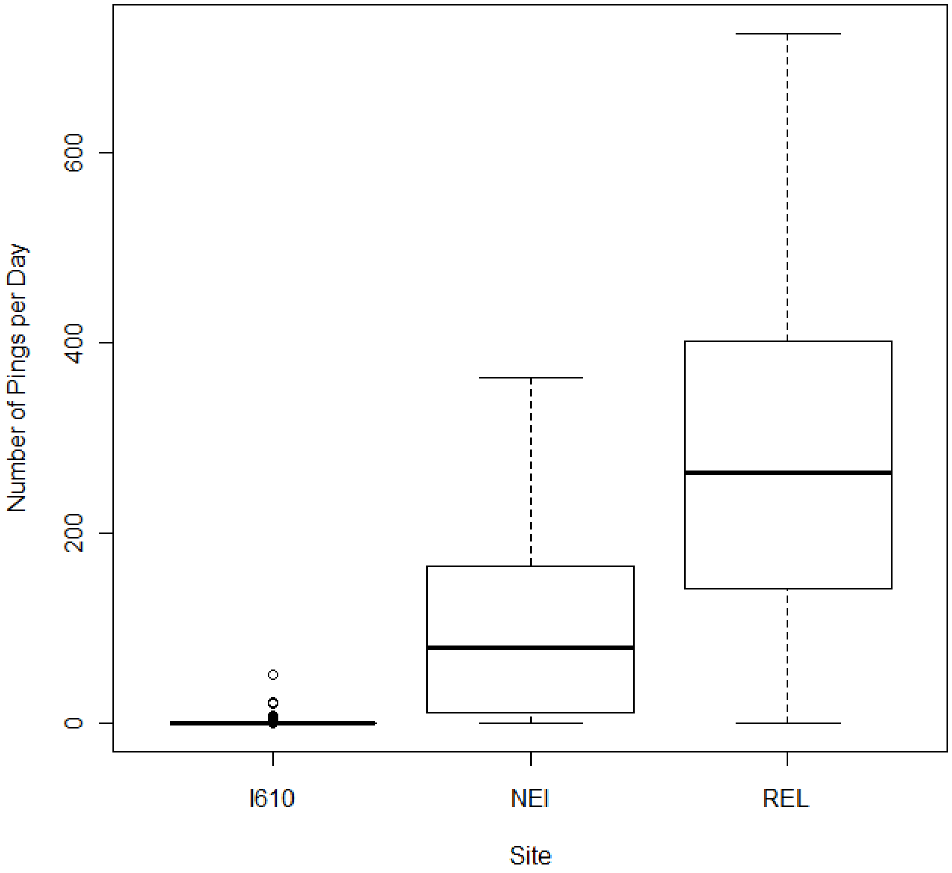

Figure 4: Box plot of mean, mode, and standard deviation in the number of pings per day for each site from September through December 2010, where I610 is Interstate 610, NEI is North End Island, and REL is Robert E. Lee. A ping occurs whenever a tagged red drum is within a receiver located at any of the three sites. The y-axis is the number of pings per day and the x-axis is the factor site.

For the entire study (May 2010 – May 2011), four species, estuarine mud crab (*Rhithropanopeus harrisii*; Family: Xanthidae), bluegill (*Lepomis macrochirus*), *M. salmoides*, and Gulf pipefish (*Syngnathus scovelli*) were collected as a part of the benthic epifaunal minnow trap survey (Table 1). Of these four species, only abundances of *R. harrisii* were analyzed using Wilcoxon non parametric tests. No analyses were conducted on the other species because of their low abundances.

During the study period of September through December 2010, two species were sampled, *R. harrisii* and *L. macrochirus* (Table 2). Again, only *R. harrisii* abundances were analyzed due to low abundances of *L. macrochirus*. Pairwise tests between Robert E. Lee and both of the other sites were found to be significantly different (Wilcoxon; REL vs. NEI: W = 62.5, $p = 0.03585$; REL vs. I610: W = 65, $p = 0.01558$). The pairwise test between North End Island and Interstate 610 was not found to be significant (Wilcoxon, W = 36, $p = 0.5848$; Table 5). The average number of mud crabs sampled was found to be higher for REL than NEI or I610 (REL: μ= 1.889 s.d. = 1.900; NEI: μ= 0.222, s.d. = 0.441; I610: μ= 0.333 s.d. = 0.333; Fig. 5).

Table 1. Number of individuals for each species collected from minnow traps sampled in Bayou St. John from May 2010 through May 2011 per site and overall. Triplicate samples were collected monthly as per the methods, except for the month of November (n=108).

Species and Number Collected per Site (5/1/2010 – 5/31/2011)

Site	*Rhithropanopeus harrisii* (estuarine mud crab)	*Lepomis macrochirus* (bluegill)	*Micropterus salmoides* (largemouth bass)	*Syngnathus scovelli* (Gulf pipefish)
REL	29	4	0	2
NEI	23	1	2	0
I610	33	10	0	0
Total	85	15	2	2

Table 2. Number of individuals for each species collected from minnow traps sampled in Bayou St. John from September through December 2010 per site and overall. Triplicate samples were collected monthly as per the methods, except for the month of November (n=27).

Species and Number Collected per Site (9/1/2010 – 12/31/2010)

Site	*Rhithropanopeus harrisii* (estuarine mud crab)	*Lepomis macrochirus* (bluegill)	*Micropterus salmoides* (largemouth bass)	*Syngnathus scovelli* (Gulf pipefish)
REL	21	1	0	0
NEI	3	0	0	0
I610	4	7	0	0
Total	28	8	0	0

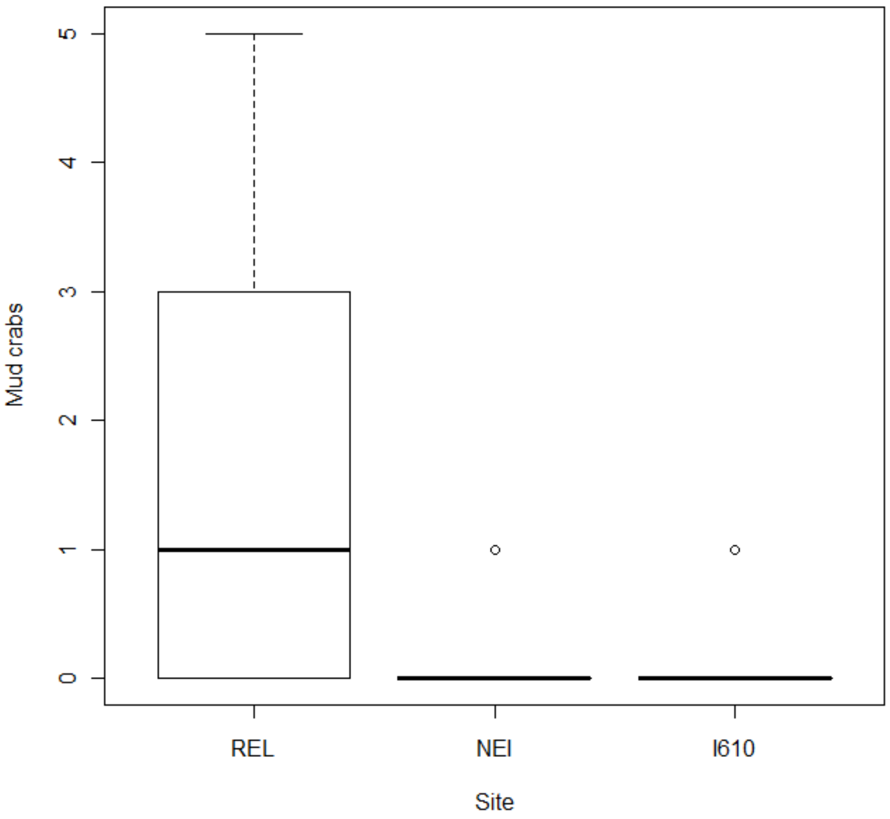

Figure 5: Box plot of mean, mode, and standard deviation in the number of *R. harrisii* collected per replicate for each site from September through December 2010, where I610 is Interstate 610, NEI is North End Island, and REL is Robert E. Lee. The mean and standard deviation for each site was REL: μ= 1.889 s.d. = 1.900; NEI: μ= 0.222, s.d. = 0.441; I610: μ= 0.333 s.d. = 0.333. The y-axis represents the number of *R. harrisii* per replicate and the x-axis is the factor site.

The results from the entire study period (May 2010 – May 2011) suggest that there is no significant difference among sites as a result of pairwise Wilcoxon tests (REL ~ NEI: W = 597.5, *p* = 0.5206, REL ~ I610: W = 670, *p* = 0.7882, NEI~I610: W = 718, *p* = 0.3744). Also, the mean and standard deviation for REL, NEI, and I610 were similar and all less than one, μ= 0.806, s.d. = 1.261; μ= 0.639, s.d. = 1.099; μ= 0.9167 s.d. = 1.380, respectively (Fig. 6). These results suggest similarly low *R. harrisii* numbers among all sites.

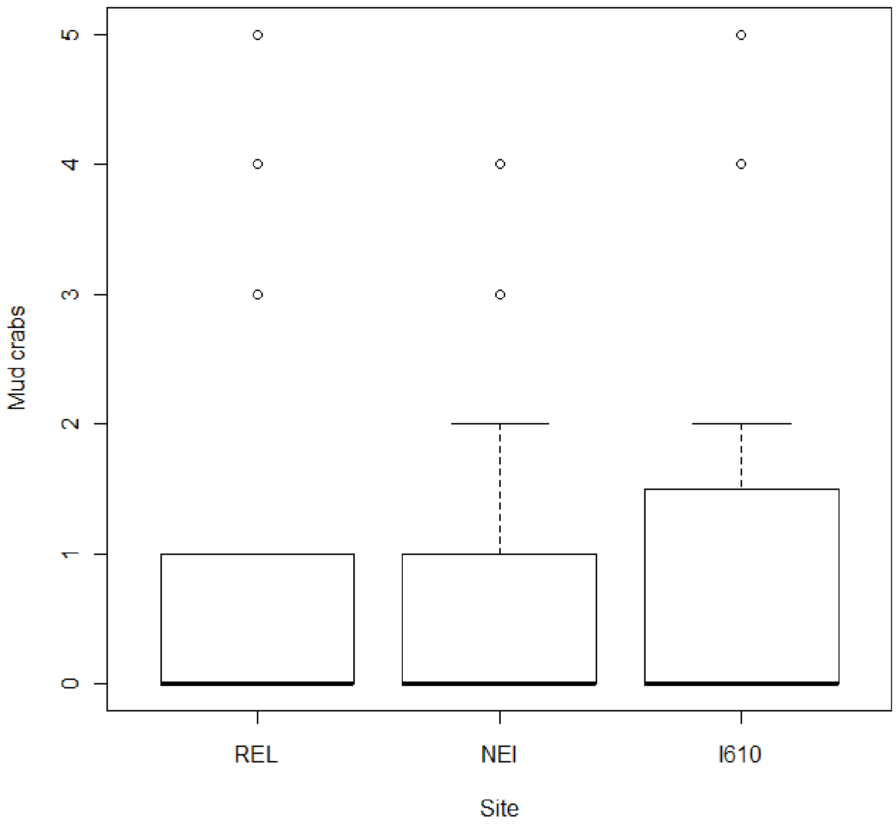

Figure 6: Box plot of mean, mode, and standard deviation in the number of *R. harrisii* collected per replicate for each site from May 2010 through May 2011, where I610 is Interstate 610, NEI is North End Island, and REL is Robert E. Lee. The mean and standard deviation for each site was REL: μ = 0.806 s.d. = 1.261; NEI: μ = 0.639, s.d. = 1.099; I610: μ = 0.9167 s.d. = 1.380. The y-axis represents the number of *R. harrisii* per replicate and the x-axis is the factor site.

Eleven species of fishes were sampled from September through December 2010 as a part of a shoreline seining survey (Table 3) and a significant difference in assemblage composition was exhibited among sites (ANOSIM, Global R = 0.266, p = 0.001). Pairwise tests between the sites indicated that each site was significantly different from every other site (REL vs. MIR, R = 0.493, p = 0.001; REL vs. DUM, R = 0.142, p = 0.02; DUM vs. MIR, R = 0.179, p = 0.014).

Table 3. Number of each species collected from seine sampling from September through December 2010 per site and overall. Triplicate samples were collected monthly as per the methods (n=36).

Species and Number collected by Seine Sept - Dec 2010

Species	REL	MIR	DUM
Menidia beryllina (inland silverside)	66	0	0
Fundulus grandis (Gulf killifish)	2	0	15
Lucania parva (rainwater killifish)	0	9	1
Gambusia affinis (western mosquitofish)	0	29	35
Cyprinodon variegatus (sheepshead minnow)	1	0	0
Lepomis macrochirus (bluegill)	88	213	194
Lepomis microlophus (redear sunfish)	0	0	1
Micropterus salmoides (largemouth bass)	2	7	6
Gobiosoma bosc (naked goby)	3	23	3
Syngnathus scovelli (Gulf pipefish)	4	2	3
Herichthys cyanoguttatus (Rio Grande cichlid)	1	0	9

Similarity percentage analysis was performed to see which of the eleven fish species were driving the dissimilarity pairwise among the sites (Table 4). The greatest contributor of dissimilarity for all pairwise tests was *L. macrochirus*. This species accounted greater than 50% of the average dissimilarity contributed to the overall dissimilarity between pairwise assemblage comparisons. It is also the most abundant species found at all sites for the duration of this study.

Table 4. Similarity Percentages (SIMPER) results for fish assemblages collected in the shoreline habitat from September through December 2010. Only the species that contributed up to 90% of the total assemblage change are shown. Values in the Contribution percentage column represent the average dissimilarity each species contributed to the overall dissimilarity between assemblages.

SIMPER Analysis Results

Robert E. Lee and Mirabeau

Species	REL Mean Abundance	Mirabeau Mean Abundance	Mean Dissimilarity	Contribution %
Lepomis macrochirus	7.33	17.75	49.35	59.66
Menidia beryllina	5.5	0	12.77	15.43
Gambusia affinis	0	2.42	7.23	8.74
Gobiosoma bosc	0.25	1.92	5.39	6.51

Robert E. Lee and Dumaine

Species	REL Mean Abundance	Dumaine Mean Abundance	Mean Dissimilarity	Contribution %
Lepomis macrochirus	7.33	16.17	36.77	62.58
Menidia beryllina	5.5	0	17.13	14.87
Gambusia affinis	0	2.92	8.81	7.44
Micropterus salmoides	0.17	0.5	4.35	4.3
Herichthys cyanoguttatus	0.08	0.75	3.61	3.99

Mirabeau and Dumaine

Species	Mirabeau Mean Abundance	Dumaine Mean Abundance	Mean Dissimilarity	Contribution %
Lepomis macrochirus	17.75	16.17	39.67	62.58
Gambusia affinis	2.42	2.92	9.42	14.87
Gobiosoma bosc	1.92	0.25	4.72	7.44
Lucania parva	0.75	0.08	2.79	4.3
Micropterus salmoides	0.58	0.52	2.53	3.99

Water Quality Modeling

Mean daily values for all abiotic data (based on measurements every 15 minutes) were calculated and plotted against time measured in days (Figs. 7, 8, 9, and 10). High daily mean temperature was on 12 September 2010 at 31.91°C and low daily mean temperature was on 27 December 2010 at 8.73°C (Fig. 7). Overall, daily mean temperature declined over the study period (Fig. 7). High daily mean salinity was on 31 December 2010 at 2.24, and low daily mean salinity was on 2 September 2010 at 1.54 (Fig. 8). Daily mean salinity increased over the study period (Fig. 8). High daily mean depth was on 3 November 2010 at 2.74 m, and low daily mean depth was on 12 October 2010 at 2.28 m (Fig. 9). There was no marked overall trend in mean depth over time (Fig. 9). High daily mean dissolved oxygen was on 14 December 2010 at 10.05 mg/L and low daily mean dissolved oxygen was on 25 October 2010 at 3.25 mg/L (Fig. 10). Mean daily dissolved oxygen appeared to increase over the study period (Fig, 10).

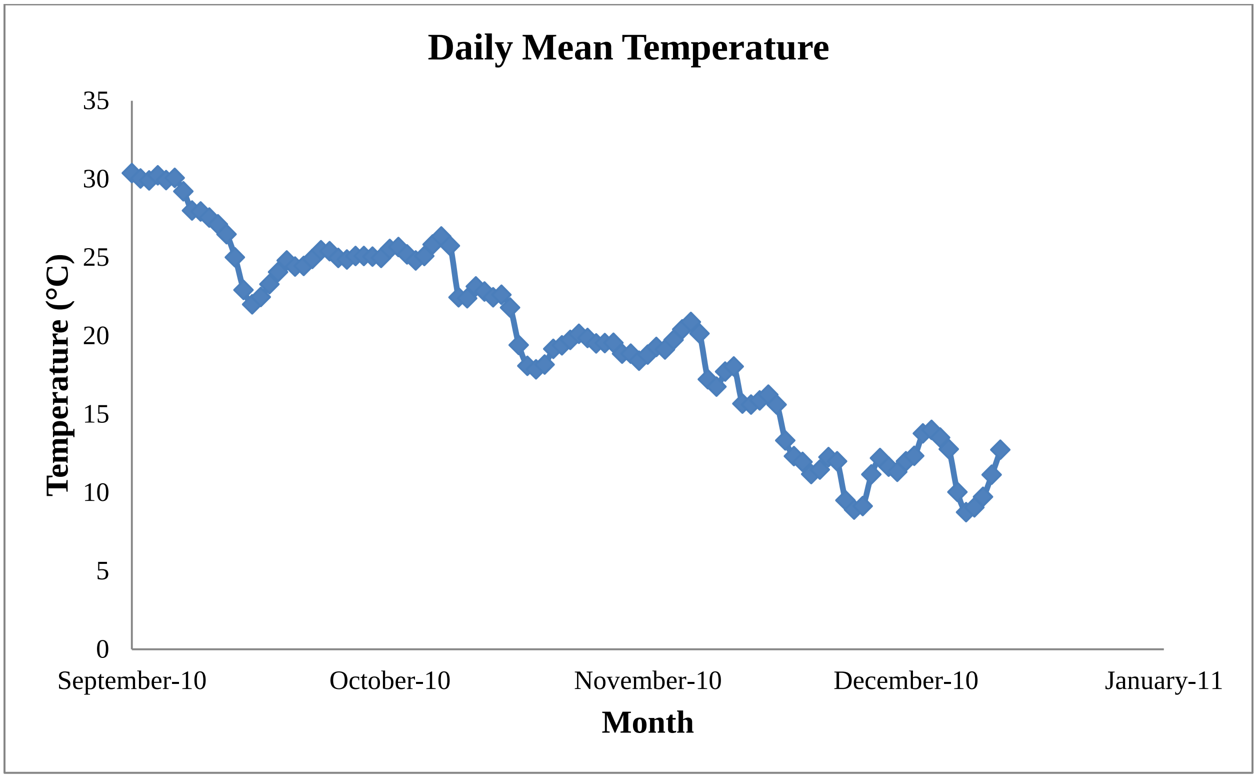

Figure 7: Daily mean temperature (°C) over the study period (1 September through 31 December 2011). Each point represents a calculated mean, with a line connecting the points to show the overall trend.

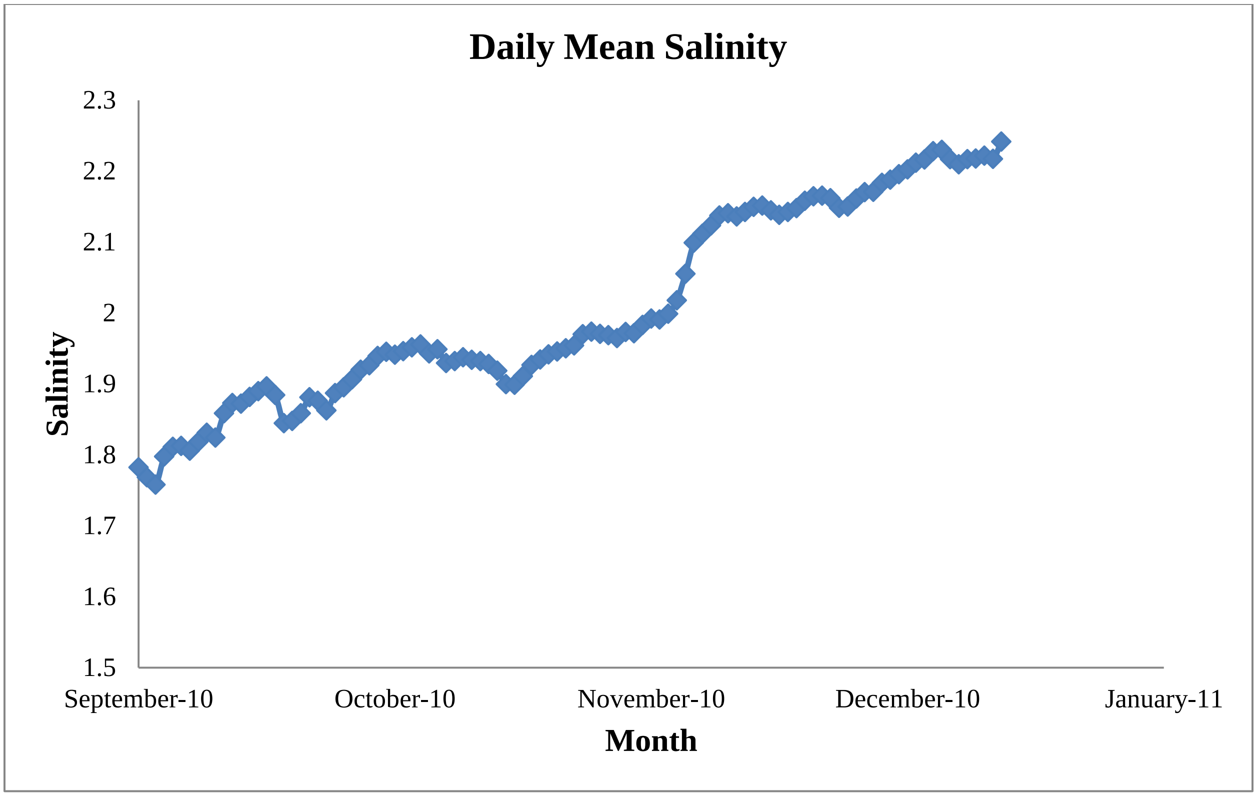

Figure 8: Daily mean salinity over the study period (1 September through 31 December 2011). Each point represents a calculated mean, with a line connecting the points to show the overall trend.

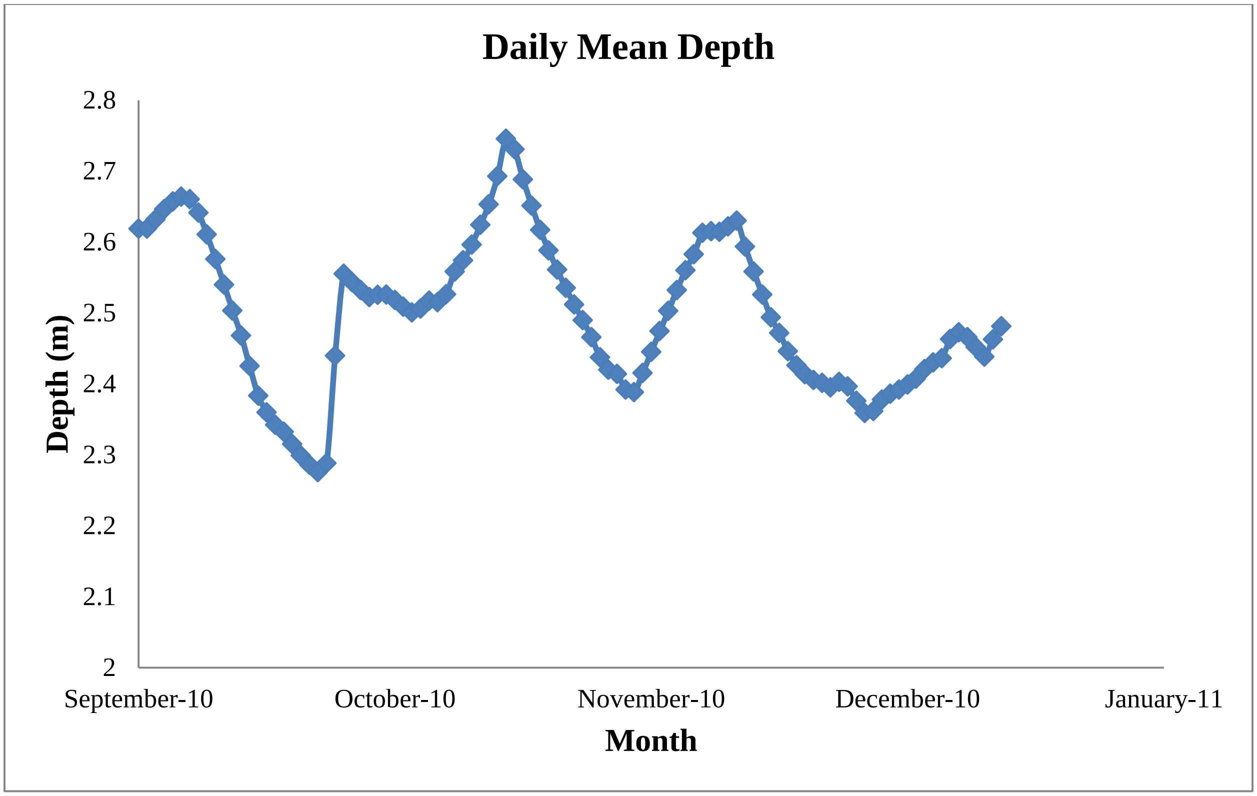

Figure 9: Daily mean depth (m) over the study period (1 September through 31 December 2011). Each point represents a calculated mean, with a line connecting the points to show the overall trend.

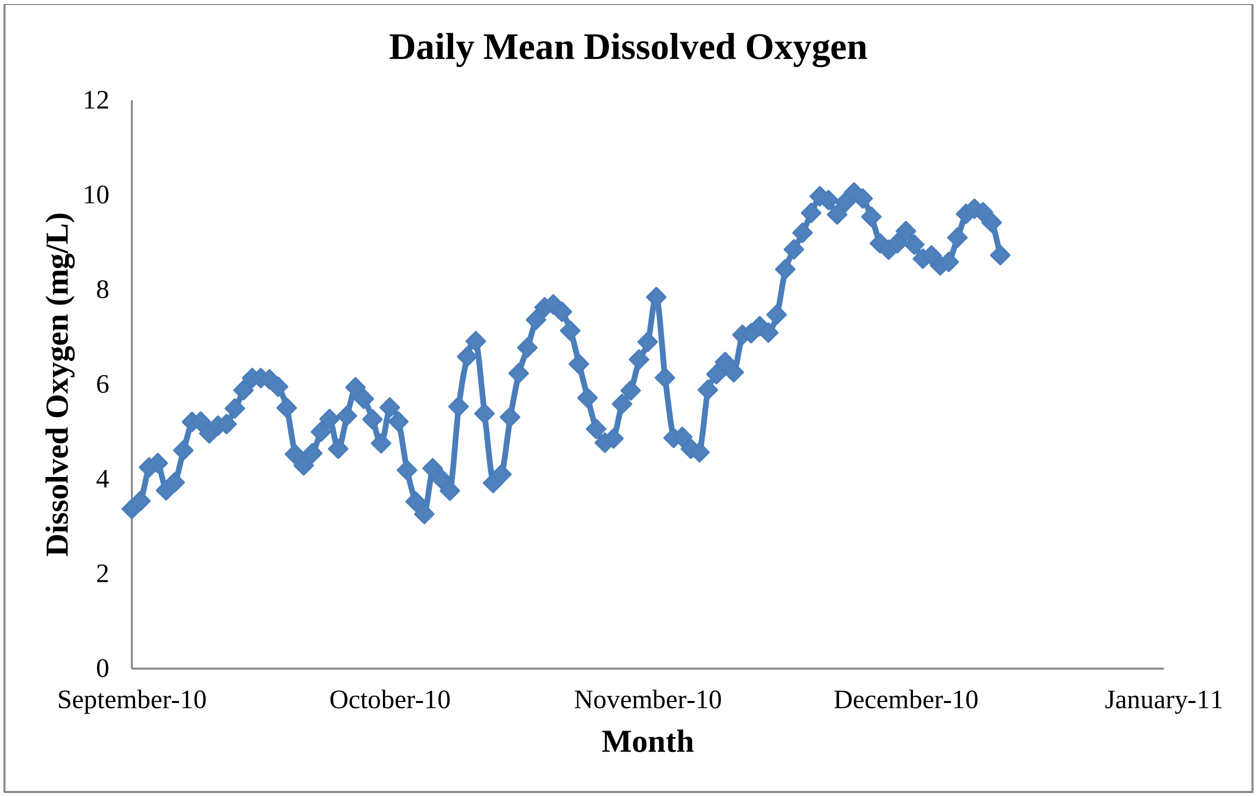

Figure 10: Daily mean dissolved oxygen (mg/L) over the study period (1 September through 31 December 2011). Each point represents a calculated mean, with a line connecting the points to show the overall trend.

Mathematical Models

The order of the relationship between each predictor variable and all four response variables was determined by a multistep process. Each ordinal relationship had to pass two tests. First, the predictor variable had to explain a statistically significant amount of variation in the response variable, and second, it had to explain significantly more variation in the response variable than other exponential values of predictor variables. For the number of pings per day at REL, a quadratic relationship was chosen for temperature and salinity ($p = 0.0001$, $p = 3.47\text{x}10^{-10}$; respectively; Table 5, Figs. 11 and 12), a straight line linear relationship was chosen for dissolved oxygen ($p = 0.01204$; Table 5, Fig. 13), and no relationship could be determined with respect to depth ($p = 0.085$, quadratic; Table 5, Fig. 14). For the number of pings per day at NEI, a quadratic relationship was chosen for temperature and dissolved oxygen ($p = 1.99\text{x}10^{-13}$, p = 0.00163; respectively; Table 6, Figs. 15 and 17), a cubic relationship fit best for salinity ($p = 1.14\text{x}10^{-8}$; Table 6, Fig. 16), and no relationship could be determined with respect to depth (p = 0.198, linear; Table 6, Fig. 12). No significant relationships between any abiotic variable and the number of pings per day at I610 could be determined (Table 7). For the total number of pings per day, a quadratic relationship was chosen for temperature and dissolved oxygen ($p = 2.98\text{x}10^{-10}$, p = 0.0123; respectively; Table 8, Figs. 18 and 20), a cubic relationship was chosen for salinity (p = 0.00914; Table 8, Fig. 119), and no relationship could be determined with respect to depth (p = 0.0635, quadratic; Table 8).

Table 5. Results from multiple regressions models comparing the number of pings per day at Site Robert E. Lee versus all four predictor variables (Temperature, Salinity, Dissolved Oxygen, and Depth). Rows labeled "GLT" indicate the p-value associated between two models, one with the variable noted, one included only the next lowest variable by the general linear test. Each section highlighted in gray represents the best fitting relationship for each variable.

REL Polynomial Test Results				
WQ Variable		Linear	Quad	Cubic
Temp	p-value	0.0282	5.60E-05	1.49E-04
	R^2	0.03149	0.1375	0.1359
	GLT		1.27E-04	0.3789226
Salinity	p-value	0.01204	1.03E-10	4.59E-10
	R^2	0.04349	0.3091	0.307
	GLT		3.47E-10	4.24E-01
DO	p-value	0.00261	4.16E-03	1.12E-02
	R^2	0.06532	0.07269	0.06623
	GLT		0.1662	0.6754
Depth	p-value	0.6237	0.08532	0.1535
	R^2	-0.006304	0.0244	0.01912
	GLT		0.03127	0.54966

Table 6. Results from multiple regressions modeling comparing the number of pings per day at Site North End Island versus all four predictor variables (Temperature, Salinity, Dissolved Oxygen, and Depth). Rows labeled "GLT" indicate the p-value associated between two models, one with the variable noted, one included only the next lowest variable by the general linear test. Each section highlighted in gray represents the best fitting relationship for each variable.

NEI Polynomial Test Results

WQ Variable		Linear	Quad	Cubic
Temp	p-value	0.327	9.06E-13	4.80E-12
	R^2	-0.0002597	0.362	0.3593
	GLT		1.99E-13	0.4762
Salinity	p-value	0.1062	7.70E-09	4.15E-15
	R^2	0.01345	0.2572	0.4323
	GLT		4.28E-11	1.14E-08
DO	p-value	0.09098	1.58E-03	4.95E-03
	r^2	0.01549	0.08765	0.08005
	GLT		0.001627	0.894456
Depth	p-value	0.1978	0.2843	0.4654
	R^2	0.005563	0.004463	-0.003551
	GLT		0.3555	0.8241

Table 7. Results from multiple regressions modeling comparing the number of pings per day at Site I610 versus all four predictor variables (Temperature, Salinity, Dissolved Oxygen, and Depth). Rows labeled "GLT" indicate the p-value associated between two models, one with the variable noted, one included only the next lowest variable by the general linear test. Each section highlighted in gray represents the best fitting relationship for each variable.

I-610 Polynomial Test Results

WQ Variable		Linear	Quad	Cubic
Temp	p-value	0.9955	1.57E-01	1.79E-01
	R^2	-0.008333	0.01434	0.01615
	GLT		5.47E-02	0.2718
Salinity	p-value	0.7324	1.24E-01	0.1288
	R^2	-0.007347	0.0182	0.02249
	GLT		4.41E-02	0.21971
DO	p-value	0.7886	8.76E-01	6.13E-01
	R^2	-0.007727	-0.01455	-0.009887
	GLT		0.6605	0.2157
Depth	p-value	0.4254	0.3602	0.5536
	R^2	-0.002986	0.0004958	-0.007483
	GLT		0.238	0.8109

Table 8. Results from multiple regression models comparing the number of pings per day at from all sites versus all four predictor variables (Temperature, Salinity, Dissolved Oxygen, and Depth). Rows labeled “GLT” indicate the p-value associated between two models, one with the variable noted, one included only the next lowest variable by the general linear test. Each section highlighted in gray represents the best fitting relationship for each variable.

Total Polynomial Test Results

WQ Variable		Linear	Quad	Cubic
Temp	p-value	0.1632	7.44E-10	3.71E-09
	R^2	0.007936	0.2858	0.2816
	anova		2.98E-10	0.5804
Salinity	p-value	0.1684	5.80E-15	< 2.2e-16
	R^2	0.007548	0.4139	0.4617
	anova		< 2.2e-16	9.14E-04
DO	p-value	0.001341	2.50E-04	8.60E-04
	R^2	0.07482	0.1155	0.1085
	anova		0.01228	0.79237
Depth	p-value	0.3382	0.06347	0.1193
	R^2	-0.0006229	0.02924	0.02395
	anova		0.0328	0.5523

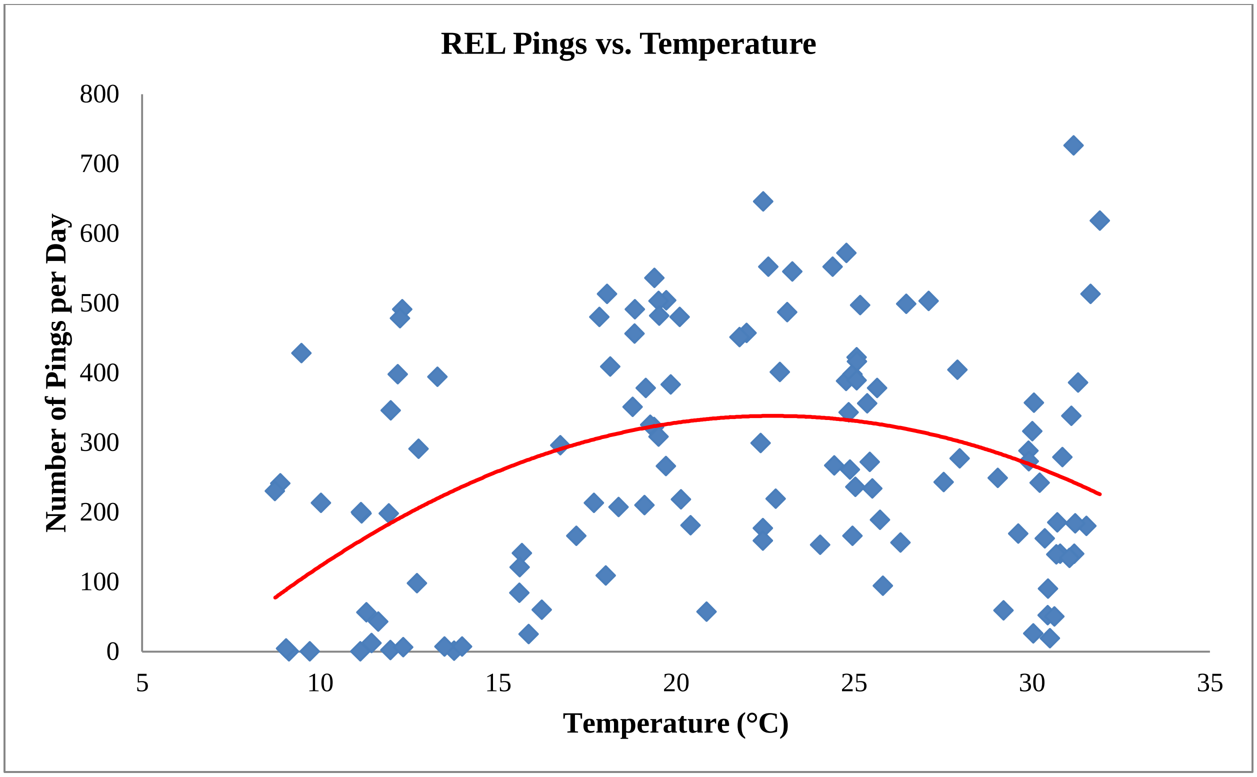

Figure 11: Relationship between the number of pings per day at Site Robert E. Lee (y-axis) and the daily mean temperature value in °C (x-axis) as a scatter plot with line of best fit. The line of best fit generated from multiple regression models indicates a quadratic relationship (p = 5.06x10^{-5}).

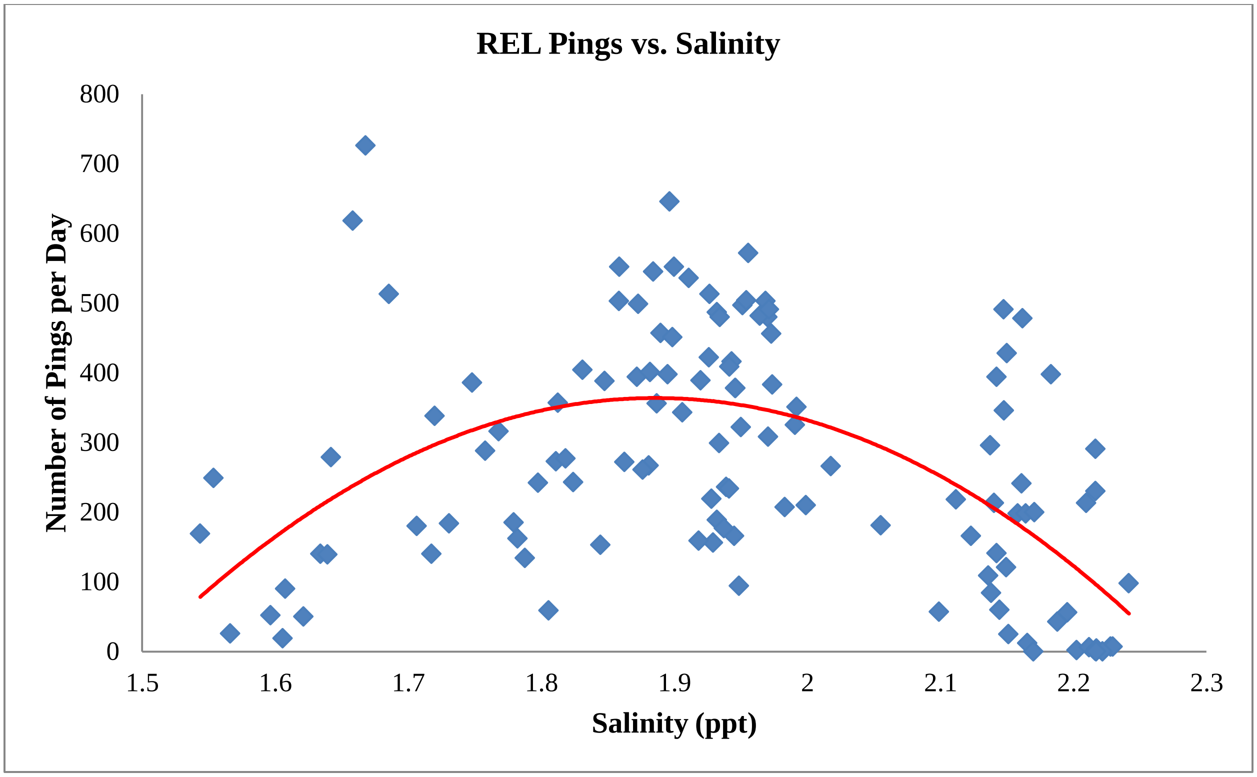

Figure 12: Relationship between the number of pings per day at Site Robert E. Lee (y-axis) and the daily mean salinity (x-axis) as a scatter plot with line of best fit. . The line of best fit generated from multiple regression models indicates a quadratic relationship (p = 1.03x10^{-10}).

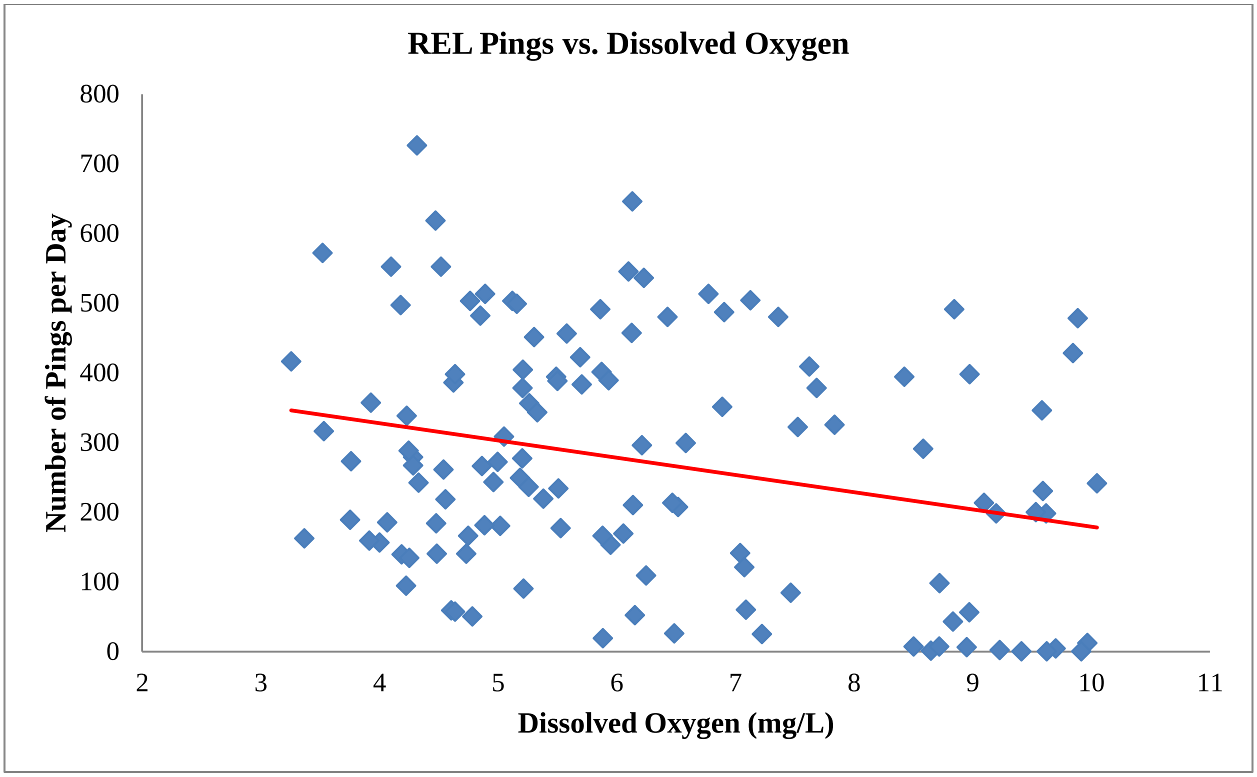

Figure 13: Relationship between the number of pings per day at Site Robert E. Lee (y-axis) and the daily mean dissolved oxygen value (mg/L; x-axis) as a scatter plot with line of best fit. The line of best fit generated from multiple regression models indicates a linear relationship ($p = 0.00261$).

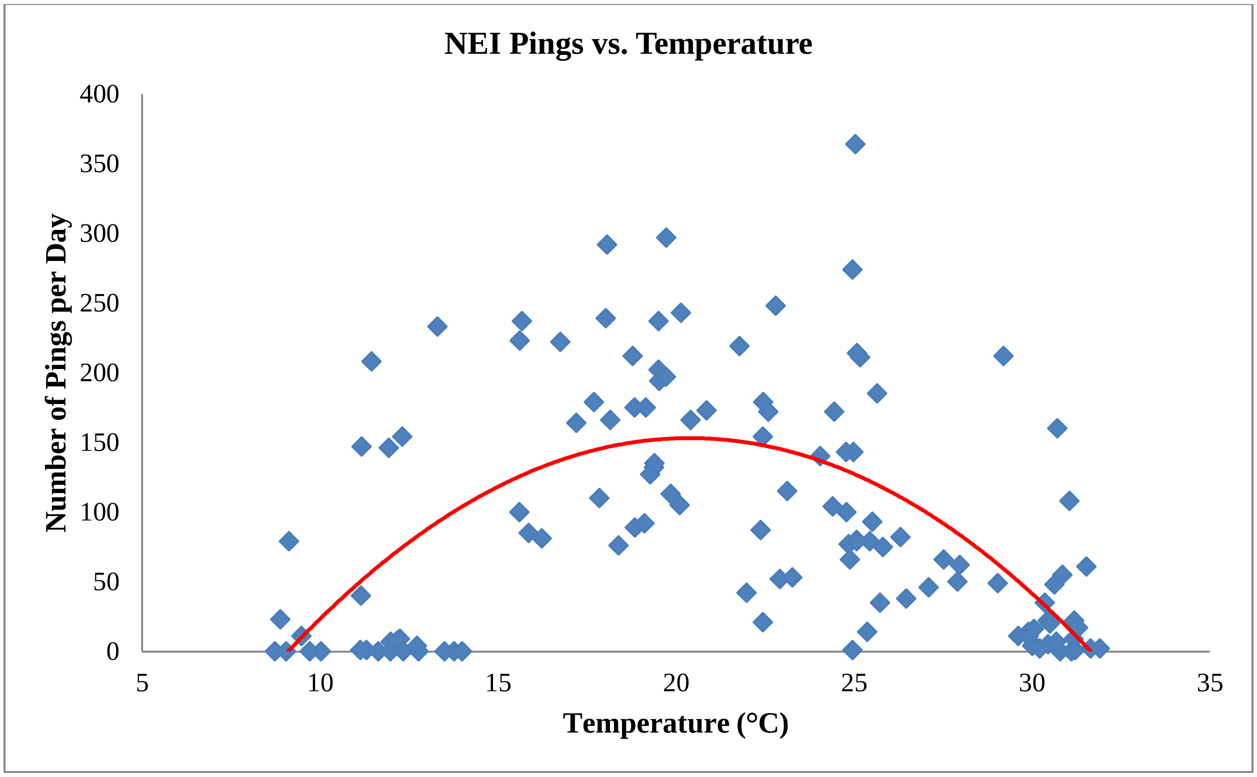

Figure 14: Relationship between the number of pings per day at North End Island (y-axis) and the daily mean temperature value (°C; x-axis) as a scatter plot with line of best fit. The line of best fit generated from multiple regression models indicates a quadratic relationship ($p = 9.06\text{x}10^{-13}$).

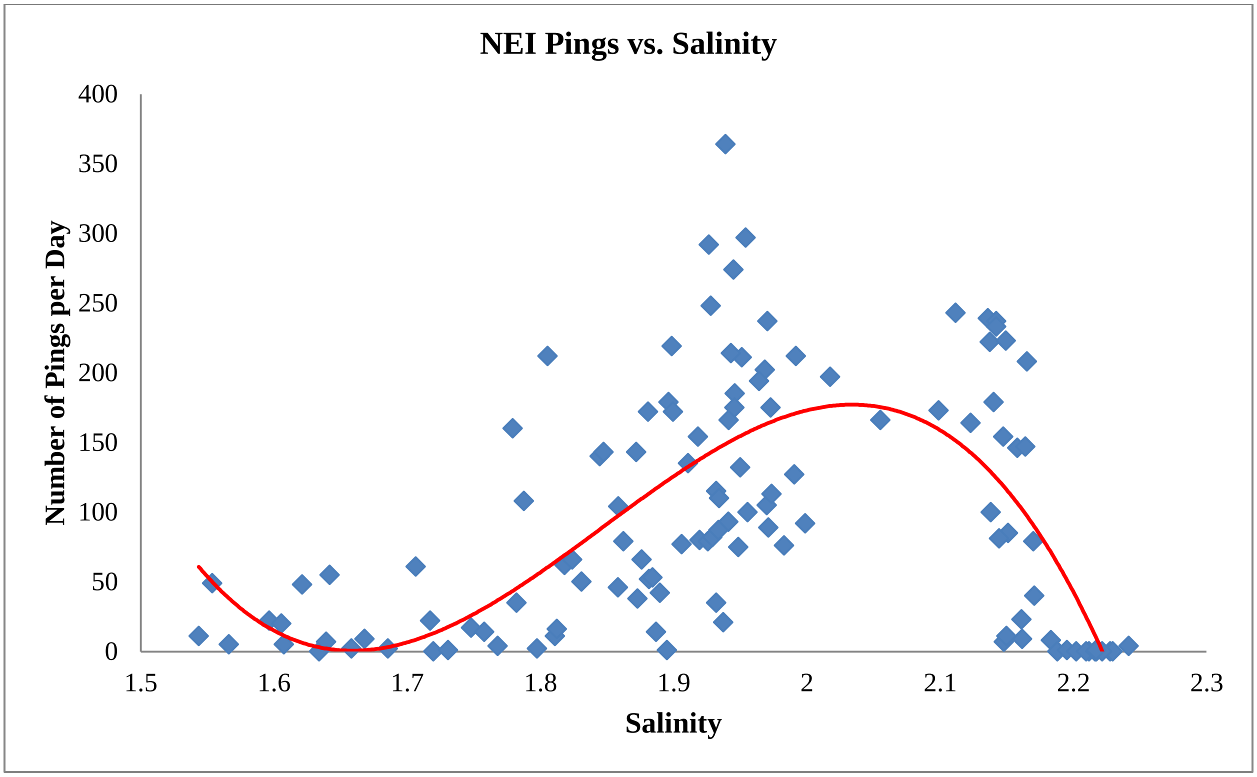

Figure 15: Relationship between the number of pings per day at North End Island (y-axis) and the daily mean salinity value (x-axis) as a scatter plot with line of best fit. The line of best fit generated from multiple regression models indicates a cubic relationship ($p = 5.15\text{x}10^{-15}$).

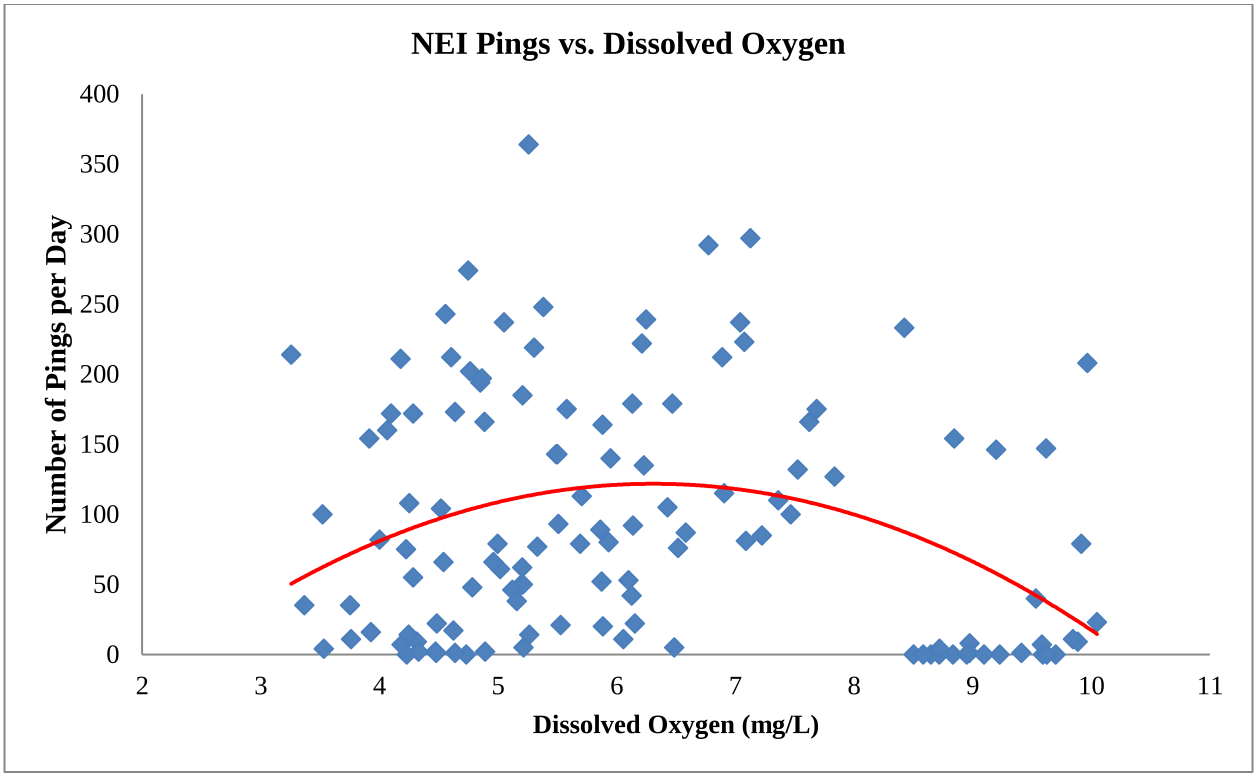

Figure 16: Relationship between the number of pings per day at North End Island (y-axis) and the daily mean dissolved oxygen value (mg/L; x-axis) as a scatter plot with line of best fit. The line of best fit generated from multiple regression models indicates a quadratic relationship (*p* = 0.00158).

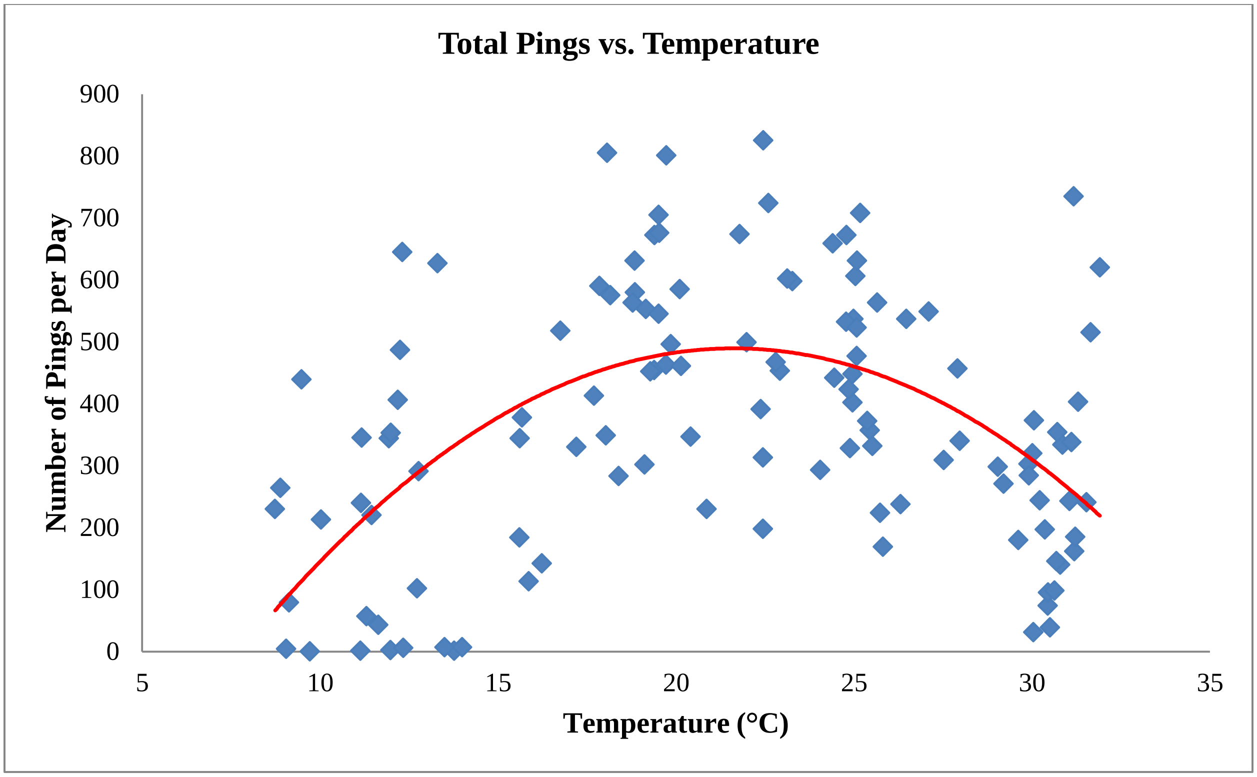

Figure 17: Relationship between the total number of pings per day (y-axis) and the daily mean temperature value (°C; x-axis) as a scatter plot with line of best fit. The line of best fit generated from multiple regression models indicates a quadratic relationship ($p = 7.44\text{x}10^{-10}$).

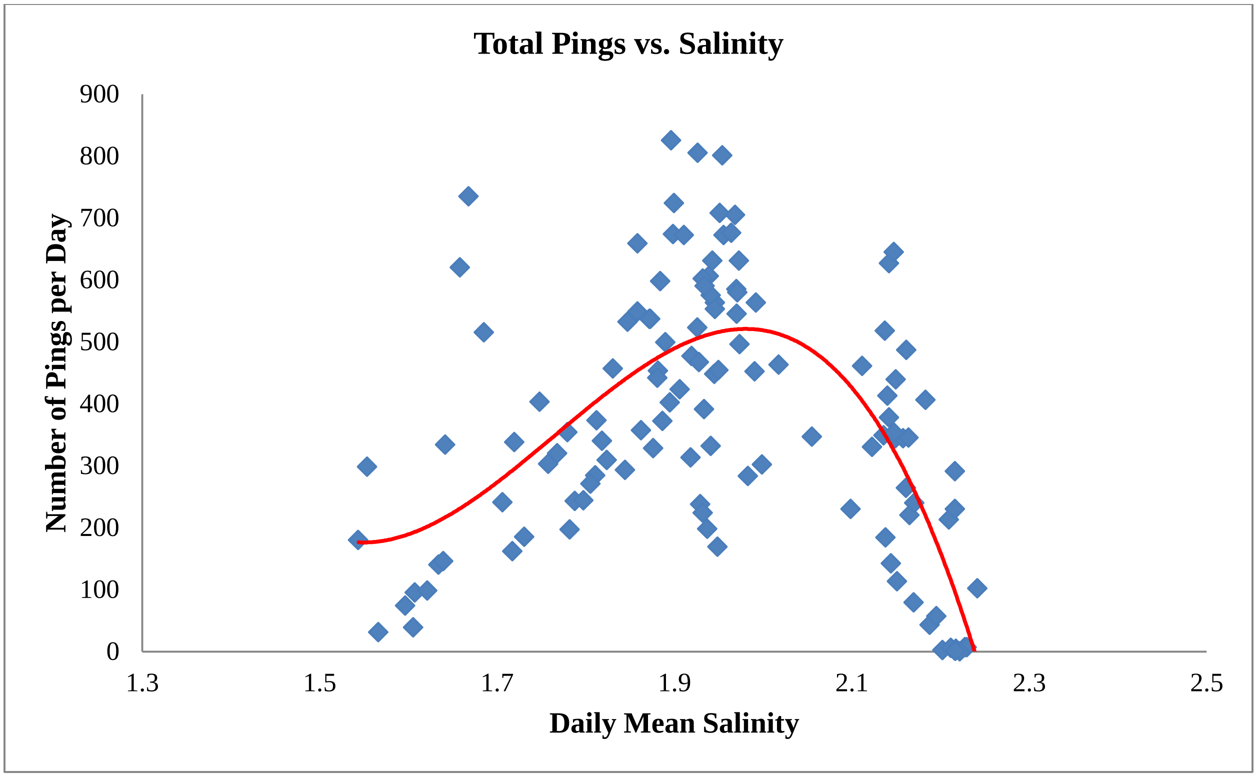

Figure 18: Relationship between the total number of pings per day (y-axis) and the daily mean salinity value (x-axis) as a scatter plot with line of best fit. The line of best fit generated from multiple regression models indicates a cubic relationship ($p < 2.2x10^{-16}$).

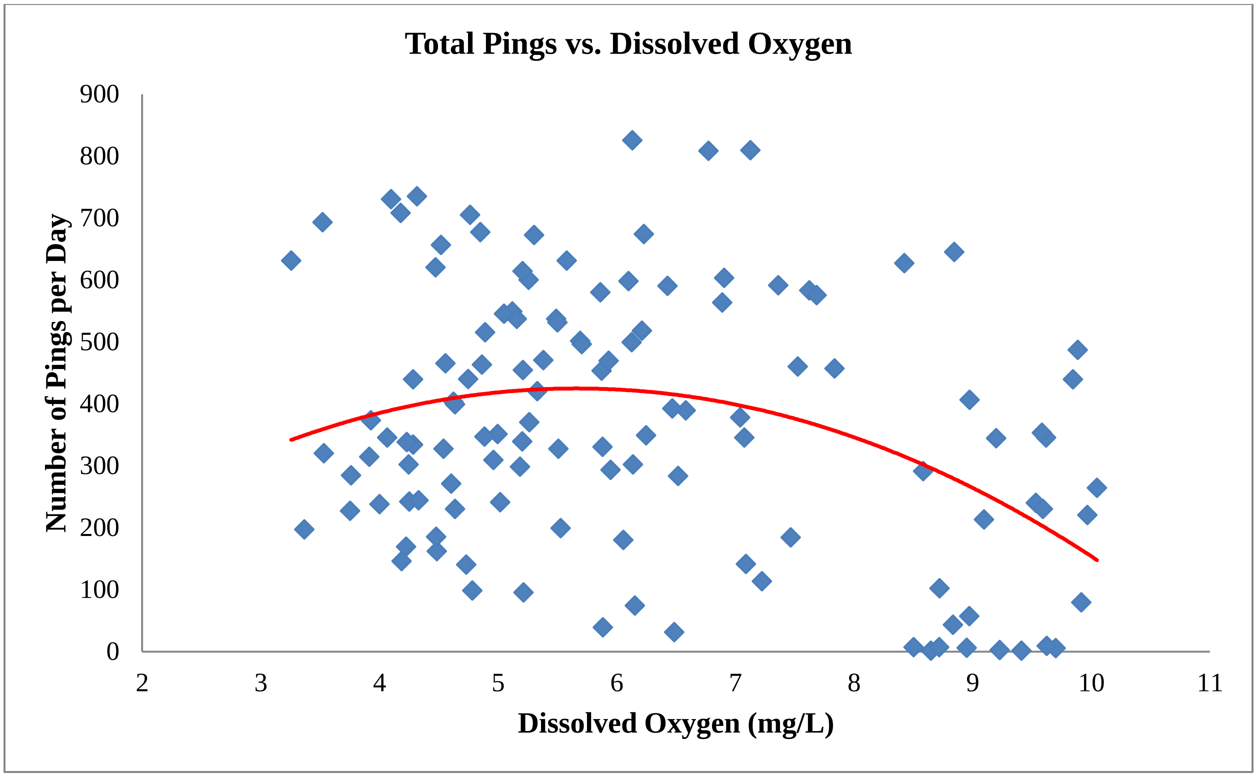

Figure 19: Relationship between the total number of pings per day (y-axis) and the daily mean dissolved oxygen value (mg/L; x-axis) as a scatter plot with line of best fit. The line of best fit generated from multiple regression models indicates a cubic relationship (*p* = 0.00025).

Mathematical GEE predictor variables were selected based on the above analysis. That is, the polynomial order for each predictor variable response variable pairing was selected based upon the above criteria. Then, three GEEs step-wise model reductions were performed. Similar to the multiple regression analyses above, no model reduction for I610 could be done because these data were unable to be transformed to fit any distributional pattern. This may have been due to low number of pings throughout the study, 95 of 122 days (78%) during this study 0 pings were detected at I610 (Fig. 20).

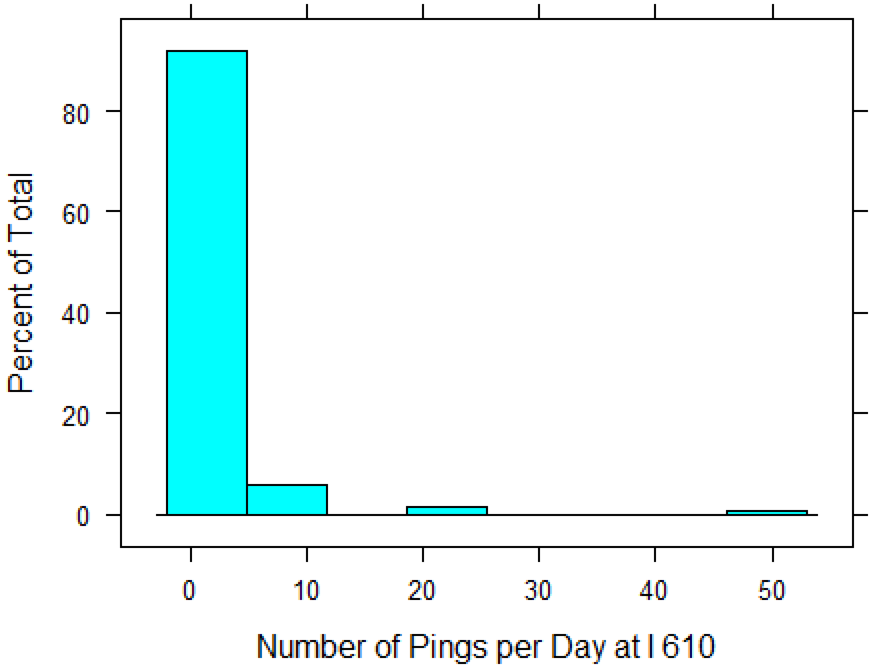

Figure 20: Frequency histogram depicting the number of pings detected per day as a percent total. This distribution shows a high frequency of low numbers, with 78% of all days having 0 detections.

REL's Mathematical GEE correlation structure was exchangeable, also termed compound symmetry correlation structure, and Gaussian distribution of errors. Before reduction, the model was $y = \beta_1T + \beta_2T^2 + \beta_3S + \beta_4S^2 + \beta_5D + \beta_0 + \varepsilon$, where y = number of pings per day at Robert E. Lee, T = daily mean temperature value, S = daily mean salinity value, and D = daily mean dissolved oxygen value. Variables were reduced stepwise, with higher order polynomials of each predictor being reduced first. The resulting model was reduced to $y = \beta_1T + \beta_2S + \beta_3S^2 + \beta_0 + \varepsilon$. (W = 4.5, $p = 0.0000054$; Table 9). The result included temperature as a first order polynomial and salinity as a second order polynomial as significant predictor variables.

Table 9. Table showing the results of a generalized estimating equation (GEE) step-wise model reduction of the original model that included: $y = \beta_1 T + \beta_2 T^2 + \beta_3 S + \beta_4 S^2 + \beta_5 D + \beta_0 + \varepsilon$, where y = number of pings per day at Robert E. Lee, T = daily mean temperature value, S = daily mean salinity value, and D = daily mean dissolved oxygen value. The model was reduced to $y = \beta_1 T + \beta_2 S + \beta_3 S^2 + \beta_0 + \varepsilon$.

GEE stepwise reduction model REL

Distribution = Gaussian, Correlation = compound symmetry

	Estimate	Standard Error	Wald Statistic	p-value
Intercept	-8647.9	1235.5	49	2.60E-12
Temperature	-20.5	4.5	20.7	5.40E-06
Salinity	10760.9	1307.2	67.8	2.20E-16
Salinity^2	-3030	345.4	77	< 2E-16

The Mathematical GEE for NEI correlation structure was exchangeable and Gaussian distribution of errors. Before reduction, the model was $y = \beta_1 T + \beta_2 T^2 + \beta_3 S + \beta_4 S^2 + \beta_5 S^3 + \beta_6 D + \beta_7 D^2 + \beta_0 + \varepsilon$, where y = number of pings per day at North End Island, T = daily mean temperature value, S = daily mean salinity value, and D = daily mean dissolved oxygen value. Variables were reduced stepwise, with higher order polynomials of each predictor being reduced first. The resulting model was reduced to $y = \beta_1 S + \beta_2 S^2 + \beta_3 S^3 + \beta_0 + \varepsilon$. (W statistic = 34.2, $p < 2 \times 10^{-16}$, Table 10). Results included salinity as a third order polynomial as the significant predictor variables.

Table 10. Results of a generalized estimating equation (GEE) step-wise model reduction of the original model that included: $y = \beta_1 T + \beta_2 T^2 + \beta_3 S + \beta_4 S^2 + \beta_5 S^3 + \beta_6 D + \beta_7 D^2 + \beta_0 + \varepsilon$, where y = number of pings per day at North End Island, T = daily mean temperature value, S = daily mean salinity value, and D = daily mean dissolved oxygen value. The model was reduced to $y = \beta_1 S + \beta_2 S^2 + \beta_3 S^3 + \beta_0 + \varepsilon$.

GEE stepwise reduction model NEI

Distribution = Poisson, Correlation = compound symmetry

	Estimate	Standard Error	Wald Statistic	p-value
Intercept	541.2	102	28.2	2.60E-12
Salinity	-895	163.4	30	5.40E-06
Salinity^2	491.6	86.7	32.2	2.20E-16
Salinity^3	-89	15.2	34.2	< 2E-16

Total's (the total number of pings per day, including all sites) Mathematical GEE correlation structure was exchangeable and Gaussian distribution of errors. Before reduction, the model was $y = \beta_1 T + \beta_2 T^2 + \beta_3 S + \beta_4 S^2 + \beta_5 S^3 + \beta_6 D + \beta_7 D^2 + \beta_0 + \varepsilon$, where y = total number of pings per day, T = daily mean temperature value, S = daily mean salinity value, and D = daily mean dissolved oxygen value. Variables were reduced stepwise, with higher order polynomials of each predictor being reduced first. The resulting model was reduced to $y = \beta_1 T + \beta_2 T^2 + \beta_3 S + \beta_4 S^2 + \beta_0 + \varepsilon$. (W statistic = 5.12, $p = 0.024$; Table 11). Results included temperature as a second order polynomial and salinity as a second order polynomial as predictor variables.

Table 11. Results of a generalized estimating equation (GEE) step-wise model reduction of the original model that included: $y = \beta_1 T + \beta_2 T^2 + \beta_3 S + \beta_4 S^2 + \beta_5 S^3 + \beta_6 D + \beta_7 D^2 + \beta_0 + \varepsilon$, where y = the total number of pings per day for all sites, T = daily mean temperature value, S = daily mean salinity value, and D = daily mean dissolved oxygen value. The model was reduced to $y = \beta_1 T + \beta_2 T^2 + \beta_3 S + \beta_4 S^2 + \beta_0 + \varepsilon$.

GEE stepwise reduction model Total

Distribution = Gaussian, Correlation = compound symmetry

	Estimate	Standard Error	Wald Statistic	p-value
Intercept	-10900	1400	60.68	6.70E-15
Temperature	11.9	16.6	0.52	4.72E-01
Temperature^2	-0.838	-0.37	5.12	2.40E-02
Salinity	13000	1600	66.13	4.4E-16
Salinity^2	-3620	43	70.8	< 2E-16

Practical Models

REL's Practical GEE was exchangeable and Gaussian distribution of errors. Before reduction, the model was $y = \beta_1 T + \beta_2 T^2 + \beta_3 S + \beta_4 D + \beta_0 + \varepsilon$, where y = number of pings per day at REL, T = daily mean temperature value, S = daily mean salinity value, and D = daily mean depth value. Variables were reduced stepwise, with higher order polynomials of each predictor being reduced first. The resulting model was reduced to $y = \beta_1 T + \beta_2 T^2 + \beta_3 S + \beta_0 + \varepsilon$. (W = 5.24, $p = 0.02212$; Table 12). The result included temperature as a second order polynomial and salinity as a first order polynomial as significant predictor variables. The straight line relationship between salinity and the number of pings per day at REL is a negative correlation (Fig. 21).

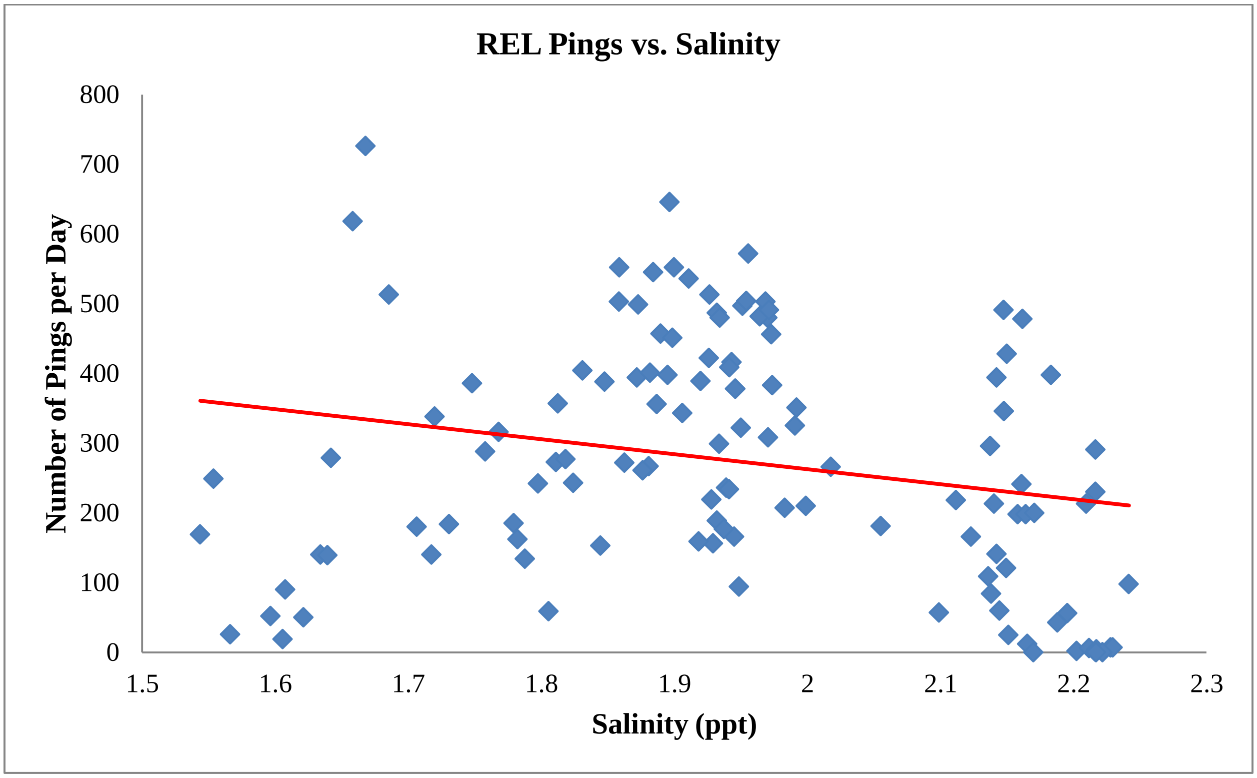

Figure 21: Graph showing the relationship between the number of pings per day at the Robert E. Lee site (y-axis) and the daily mean salinity value (x-axis) as a scatter plot. The fitted line (in red) shows the first-order polynomial line of best fit generated from a regression model.

Table 12. This table shows the results of a generalized estimating equation (GEE) step-wise model reduction of the original model that included: $y = \beta_1 T + \beta_2 T^2 + \beta_3 S + \beta_4 D + \beta_0 + \varepsilon$, where y = the number of pings per day at site REL, T = daily mean temperature value, S = daily mean salinity value, and D = daily mean depth value. The model was reduced to $y = \beta_1 T + \beta_2 T^2 + \beta_3 S + \beta_0 + \varepsilon$.

GEE Model Selection Results REL

Distribution = Gaussian, Correlation = compound symmetry

	Estimate	Standard Error	Wald Statistic	p-value
Intercept	945.24	607.06	2.42	0.11945
Temperature	56.39	14.6	14.92	0.00011
Temperature^2	-1.56	0.33	22.33	2.30E-06
Salinity	-558.02	243.97	5.24	0.02212

The Practical GEE for NEI was exchangeable and Poisson distribution of errors. Before reduction, the model was $y = \beta_1 T + \beta_2 T^2 + \beta_3 S + \beta_4 D + \beta_0 + \varepsilon$, where y = number of pings per day at NEI, T = daily mean temperature value, S = daily mean salinity value, and D = daily mean depth value. Variables were reduced stepwise, with higher order polynomials of each predictor being reduced first. The resulting model was reduced to $y = \beta_1 T + \beta_2 T^2 + \beta_3 D + \beta_0 + \varepsilon$. (W statistic = 6.81, *p* = 0.009, Table 18). Results included salinity as a third order polynomial as the significant predictor variables. The straight line relationship between depth and the number of pings per day at REL is a positive correlation (Fig. 22).

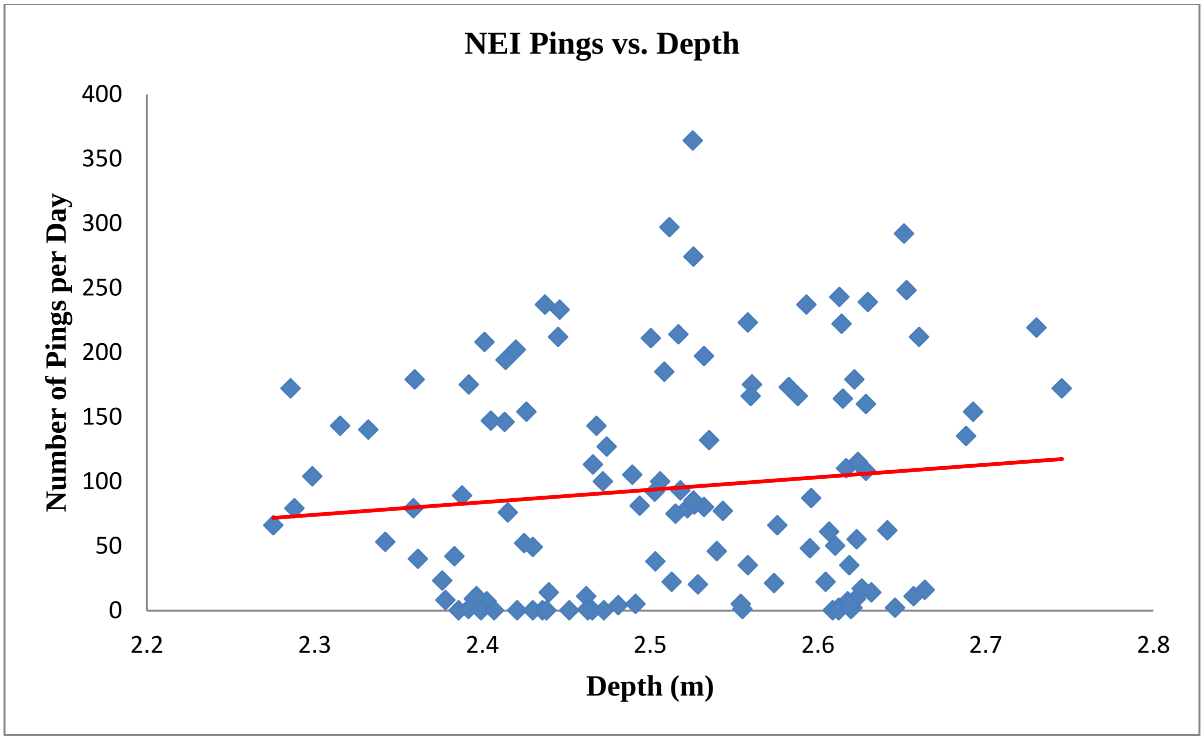

Figure 22: Graph showing the relationship between the number of pings per day at North End Island site (y-axis) and the daily mean depth in meters (x-axis) as a scatter plot. The fitted line (in red) shows the first-order polynomial line of best fit generated from a regression model.

Table 13. Results of a generalized estimating equation (GEE) step-wise model reduction of the original model that included: $y = \beta_1 T + \beta_2 T^2 + \beta_3 S + \beta_4 D + \beta_0 + \varepsilon$, where y = the number of pings per day at site NEI, T = daily mean temperature value, S = daily mean salinity value and D = daily mean depth value. The model was reduced to $y = \beta_1 T + \beta_2 T^2 + \beta_3 D + \beta_0 + \varepsilon$.

GEE Model Selection Results NEI

Distribution = Poisson, Correlation = compound symmetry

	Estimate	Standard Error	Wald Statistic	p-value
Intercept	-4.58497	1.61866	8.02	0.0046
Temperature	0.65758	0.10633	38.24	6.20E-10
Temperature^2	-0.01631	0.00241	45.82	2.30E-06
Depth	1.22792	0.47041	6.81	0.009

Total's (the total number of pings per day, including all sites) Practical GEE was exchangeable and Gaussian distribution of errors. Before reduction, the model was $y = \beta_1 T + \beta_2 T^2 + \beta_3 S + \beta_4 D + \beta_0 + \varepsilon$, where y = total number of pings per day, T = daily mean temperature value, S = daily mean salinity value, and D = daily mean depth value. Variables were reduced stepwise, with higher order polynomials of each variable being reduced first. The resulting model was reduced to $y = \beta_1 T + \beta_2 T^2 + \beta_0 + \varepsilon$. (W statistic = 47, $p = 7.10\text{x}10^{-12}$; Table 14). Results included temperature as a second order polynomial as predictor variables.

Table 14. Results of a generalized estimating equation (GEE) step-wise model reduction of the original model that included: $y = \beta_1 T + \beta_2 T^2 + \beta_3 S + \beta_4 D + \beta_0 + \varepsilon$, where y = the total number of pings per day from all sites, T = daily mean temperature value, S = daily mean salinity value, and D = daily mean depth value. The model was reduced to $y = \beta_1 T + \beta_2 T^2 + \beta_0 + \varepsilon$.

GEE Model Selection Results Total				
Distribution = Gaussian, Correlation = compound symmetry				
	Estimate	Standard Error	Wald Statistic	p-value
Intercept	-704.133	152.342	21.4	3.80E-06
Temperature	110.585	0.10633	50.8	1.00E-12
Temperature^2	-2.559	0.373	47	7.10E-12

Discussion

Prey Availability and Red Drum Location

My results on the number of pings per day (indicating the presence of a tagged red drum) support the previous findings with the highest number of pings occurring in the northernmost site (Brogan, 2010). The average number of pings for the most southern site (I610) was markedly low (1.43 ± 5.52), which also agrees with the previous study (Brogan, 2010). The results from both studies suggest that red drum are avoiding areas south of I610.

The possibility that red drum can pass a receiver without detection is low. The maximum overall width of BSJ is 200 m. Based upon expected detection radius for the receivers, tagged red drum cannot swim throughout the Bayou without passing within the range of detection for the receiver transmitter combination (between 300-540 m, depending on conditions). The transmitters were designed to send a ping every 180 seconds on average (Brogan, 2010). With this interval, a tagged fish could potentially pass through a receiver's range of detection without the transmitter sending a signal. However, there were no instances where a red drum was detected at REL and then detected at I610, or vice versa. Additionally, the middle receiver (NEI) is near the widest point in the Bayou. This suggests that a red drum cannot easily travel through a receiver's detection radius without being recorded.

Of the twelve organisms sampled in both the shoreline and the epibenthic surveys, five have been observed as stomach contents for large (> 300 mm) juvenile red drum in the literature: *Menidia beryllina* (inland silverside), *Cyprinodon variegatus* (sheepshead minnow), *Gobiosoma bosc* (naked goby), *Fundulus grandis* (Gulf killifish), and *R. harrisii* (Boothby and Avault Jr., 1971; Overstreet and Heard, 1978). Across studies, red drum were found to ingest abundant crab species, with Xanthid crabs being particularly important in impoundments (Llanso *et al.*, 1998; Matlock, 1987). Teleost fishes were not found to be as important a food item as crabs (Boothby and Avault Jr., 1971; Overstreet and Heard, 1978; Llanso et la., 1998). The only stomach content observed in a red drum from BSJ was a *C. sapidus* (Brogan, 2010). Additionally, most research suggests blue crabs are the primary prey item for red drum (Guillory and Prejean, 1999). No blue crabs were ever sampled at any of the sites during this period. Without more knowledge of the actual diet of BSJ red drum, it is difficult to determine which prey items this species prefers.

If the abundance of potential prey items are an important reason why the southern portion of BSJ is underutilized, differences in prey items would have been observed across broad temporal periods because red drum have been found in the northern sites in BSJ across all methods and studies. On average, more organisms were observed at the northernmost site than the other sites from September through December 2010. At first glance, this may lead to the conclusion that higher abundances occur at the same site in which higher numbers of pings per day do. However, the standard deviation for each site is greater than or equal to the mean for each site, suggesting that means are still low overall and the dataset is zero-inflated. Additionally, when analyzing a much larger dataset (May 2010 – May 2011), no statistically significant difference was observed for any pairwise combination. Since previous studies show a similar relationship for the total number of pings per site and the larger dataset does not reveal a statistically significant difference among sites, the correlation between the number of pings per site and higher abundances of *R. harrisii* seen during September through December 2010 may be a statistical artifact that does not reflect actual relationships. Possibly, the apparent relationship may be due to low sample size (n = 27). Selection of habitat based up prey items could not be inferred using data from the benthic survey.

One of the issues concerning analysis of the shoreline assemblages is the lack of overlap between receiver site I610 and seining site at Dumaine Bridge. The number of pings per day at I610 was low, with the vast majority of the days having zero pings recorded. This ultimately is more problematic than the lack of overlap between sampling sites, because of a heavily zero-weighted dataset. Therefore, the only conclusion that can come from analysis of pings per day at I610 is they are low to the point of almost complete avoidance. Therefore any assemblage difference at Dumaine Bridge could be considered a surrogate for habitats with extremely low red drum occurrences.

Lower abundances of all organisms, except one, that contributed to assemblage differences between pairwise site tests were observed at REL. Only *M. beryllina* was observed in higher abundances at REL and it was not collected at the other sites. However, these fishes have been sampled at both of the other sites, outside of the study period (see Chapter 2). Even though *M. beryllina* were only sampled at the site in which the most pings per day were observed, and these fishes have been shown to be a part of red drum diet from other studies, it is not believed that this solely would cause the marked difference in occurrence between the northern and southern sections. The reasoning for this is three-fold: *M. beryllina* have not been shown to be an important prey item in any previously published study, they have not been found in the stomachs of red drum from BSJ, and they have been sampled at Mirabeau and Dumaine bridge sites, just not from September through December 2010.

Water Quality Modeling

While the Mathematical Modeling method did not allow for biases towards any practical relationships between habitat selections, the results did suggest some relationships that may not be ecologically relevant. At the least, some of these relationships are difficult to explain. The nature of these data, high between-variable collinearities and non-random sampling, calls for a careful interpretation of these results as well.

Total's (pings per day from all three sites) reduced Mathematical GEE and all reduced Practical GEEs included a second-order polynomial relationship with temperature. This seems likely as my dataset included a wide range of temperatures (minimum = 8.73 °C, maximum = 31.91 °C). The second order polynomial observed in REL's reduced Mathematical GEE probably has more to do with salinity's collinearity with temperature. It is doubtful that red drum occurred more often at REL because of this, especially with such a small range of salinities observed. The complicated third-order relationship with salinity in NEI's reduced Mathematical GEE is difficult to explain, but may be a combination of high model variance and collinearity with temperature. The results of the reduced Practical GEEs suggest that red drum become more active as temperature reaches median values over this study period.

All three reduced Mathematical Models included the predictor variable Salinity^2 and NEI's model also included Salinity^3. A second or third order polynomial relationship between salinity and the number of pings per day may not be ecologically relevant, especially in an area with such low salinities and little change (min = 1.54, max = 2.24). Only one of the reduced Practical Models included Salinity as a variable. REL's reduced Practical Model included a straight line negative correlation with salinity. This relationship suggests that as salinity decreases in the bayou, red drum select northern habitats. Since this area is closer to Lake Pontchartrain, where all saline water enters BSJ, this relationship may be ecologically factual. As salinities decrease red drum were observed more often at REL, the area closest to higher salinity waters.

Depending on the response variable, either a first-order or second-order polynomial was selected to explain the relationship between number of pings per day and dissolved oxygen concentration for the Mathematical Models. This relationship is difficult to explain. Additionally, no extreme low mean daily dissolved oxygen concentration was ever measured throughout the study (minimum = 3.25 mg/L). Since no extreme low mean daily value was recorded and each response variable shows a negative first-order polynomial correlation, it is unlikely that an ecologically pertinent relationship between dissolved oxygen and any response variable exists.

No statistically significant regression was ever generated using depth as a predictor variable for any response variable and it was not included in any of the models generated for the

Mathematical GEEs. A relationship between habitat selection and tidal periodicity, with habitat selection being significantly different between low and high tides has been observed for red drum (Dresser and Kneib, 2007). The study area in which these observations took place has a normal daily tidal cycle of two low tides and two high tides per day. The Gulf of Mexico usually has only one high and low tide per day. Specifically along coastal Louisiana and the Lake Pontchartrain Basin, tides are mostly influenced by wind (Sikora, 1985). The confusing water management practices of BSJ further complicate any natural, daily tidal cycle. Despite this, daily tides have been found to be measureable in BSJ (Schroeder, 2011). NEI's reduced Practical GEE indicated a straight line relationship between the daily number of pings and depth. This receiver covered the deepest and most variety of depths among receivers in BSJ (Martinez *et al.*, 2008). This relationship may be a result of red drum using microhabitats at NEI during higher tides.

Conclusions

Red drum continue to occur more frequently in the northern habitats in Bayou St. John based upon the number of pings per day from September through December 2010. The epibenthic faunal survey from this time period indicates more *R. harrisii* at REL. This result, though, is based on a dataset that appears to be too small to indicate the true abundance: there were no significant differences when the larger dataset was analyzed. Significant differences in shoreline fish assemblages occurred overall and for each pairwise test between sites during this study. However, analysis of the species that contribute to these similarities and previous studies do not suggest that potential red drum prey items drive these assemblage differences. These results suggest red drum in Bayou St. John did not select any habitat measured (REL, NEI, and

I610) based upon differences in organisms measured by shoreline seining and benthic epifaunal minnow traps.

Changes in temperature, salinity, and depth may influence red drum habitat selection in Bayou St. John. Using multiple order polynomial predictor variables in mathematically driven models may increase the overall model effectiveness and reduce bias, but in this case produced results that were difficult to interpret. Practical Models indicated relationships easier to understand in an ecological sense, but may include some biases. I found that interpretability was more important. For Total, REL, and NEI, the number of pings per day could be predicted by a second order relationship with temperature. It appears that a first-order relationship between the number of pings per day and salinity exists at REL, with a decrease in the number of pings as salinity increases. This may be a response to fish seeking higher salinities in times of low salinity in BSJ. The number of pings per day at NEI is more closely related to change in depth, with a positive first-order correlation. This may be due to microhabitat differences at this site. Red drum activity appears to be highest at median temperature values in Bayou St. John during this study period.

Chapter 2

Introduction

As the human population increases and becomes more urbanized, anthropogenic impacts become more widespread. Greater than 75% of the U.S. population lives in urban areas and this number continues to grow (U.S. Census Bureau, 2001). As these trends unfold, understanding how urbanization affects waterways and how urban waterways becomes increasingly important. Studying the natural response of an aquatic ecosystem to anthropogenic impacts not only has local impacts. It also provides insight into how other ecosystems may respond to increased human development.

Impoundment, urbanization, and other anthropogenic impacts can negatively affect aquatic ecosystems and fish assemblages (Harrington and Harrington, 1982; Herke, 1995; Llanso *et al*., 1998; Chesney *et al*., 2000; Miller and Able, 2002; O'Connell *et al*., 2004; Stevens *et al*., 2006; O'Connell *et al*., 2009). In a study of fish assemblages before and after impoundment, 11 of 16 fish species collected before impoundment were not found after impoundment (Llanso, 1998). Impoundment and habitat alteration of inland waterways in Louisiana has reduced fisheries production (Herke, 1995). This reduction was partially caused by blocking naturally occurring passageways resulting in a lack of access and low escapement, immigration, and emigration. Not only are assemblages affected, but trophic interactions of organisms after impoundment can be altered (Llanso, 1998). However, restoring waterways along with the proper management approach can increase diversity and restore ecosystem function (Llanso, 1998).

City Park of New Orleans is an urban park located immediately west of Bayou St. John (Fig. 23). The City Park Lakes and Lagoons (CPLL) are all located on this property which is owned by the City of New Orleans. It is approximately 540 hectares and supports different habitats and land use practices than BSJ. There is little concrete bank stabilization and less of the waters in CPLL are bordered by roadways. However, in some cases similar effects have taken place at both CPLL and BSJ. For example, CPLL and BSJ were inundated with one to two meters of storm surge water for a period of 2 to 3 weeks in August and September of 2005 due to floodwall failures during Hurricane Katrina. This essentially created one large body of water as little of the land surrounding any of these normally contained waterways is above 1 m sea-level (BKI, 2011). As a result, land use practices have changed since Hurricane Katrina in City Park but, the land surrounding BSJ remains similar to the way it was before 2005. Approximately 23% of City Park of New Orleans was managed as undeveloped space in 2005, with that percentage closer to 60% today (New Orleans City Park Master Plan, 2011). Plans exist to repurpose much of the land, but the vast majority remains undeveloped as of this study (New Orleans City Park Master Plan, 2011).

City Park Lakes and Lagoons and Bayou St. John represent impounded brackish waterways within New Orleans, Louisiana. CPLL receives water from BSJ with salinities typically ranging from 0.5 to 5.0 (Cali, 1972; O'Connell, unpublished data; Fig. 23). Similar to BSJ, City Park offers nearby access for many of New Orleans' citizens to less urbanized and more natural settings. Many of the same activities that occur in BSJ also occur in City Park (See previous chapter for examples). There are three major sources of water supply for CPLL: BSJ, rainwater, and runoff. Water from BSJ travels into CPLL at three points: a 40.64 cm gravity fed pipe north of Mirabeau Avenue, pumps located at City Park Avenue and Carrollton Avenue, and pumps between LeLong Drive and Friedrichs Avenue (Fig. 23). Historically and currently, CPLL has lower salinity than BSJ, suggesting that rain and runoff have a more substantial impact than they do on BSJ (Cali, 1972; O'Connell, unpublished data).

I assessed fish assemblage change across a broad temporal scale (1971 – 2011) for CPLL and BSJ. Over this time period, multiple gear types and personnel sampled these waters. Each study had different goals and objectives. Presence-absence data were used to compare differences in taxonomic distinctness across all surveys. The 2006 – 2010 shoreline seining survey was designed specifically for assemblage analyses, with gears and effort standardized. I analyzed these data a second time, independently, to determine any differences in assemblages between CPLL and BSJ, among all sites, months, and years. Analysis and interpretation of these data were executed with the goal of understanding assemblage change in CPLL and BSJ over spatial, and broad temporal scales. Specifically, the goals of this study were to ask:

1. Has there been a change in taxonomic distinctness (a measure of biodiversity) of fish assemblages in City Park Lakes and Lagoons and Bayou St. John from 1971 to 2010?
2. Are there any compositional and abundance differences in shoreline fish assemblages in City Park Lake and Lagoons and Bayou St. John over a short temporal period (2006-2010)?

Figure 23: Image of City Park (in yellow) and the position of the pumps that control water into City Park from Bayou St. John. See Fig. 1 for location of City Park and Bayou St. John in relation to New Orleans, Louisiana and Lake Pontchartrain. Image adapted from Google Earth©.

Materials and Methods

Data Sources

Multiple fisheries independent surveys from CPLL and BSJ were utilized for my analyses (Cali, 1972; Ward, 1982; current study). From September 1971 - March 1972 fishes were sampled from two sites within BSJ and eight from CPLL, periodically. The goals of this project were to obtain a "qualitative survey" of all biota, including fishes. Seines, dip nets, traps and baited hooks were utilized (Cali, 1972). From February 1981 – January 1982 fishes were sampled from four sites within BSJ (Ward, 1982). Once a month, every month, shoreline fishes were sampled for 30 minutes using a 3.3 m seine (Ward, 1982). Six gillnet samples were collected from BSJ by the Louisiana Department of Wildlife and Fisheries (LDWF) between 1 December 1981 to 2 March 1982 (Ward, 1982). On 21 and 22 November 2002 the Nekton Research Lab (NRL) at the University of New Orleans (UNO) collected fishes at twelve sites in City Park by electrofishing. In February and October of 2008, the Louisiana Department of Wildlife and Fisheries collected fishes at nine different sites in City Park by electrofishing. Both electrofishing surveys used direct current from aluminum boats with a generator. From January 2006 to December 2010 the NRL has sampled six sites monthly using a 5 m beach seine. These samples were geared towards assemblage analyses by using three standardized seine hauls per site each month with data recorded for each haul. During 2010 and 2011, the NRL sampled two sites using a 30 m gill net with 50.8 mm and 203.2 mm stretch mesh. These samples were not standardized as they were not geared towards assemblage analysis. The objective of this was to recapture red drum as part of another study.

Some of these surveys collected data from CPLL only, BSJ only, or both. All data sets that included both locations included site information that allowed for division of fishes collected between CPLL and BSJ (Table 15). Therefore, it was possible to treat these two areas separately. Coverage within CPLL and BSJ differed among the tests, but each dataset had a minimum of two separate sites for one of the areas (Fig. 24).

Table 15. Summary of collectors, year(s), localities and gear types used for each dataset. The "X"s under each location represent that the area was sampled as a part of the study. Under the column gear type M = multiple, S = seine, G = gillnet, E = electrofishing. Multiple gear types include seines, dip nets, baited hooks, and traps. Under the Coverage column, Sh = shoreline, P = pelagic, and B = benthic. Each of the five years as a part of the NRL 2006-2011 seining survey were analyzed separately.

Bayou St. John and City Park Lakes and Lagoons Fisheries Independent Datasets

Collector	Year	City Park	Bayou St. John	Gear Type	Coverage
Cali III, F. J.	1971-1972	X	X	M	Sh
Ward, K. A.	1981-1982		X	S	Sh
LDWF	1981-1982		X	G	P
NRL	2002	X		E	Sh,P
LDWF	2008	X		E	Sh,P
NRL	2006-2011	X	X	S	Sh
NRL	2010-2011		X	G	P
LDWF	2008	X		E	Sh,P

Figure 24: Image of City Park lakes and lagoons system and Bayou St. John with the location of all sites from all surveys being analyzed. Black represents Cali multiple gears 1971 – 1972, purple – Ward seine 1981 – 1982, orange – LDWF gillnet 1982, blue – NRL electrofishing 2002, red – NRL seine 2006 – 2011, yellow – LDWF electrofishing 2008, green – NRL gillnet 2010 – 2011. Image adapted from Google Earth©.

Statistical analysis

Taxonomic distinctness statistical tests were performed using PRIMER (version. 5) software. This test was chosen because it can compare assemblages across a broad temporal scale without the need for standardizing effort or method (Warrick and Clarke, 1995; Clarke and Warrick, 1998; Clarke and Warrick, 2001). Even where multiple and different gear types were used without standardization, such as the 1971-1972 qualitative data, taxonomic distinctness tests can still be used to estimate biodiversity. This is done by comparing the presence-absence of each species from the master list from one dataset to the distribution of 1000 randomly generated subsamples from the master list with the same number of species. Two statistical values were generated: Average taxonomic distinctness (AvgTD, $\Delta+$) and variation in taxonomic distinctness (VarTD, $\Lambda+$). AvgTD measures the path lengths through the classification tree between all species pairs. From these lengths a mean is taken, resulting in the $\Delta+$ value for that survey. Lower than expected $\Delta+$ values indicate an assemblage with closely related individuals and decreased diversity (Warwick and Clarke, 1995). VarTD is the total variation of the taxonomic measurements for one survey (Clarke and Warrick, 2001). AvgTD and VarTD are generated for all randomly generated subsamples with the same n. From these confidence intervals are obtained and p-values are attributed to each dataset ($\alpha = 0.10$).

I compiled a master list that included all fishes sampled across all surveys, excluding non-native species. Non-native fishes were not included, because the purpose of the tests is to determine ecosystem change and these organisms were not a part of the original pre-European ecosystem. Inclusion of non-native fishes may increase diversity for some surveys or habitats, and the focus of my computations was to avoid this. Each species was assigned as many taxonomic levels as possible. The taxonomic levels were chosen based upon general acceptance and Nelson (2006). All fishes collected belonged to the same class, Actinopterygii, so it was deemed unnecessary to define above this level. Varying amounts of taxonomic division are available for each species, with some having many divisions and others having few. For example, *Lepisosteus oculatus* (spotted gar) was only divided into Class, Subclass, Order, Family, Genus, and Species, while *Cyprinodon variegatus* (sheepshead minnow) was divided into Class, Subclass, Division, Subdivision, Superorder, Series, Order, Suborder, Family, Subfamily, Tribe, Genus, and Species. If a species included a widely accepted division, it was included. The total list included 14 levels with the most inclusive being Class and the most exclusive being species (Class, Subclass, Division, Subdivision, Superorder, Series, Order, Suborder, Superfamily, Family, Subfamily, Tribe, Genus, Species). In cases where a species did not include a level, the next highest level was used in its place. This was done so that each species would have the same amount of divisions and each level of division was treated the same.

A statistical significance of $\alpha = 0.10$ was chosen for several reasons. No survey of fishes exists before any anthropogenic influences (such as impoundment) in the region; therefore it is likely that historically many more fish species occupied this area (Llanso, 1998). A safe assumption based upon this would be that many more species of fish from varied taxonomic backgrounds once occurred in CPLL, and BSJ. Also, Taxonomic distinctness tests were created to handle large databases with large master species lists (Warwick and Clarke, 1995). A function of this test is that variance in statistical values generated varies with the number of species in each master list and survey-sample. As the number of species in a master list or survey-sample decreases, the confidence intervals increase (Warrick and Clarke, 1995; Clarke and Warrick, 2001). In addition, a study comparing results of taxonomic distinctness tests to traditional univariate statistics found that taxonomic distinctness may not be very sensitive to anthropogenic disturbances (Bevilacqua *et al*., 2011). This, combined with a low number of species, may yield conservative results. In conclusion, $\alpha = 0.10$ was selected because the fish assemblages tested here are probably less specious and less taxonomically diverse than what was historically present, and the test may favor type I errors with datasets with a small number of species and possible anthropogenic impacts.

I grouped data based upon coverage, year, and site. This was done because the efficiency of gear types varies differently across habitats. Equating gear types with the habitats they sample allows for comparisons of specific habitats across years (Table 15). While boat electrofishing has been shown to be biased, it still could be considered as an estimate of pelagic and shoreline habitat diversity. Many biological, environmental, and technical factors affect efficiency of electrofishing, and because of this, results of diversity tests using electrofishing needs to be carefully considered (Reynolds, 1996). For any electrofishing surveys indicate lower than expected taxonomic distinctness values, determining whether any possible missing species could be a result of gear bias is important. Many biological biases are associated with electrofishing that could lead to type II error, where the hypothesis that the assemblage had a lower than expected taxonomic distinctness was incorrectly rejected (Sullivan, 1956; Larimore, 1961; Reynolds and Simpson, 1978). This will be considered when interpreting results from the electrofishing surveys. It was considered an estimate of both shoreline and pelagic habitats because all electrofishing surveys were performed from a boat. Gillnets sample pelagic habitats well and was used to compare this habitat over time. Seining surveys were used to estimate shoreline habitat across areas over time.

Analyses of similarity (ANOSIM) were performed after creating a Bray-Curtis similarity matrix, again using PRIMER to assess the 2006-2010 NRL seining survey (Clarke, 1993). These data were standardized with three samples taken at the same six sites, monthly (Fig 24). Three sampling sites were located within Area CPLL (Pontchartrain Lagoon, PON; Metairie Bayou, MET; Marconi, MAR) and three within Area BSJ (Robert E. Lee, REL; Mirabeau, MIR; Dumaine, DUM). One nested two-way ANOSIM test was performed for Areas (CPLL and BSJ) with Sites (all six sites) used as subgroups ($\alpha = 0.05$). Three different crossed two-way ANOSIMs were performed: Site x Month, Site x Year, and Month x Year ($\alpha = 0.05$). Any significant differences between Year pairs were analyzed using similarity percentage analysis (SIMPER) again using PRIMER (Clarke, 1993). This analysis lists species that drive the dissimilarity in the pairwise ANOSIM tests.

Results

Taxonomic distinctness

From all studies, 47 species of fishes were collected from CPLL and BSJ (Table 16). The number of species from each dataset varied from 8 to 21 for CPLL (Seine 2006 and Multiple gears from 1971, respectively) and from 10 to 27 from BSJ (Gillnet 2010-2011 and Seine 1981-1982, respectively). Twenty-seven fishes were sampled from CPLL from all surveys and 44 were sampled from BSJ (Table 21). Twenty-four of the fishes were sampled at both CPLL and BSJ. Three species were sampled in CPLL, but not BSJ. Twenty fishes were sampled in BSJ but not CPLL.

Table 16. A list of all of the species collected in Bayou St. John and City Park Lakes and Lagoons. This list includes species collected from all datasets from 1971 through 2010.

Species Collected from City Park Lakes and Lagoon and Bayou St. John

Scientific Name	Common Name	CPLL	BSJ
Atractosteus spatula	alligator gar		X
Lepisosteus oculatus	spotted gar	X	X
Lepisosteus osseus	longnose gar		X
Elops saurus	ladyfish		X
Anguilla rostrata	American eel	X	X
Anchoa mitchilli	bay anchovy		X
Brevoortia patronus	Gulf menhaden		X
Dorosoma cepedianum	gizzard shad	X	X
Dorosoma petenense	threadfin shad	X	X
Ictiobus bubalus	smallmouth buffalo		X
Ameiurus natalis	yellow bullhead	X	X
Ictalurus furcatus	blue catfish		X
Ictalurus punctatus	channel catfish		X
Mugil cephalus	striped mullet	X	X
Menidia beryllina	inshore silverside	X	X
Strongylura marina	Atlantic needlefish	X	X
Adinia xenica	diamond killifish		X
Fundulus chrysotus	golden topminnow	X	
Fundulus grandis	Gulf killifish	X	X
Fundulus majalis	striped killifish		X
Lucania parva	rainwater killifish	X	X
Gambusia affinis	western mosquitofish	X	X
Heterandria formosa	least killifish	X	X
Poecilia latipinna	sailfin molly	X	X
Cyprinodon variegatus	sheepshead minnow	X	X
Syngnathus scovelli	Gulf pipefish	X	X
Morone chrysops x Morone saxatilis	hybrid striped bass	X	X
Morone mississippiensis	yellow bass	X	X
Lepomis cyanellus	green sunfish	X	
Lepomis gulosus	warmouth	X	X
Lepomis macrochirus	bluegill	X	X
Lepomis microlophus	redear sunfish	X	X
Lepomis miniatus	redspotted sunfish	X	X
Lepomis symmetricus	bantam sunfish		X
Micropterus salmoides	largemouth bass	X	X

Table 16 Continued

Scientific Name	Common Name	CPLL	BSJ
Pomoxis nigromaculatus	black crappie	X	
Lagodon rhomboides	pinfish		X
Cynoscion nebulosus	spotted seatrout		X
Leiostomus xanthurus	spot		X
Sciaenops ocellatus	red drum		X
Sciaenops ocellatus x Pogonias cromis	red drum x black drum		X
Hypsoblennius ionthas	freckled blenny		X
Dormitator maculatus	fat sleeper		X
Gobionellus shufeldti	freshwater goby	X	X
Gobiosoma bosc	naked goby	X	X
Microgobius gulosus	clown goby		X
Trinectes maculatus	hogchoker		X
	Total		
47		27	44

Of the eight surveys from CPLL analyzed using taxonomic distinctness, three were found to have significant differences in either average taxonomic distinctness, variation in taxonomic distinctness or both. The electrofishing survey taken by the NRL in 2002 showed a significant value for variation in taxonomic distinctness (n = 15; Λ+ =622.213, *p* = 0.02; Table 17; Fig. 25). When comparing this value to the frequency histogram generated by 1000 random samples of the same number of species, the Λ+ value from this sample is higher than expected. The Δ+ value was not found to be significant for the 2002 electrofishing subsample (Δ+ =58.776, *p* = 0.152). The shoreline seining subsample from 2006 had a significant Δ+ and Λ+ values (n = 8; Δ+ = 49.745, *p* = 0.042; Λ+ = 592.136, *p* = 0.088; Table 17; Fig 25). Comparison between Δ+ values and the randomly generated frequency histogram generated indicates the value from the subsample to be lower than expected. Comparing Λ+ to the randomly generated frequency histogram indicates the subsample's Λ+ to be higher than expected. The shoreline seining subsample from 2007 had a significant Δ+ that was found to be lower than expected when compared to the randomly generated frequency histogram (n = 11; Δ+ = 48.442, *p* = 0.012; Table 17; Fig. 25). The Λ+ value was not found to be significant for the 2007 seining subsample (n = 11; Λ+ = 331.523, *p* = 0.973; Table 17; Fig. 25). The remaining five datasets, multiple gears from 1971-1972, electrofishing in 2008, and seine samples from 2008 through 2010, were not found to exhibit any significant differences with respect to Δ+ or Λ+ (Table 17; Fig. 25).

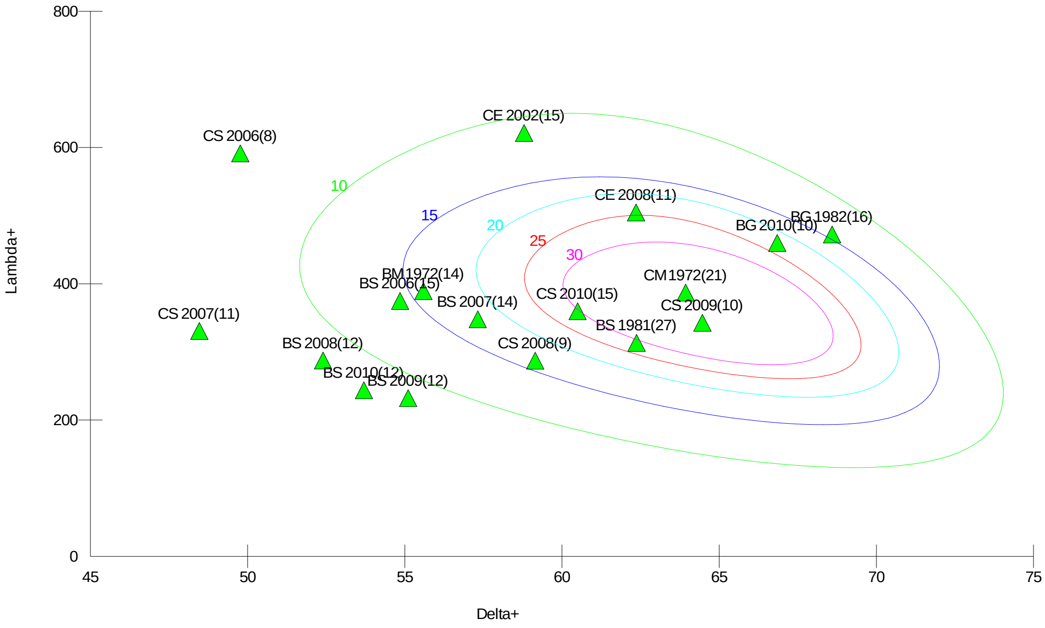

Figure 25: Scatter plot of variation in taxonomic distinctness (Delta +; Δ+) and average taxonomic distinctness (Lambda+; Λ+) pairs for each of the surveys analyzed from 1971-2011 in BSJ and CPLL with superimposed probability ellipses (α = 0.10). Each triangle represents a survey analyzed, with the first letter denoting area (C = CPLL; B = BSJ), the second denoting gear type (S = seine, M = multiple, E = electrofishing, G = gillnet), the first number representing year, and the number in parenthesis represents the number of species sampled during the survey. Each ellipse includes the expected range of Δ+ and Λ+ values randomly generated from 1000 simulations for a given number of species (denoted on each ellipse).

Table 17. Results on City Park Lakes and Lagoons surveys from Taxonomic distinctness tests analyzed with Primer 5 software. Gear type and year distinguish between each dataset analyzed. AvgTD (Δ+) refers to average taxonomic distinctness, and VarTD (Λ+) refers to variation in taxonomic distinctness. Significant p-values are in bold (α = 0.10).

CPLL Taxonomic Distinctness Results

Gear Type	Year	Number of Species	AvgTD (Δ+)	AvgTD (Δ+) p-value	VarTD (Λ+)	VarTD (Λ+) p-value
Multiple	1971-1972	21	63.912	0.689	387.858	0.619
Electrofishing	2002	15	58.776	0.152	622.213	**0.02**
Seine	2006	8	49.745	**0.042**	592.136	**0.088**
Seine	2007	11	48.442	**0.012**	331.523	0.973
Seine	2008	9	59.127	0.328	288.013	0.833
Seine	2009	10	64.444	0.863	343.512	0.883
Seine	2010	15	60.476	0.304	360.609	0.849
Electrofishing	2008	11	62.338	0.579	505.482	0.156

Of the nine different sampling datasets analyzed using taxonomic distinctness from Bayou St. John six were found to be significantly different with respect to Δ+ (Table 18; Fig. 25). There were no significant differences found for any of the nine Λ+ values for any dataset (Table 18). The dataset using multiple gear types form 1971-1972 was found have a significant Δ+ value, and when compared to the randomly generated frequency distribution was lower than expected (n = 14; Δ+ = 55.573, *p* = 0.046; Table 18; Fig. 25). The seining datasets from 2006, 2007, 2008, 2009, and 2010 were all found to have significant Δ+ values and when compared to the randomly generated frequency distribution all were lower than expected (Table 18; Fig. 25). The seining dataset from 1981-1982 did not have a significant Δ+ or Λ+ (Table 18; Fig. 25).

Similarly, neither of the two gillnet datasets (1982, 2010-2011) had significant Δ+ or Λ+ values (Table 18; Fig. 25).

Table 18. Results on Bayou St. John surveys from Taxonomic distinctness tests analyzed with Primer 5 software. Gear type and year distinguish between each dataset analyzed. AvgTD (Δ+) refers to average taxonomic distinctness, and VarTD (Λ+) refers to variation in taxonomic distinctness. Significant p-values are in bold ($\alpha = 0.10$).

BSJ Taxonomic Distinctness Results

Gear Type	Year	Number of Species	AvgTD (Δ+)	AvgTD (Δ+) p-value	VarTD (Λ+)	VarTD (Λ+) p-value
Multiple	1971-1972	14	55.573	**0.046**	390	0.593
Seine	1981-1982	27	62.352	0.3	314.159	0.308
Gillnet	1982	16	68.571	0.296	473.129	0.162
Seine	2006	15	54.83	**0.038**	375.609	0.783
Seine	2007	14	57.3	**0.098**	348.708	0.927
Seine	2008	12	52.381	**0.02**	288.085	0.599
Seine	2009	12	55.087	**0.07**	233.094	0.246
Seine	2010	12	53.68	**0.038**	244.654	0.312
Gillnet	2010-2011	10	66.825	0.743	460.67	0.3

Analysis of Similarity

Twenty-seven different fish species were sampled across the five year seining survey (Table 19). Four species were collected from the three sites at CPLL and not in BSJ: *L. oculatus*, *Fundulus chrysotus* (golden topminnow), *Morone chrysops* x *saxatilis* (hybrid striped bass), and *Pomoxis nigromaculatus* (black crappie). Seven species were sampled from at least one site in Area BSJ that was not sampled from Area CPLL: *Mugil cephalus* (striped mullet), *Strongylura marina* (Atlantic needlefish), *Adinia xenica* (diamond killifish), *Lepomis microlophus* (striped mullet), *L. miniatus* (striped mullet), *Cynoscion nebulosus* (spotted seatrout) and *Microgobius gulosus* (clown goby). Seventeen species were found to co-occur in both Areas (Table 19). The nested two-way ANOSIM, testing for dissimilarities between Areas using sites as subgroups indicated there was not a significant difference (Global R = 0, p = 0.6). Two-way crossed analysis indicated a significant difference among sites (Global R = 0.092, p = 0.001).

Table 19. A list of all of the species collected in Bayou St. John (BSJ) and City Park Lakes and Lagoons (CPLL) from the shoreline seine samples from 2006-2010.

Species sampled from City Park Lakes and Lagoons and Bayou St. John

Scientific Name	Common Name	CPLL	BSJ
Lepisosteus oculatus	spotted gar	X	
Dorosoma petenense	threadfin shad	X	X
Mugil cephalus	striped mullet		X
Menidia beryllina	inshore silverside	X	X
Strongylura marina	Atlantic needlefish		X
Adinia xenica	diamond killifish		X
Fundulus chrysotus	golden topminnow	X	
Fundulus grandis	Gulf killifish	X	X
Lucania parva	rainwater killifish	X	X
Gambusia affinis	western mosquitofish	X	X
Heterandria formosa	least killifish	X	X
Poecilia latipinna	sailfin molly	X	X
Cyprinodon variegatus	sheepshead minnow	X	X
Syngnathus scovelli	Gulf pipefish	X	X
Morone saxatilis x chrysops	hybrid striped bass	X	
Lepomis gulosus	warmouth	X	X
Lepomis macrochirus	bluegill	X	X
Lepomis microlophus	redear sunfish		X
Lepomis miniatus	redspotted sunfish		X
Micropterus salmoides	largemouth bass	X	X
Pomoxis nigromaculatus	black crappie	X	
Cynoscion nebulosus	spotted seatrout		X
Gobiosoma bosc	naked goby	X	X
Microgobius gulosus	clown goby		X

Crossed two-way ANOSIM, testing for significant dissimilarities among Sites and Months indicated significant differences among both groupings (Global R = 0.166, *p* = 0.001; Global R = 0.093, *p* = 0.001; respectively). Pairwise testing between Sites averaged across Years indicated each site was significantly different from all other sites except for Pontchartrain Lagoon and Robert E. Lee (R = 0.02, *p* = 0.076; Table 20). Pairwise ANOSIM between Months averaged across Sites indicated that all Month pairs averaged across sites were significantly different except: January, February; January, November; February, March; February, November; March, April; April, May; May, June; June, July; July, August; August, September; September, October; October, November; October, December; and November, December (Table 21).

Table 20. Pairwise results generated from a crossed two-way ANOSIM testing for significant dissimilarities between Sites crossed with Months. Values in bold are significant at $\alpha = 0.05$.

Pairwise Results: Sites Averaged Across Months

Groups	R Statistic	p-value
REL, MIR	0.133	**0.001**
REL, DUM	0.125	**0.001**
REL, PON	0.222	**0.001**
REL, MET	0.189	**0.001**
REL, MAR	0.245	**0.001**
MIR, DUM	0.092	**0.001**
MIR, PON	0.346	**0.001**
MIR, MET	0.206	**0.001**
MIR, MAR	0.206	**0.001**
DUM, PON	0.165	**0.001**
DUM, MET	0.093	**0.001**
DUM, MAR	0.098	**0.001**
PON, MET	0.02	0.076
PON, MAR	0.243	**0.001**
MET, MAR	0.113	**0.001**

Table 21. Pairwise results generated from a crossed two-way ANOSIM testing for significant dissimilarities between Months crossed with Sites. Values in bold are significant at α = 0.05.

Pairwise Results: Months Averaged Across Sites

Groups	R Statistic	p-value
January, February	-0.006	0.579
January, March	0.039	**0.043**
January, April	0.113	**0.001**
January, May	0.155	**0.001**
January, June	0.161	**0.001**
January, July	0.129	**0.001**
January, August	0.086	**0.001**
January, September	0.065	**0.004**
January, October	0.088	**0.001**
January, November	0.024	0.11
January, December	0.038	**0.045**
February, March	-0.015	0.744
February, April	0.063	**0.007**
February, May	0.109	**0.001**
February, June	0.125	**0.001**
February, July	0.125	**0.001**
February, August	0.106	**0.001**
February, September	0.061	**0.004**
February, October	0.081	**0.001**
February, November	0.034	0.052
February, December	0.078	**0.001**
March, April	0.017	0.176
March, May	0.068	**0.002**
March, June	0.102	**0.002**
March, July	0.127	**0.001**
March, August	0.138	**0.001**
March, September	0.113	**0.002**
March, October	0.089	**0.001**
March, November	0.073	**0.004**
March, December	0.108	**0.001**
April, May	0.017	0.174
April, June	0.041	**0.03**
April, July	0.057	**0.009**
April, August	0.109	**0.001**
April, September	0.146	**0.001**
April, October	0.112	**0.001**
April, November	0.159	**0.001**
April, December	0.216	**0.001**

Table 21 continued

Groups	R Statistic	Significance level
May, June	0.023	0.116
May, July	0.054	**0.009**
May, August	0.132	**0.001**
May, September	0.182	**0.001**
May, October	0.166	**0.001**
May, November	0.244	**0.001**
May, December	0.289	**0.001**
June, July	0.001	0.448
June, August	0.074	**0.003**
June, September	0.106	**0.001**
June, October	0.169	**0.001**
June, November	0.206	**0.001**
June, December	0.268	**0.001**
July, August	0.001	0.428
July, September	0.042	**0.029**
July, October	0.097	**0.002**
July, November	0.183	**0.001**
July, December	0.194	**0.001**
August, September	0	0.458
August, October	0.072	**0.002**
August, November	0.115	**0.001**
August, December	0.1	**0.001**
September, October	0.024	0.123
September, November	0.06	**0.005**
September, December	0.05	**0.018**
October, November	0.02	0.145
October, December	0.025	0.088
November, December	-0.004	0.52

Crossed two-way ANOSIM, testing for significant dissimilarities among Sites and Years indicated significant differences among both groupings (Global R = 0.231, *p* = 0.001; Global R = 0.273, *p* = 0.001; respectively). Pairwise testing between Sites averaged across Years indicated each Site was significantly different from every other Site (Table 22). Pairwise ANOSIM between Years averaged across Sites indicated that all Years were significantly different every other Year (Table 23).

Table 22. Pairwise results generated from a crossed two-way ANOSIM testing for significant dissimilarities between Sites crossed with Years.

Pairwise Results: Sites Averaged Across Years		
Groups	R Statistic	Significant Level
REL, MIR	0.142	0.001
REL, DUM	0.146	0.001
REL, PON	0.311	0.001
REL, MET	0.318	0.001
REL, MAR	0.29	0.001
MIR, DUM	0.102	0.001
MIR, PON	0.419	0.001
MIR, MET	0.306	0.001
MIR, MAR	0.226	0.001
DUM, PON	0.243	0.001
DUM, MET	0.192	0.001
DUM, MAR	0.137	0.001
PON, MET	0.083	0.001
PON, MAR	0.39	0.001
MET, MAR	0.251	0.001

Table 23. Pairwise results generated from a crossed two-way ANOSIM testing for significant dissimilarities between Years crossed with Sites.

Pairwise Results: Years Averaged Across Sites		
Groups	R Statistic	Significance level
2006, 2007	0.152	0.001
2006, 2008	0.308	0.001
2006, 2009	0.558	0.001
2006, 2010	0.447	0.001
2007, 2008	0.143	0.001
2007, 2009	0.363	0.001
2007, 2010	0.34	0.001
2008, 2009	0.162	0.001
2008, 2010	0.157	0.001
2009, 2010	0.075	0.001

Crossed two-way ANOSIM, testing for significant dissimilarities among Months and Years indicated significant differences among both groupings (Global R = 0.231, $p = 0.001$;

Global R = 0.273, *p* = 0.001; respectively). Pairwise testing between Months averaged across Years indicated all Month pairs were significantly different except: January, February; February, March; June, July; August, September; September, October; October, November; and November, December (Table 24). Pairwise ANOSIM between Years averaged across Sites indicated that all Years were significantly different (Table 25).

Table 24. Pairwise results generated from a crossed two-way ANOSIM testing for significant dissimilarities between Months crossed with Years. Values in bold are significant at α = 0.05

Pairwise Crossed Two-way ANOSIM among Months across Years

Groups	R Statistic	Significance level
January, February	0.028	0.056
January, March	0.074	**0.003**
January, April	0.153	**0.001**
January, May	0.217	**0.001**
January, June	0.27	**0.001**
January, July	0.211	**0.001**
January, August	0.151	**0.001**
January, September	0.141	**0.001**
January, October	0.138	**0.001**
January, November	0.142	**0.001**
January, December	0.158	**0.001**
February, March	0.022	0.123
February, April	0.09	**0.001**
February, May	0.144	**0.001**
February, June	0.2	**0.001**
February, July	0.162	**0.001**
February, August	0.169	**0.001**
February, September	0.157	**0.001**
February, October	0.157	**0.001**
February, November	0.169	**0.001**
February, December	0.222	**0.001**
March, April	0.06	**0.004**
March, May	0.142	**0.001**
March, June	0.171	**0.001**
March, July	0.187	**0.001**
March, August	0.194	**0.001**
March, September	0.213	**0.001**
March, October	0.201	**0.001**
March, November	0.179	**0.001**
March, December	0.249	**0.001**
April, May	0.046	**0.007**
April, June	0.117	**0.001**
April, July	0.102	**0.001**
April, August	0.202	**0.001**
April, September	0.252	**0.001**
April, October	0.218	**0.001**
April, November	0.25	**0.001**
April, December	0.352	**0.001**

Table 24 continued

Groups	R Statistic	Significance level
May, June	0.075	**0.002**
May, July	0.067	**0.003**
May, August	0.138	**0.001**
May, September	0.194	**0.001**
May, October	0.175	**0.001**
May, November	0.243	**0.001**
May, December	0.354	**0.001**
June, July	0.009	0.249
June, August	0.119	**0.001**
June, September	0.153	**0.001**
June, October	0.14	**0.001**
June, November	0.197	**0.001**
June, December	0.336	**0.001**
July, August	0.088	**0.003**
July, September	0.111	**0.001**
July, October	0.115	**0.001**
July, November	0.194	**0.001**
July, December	0.294	**0.001**
August, September	0.019	0.139
August, October	0.041	**0.026**
August, November	0.116	**0.001**
August, December	0.181	**0.001**
September, October	-0.015	0.808
September, November	0.023	0.106
September, December	0.07	**0.003**
October, November	0.01	0.258
October, December	0.044	**0.025**
November, December	-0.001	0.457

Table 25. Pairwise results generated from a crossed two-way ANOSIM testing for significant dissimilarities between Years crossed with Months.

Pairwise Crossed Two-way ANOSIM among Months across Years

Groups	R Statistic	p-value
2006, 2007	0.151	0.001
2006, 2008	0.332	0.001
2006, 2009	0.491	0.001
2006, 2010	0.465	0.001
2007, 2008	0.113	0.001
2007, 2009	0.227	0.001
2007, 2010	0.213	0.001
2008, 2009	0.172	0.001
2008, 2010	0.169	0.001
2009, 2010	0.097	0.001

SIMPER Analysis

Similarity percentages were generated for each pairwise Year combination because each pair was found to be significantly different for both crossed two-way ANOSIMs (Tables 26 - 35). SIMPER analysis between 2006 and 2007 indicate that higher abundances of *Lucania parva* (rainwater killifish), *Gambusia affinis* (western mosquitofish), *M. beryllina*, *C. variegatus* and *Heterandria formosa* (least killifish) for 2006 and lower abundances of *Poecilia latipinna* (sailfin molly) and *G. bosc* for 2006 drive the assemblage change (Table 35). Similarly, higher abundances of *L. parva*, G. *affinis*, *M. beryllina C. variegatus*, *H. formosa*, along with higher abundances of *P. latipinna* in 2006 appear to be driving the differences between 2006 and 2008 (Table 36). Lower abundances of *L. macrochirus* also contribute. A lower mean abundance of *M. beryllina* in 2006 versus 2009 contributes the most to differences between these years by species (Table 37). Higher abundances of *L. parva*, *G. affinis*, *P. latipinna*, *C. variegates*, and *H. formosa* were seen in 2006 compared to 2009. Lower abundances of *L. macrochirus* in 2006 versus 2009 also contributed. *L. parva* was the species that contributed most to the difference between 2006 and 2010, with higher mean abundance observed in 2006 (Table 38). *P. latipinna*, *C. variegatus*, and *H. formosa* contributed, with higher mean abundances in 2006. *M. beryllina*, *L. macrochirus*, and *M. salmoides* were all included in the list of species that contributed 90% of the differences between 2006 and 2010, with higher mean abundances seen in 2010. Between 2007 and 2008, *L. parva*, *G. affinis*, *P. latipinna*, and *C. variegatus* all contributed to differences with higher mean abundances observed in 2007. Higher abundances were observed in 2008 when compared to 2007 for *L. macrochirus* and *M. salmoides*. Higher abundances of *M. beryllina* in 2009 contributed the most to the differences between 2007 and 2009. Lower mean abundances of *L. parva*, *P. latipinna*, and *G. affinis* in 2009 also contributed. Mean abundances of *L. macrochirus* and *M. salmoides* were higher in 2009. The same relationships described between 2007 and 2009 were also seen between 2007 and 2010. The species that contributed the most to the differences between 2008 and 2009 was *M. beryllina* with a higher mean abundance in 2009. Higher mean abundances of *L. parva* and *P. latipinna* in 2008 than in 2009 contributed. The species that contributed the most to the differences between 2008 and 2010 was *M. beryllina*, with a higher mean abundance in 2010. Higher mean abundances for *P. latipinna* and *L. parva* were seen in 2008, while *M. salmoides* and *L. macrochirus* had higher abundances in 2010. Again, *M. beryllina* was the species that contributed the most to the differences in fish assemblage between 2009 and 2010, with a higher mean abundance in 2009. Higher mean abundances in *L. macrochirus*, *G. affinis*, and *M. salmoides* were found in 2010, with *G. bosc* being lower in mean abundance for 2010.

Across years, SIMPER analysis indicates a decrease in five fishes from the Order Cyprinodontiformes, representing three families (Fundulidae: *L. parva*; Poeciliidae: G. *affinis*, *P. latipinna*, *H. formosa*, and Cyprinodontidae: *C. variegatus*), while average abundances of *M.*

beryllina, *L. macrochirus* and *M. salmoides* increased across years (Figs. 26 and 27). This trend becomes clear when comparing extreme abundance values for 2006 and 2007 to extreme values from 2009 and 2010. Mean abundance for *L. parva* was 22.46 in 2006 and 0.47 in 2010. Mean abundance for *P. latipinna* was 16.64 for 2007 and 0.28 in 2010. Mean abundance for *H. formosa* was 4.19 in 2006 and 0 in 2009. From July 2008 through June 2010 there were 0 *H. formosa* sampled from any site (n = 432 samples). Mean abundance for *C. variegatus* was 4.44 in 2006 and 0.06 in 2009. Mean abundance for *L. macrochirus* was 0.03 in 2006 and it was 3.88 in 2010. Mean abundance for *M. salmoides* was 0.18 in 2007 and it was 1.52 in 2010. Mean abundance for *M. beryllina* was 4.55 in 2007 and 19.62 in 2009.

Table 26. Similarity Percentages (SIMPER) showing the species that contributes up to 90% of the differences between 2006 and 2007. Only the species that contributed up to 90% of the total assemblage change are shown. Values in the Contribution percentage column represent the average dissimilarity each species contributed to the overall dissimilarity between assemblages.

SIMPER Analysis Results

2006 : 2007

Species	2006 Mean Abundance	2007 Mean Abundance	Mean Dissimilarity	Contribution %
Lucania parva	22.46	14.63	24.91	28.43
Gambusia affinis	18.12	2.75	17.06	19.47
Poecilia latipinna	11.86	16.64	14.69	16.76
Menidia beryllina	8.39	4.55	8.25	9.42
Cyprinodon variegatus	4.44	2.74	5.53	6.31
Heterandria formosa	4.19	0.64	4.54	5.18
Gobiosoma bosc	0.31	0.57	4.12	4.7

Table 27. Similarity Percentages (SIMPER) showing the species that contributes up to 90% of the differences between 2006 and 2008. Only the species that contributed up to 90% of the total assemblage change are shown. Values in the Contribution percentage column represent the average dissimilarity each species contributed to the overall dissimilarity between assemblages.

SIMPER Analysis Results

2006 : 2008

Species	2006 Mean Abundance	2008 Mean Abundance	Mean Dissimilarity	Contribution %
Lucania parva	22.46	5.19	24.51	26.48
Gambusia affinis	18.12	0.28	17.98	19.42
Poecilia latipinna	11.86	4.02	11.79	12.74
Menidia beryllina	8.39	2.8	10.84	11.71
Lepomis macrochirus	0.03	2.02	6.46	6.98
Cyprinodon variegatus	4.44	0.61	5.22	5.64
Heterandria formosa	4.19	0.06	4.48	4.85
Gobiosoma bosc	0.31	0.27	4.05	4.83

Table 28. Similarity Percentages (SIMPER) showing the species that contributes up to 90% of the differences between 2006 and 2009. Only the species that contributed up to 90% of the total assemblage change are shown. Values in the Contribution percentage column represent the average dissimilarity each species contributed to the overall dissimilarity between assemblages.

SIMPER Analysis Results

2006 : 2009

Species	2006 Mean Abundance	2009 Mean Abundance	Mean Dissimilarity	Contribution %
Menidia beryllina	8.39	19.62	22.96	24.18
Lucania parva	22.46	0.08	20.03	21.1
Gambusia affinis	18.12	0.85	17.42	18.34
Poecilia latipinna	11.86	0.08	8.74	9.2
Lepomis macrochirus	0.03	2.02	6.91	7.27
Gobiosoma bosc	0.31	2.01	4.72	4.97
Cyprinodon variegatus	4.44	0.06	4.51	4.75
Heterandria formosa	4.19	0	4.03	4.25

Table 29. Similarity Percentages (SIMPER) showing the species that contributes up to 90% of the differences between 2006 and 2010. Only the species that contributed up to 90% of the total assemblage change are shown. Values in the Contribution percentage column represent the average dissimilarity each species contributed to the overall dissimilarity between assemblages.

SIMPER Analysis Results

2006 : 2010

Species	2006 Mean Abundance	2010 Mean Abundance	Mean Dissimilarity	Contribution %
Lucania parva	22.46	0.47	19.83	21.21
Menidia beryllina	8.39	13.26	18.71	20.02
Gambusia affinis	18.12	1.64	17.92	19.17
Lepomis macrochirus	0.03	3.88	8.82	9.44
Poecilia latipinna	11.86	0.28	8.71	9.32
Micropterus salmoides	0.32	1.52	4.83	5.17
Cyprinodon variegatus	4.44	0.32	4.83	5.17
Heterandria formosa	4.19	0.06	4.01	4.29

Table 30. Similarity Percentages (SIMPER) showing the species that contributes up to 90% of the differences between 2007 and 2008. Only the species that contributed up to 90% of the total assemblage change are shown. Values in the Contribution percentage column represent the average dissimilarity each species contributed to the overall dissimilarity between assemblages.

SIMPER Analysis Results

2007 : 2008

Species	2007 Mean Abundance	2008 Mean Abundance	Mean Dissimilarity	Contribution %
Lucania parva	14.63	5.19	22.46	25.1
Poecilia latipinna	16.64	4.02	14.73	16.46
Menidia beryllina	4.55	2.8	12.86	14.36
Lepomis macrochirus	0.88	2.02	12.44	13.91
Gambusia affinis	2.75	0.28	7	7.82
Gobiosoma bosc	0.57	0.27	6.03	6.74
Micropterus salmoides	0.18	0.37	3.52	3.93
Cyprinodon variegatus	2.74	0.61	3.42	3.82

Table 31. Similarity Percentages (SIMPER) showing the species that contributes up to 90% of the differences between 2007 and 2009. Only the species that contributed up to 90% of the total assemblage change are shown. Values in the Contribution percentage column represent the average dissimilarity each species contributed to the overall dissimilarity between assemblages.

SIMPER Analysis Results

2007 : 2009

Species	2007 Mean Abundance	2009 Mean Abundance	Mean Dissimilarity	Contribution %
Menidia beryllina	4.55	19.62	27.76	30.8
Lucania parva	14.63	0.08	15.66	17.38
Lepomis macrochirus	0.88	2.02	11.5	12.76
Poecilia latipinna	16.64	0.08	10.72	11.9
Gambusia affinis	2.75	0.85	7.25	8.04
Gobiosoma bosc	0.57	2.01	6.79	7.54
Micropterus salmoides	0.18	0.32	2.72	3.01

Table 32. Similarity Percentages (SIMPER) showing the species that contributes up to 90% of the differences between 2007 and 2010. Only the species that contributed up to 90% of the total assemblage change are shown. Values in the Contribution percentage column represent the average dissimilarity each species contributed to the overall dissimilarity between assemblages.

SIMPER Analysis Results

2007 : 2010

Species	2007 Mean Abundance	2010 Mean Abundance	Mean Dissimilarity	Contribution %
Menidia beryllina	4.55	13.26	22.23	24.72
Lucania parva	14.63	0.47	15.51	17.25
Lepomis macrochirus	0.88	3.88	13.75	15.29
Poecilia latipinna	16.64	0.28	10.68	11.88
Gambusia affinis	2.75	1.64	8.63	9.6
Micropterus salmoides	0.18	1.52	6.15	6.84
Gobiosoma bosc	0.57	0.43	4.79	5.33

Table 33. Similarity Percentages (SIMPER) showing the species that contributes up to 90% of the differences between 2008 and 2009. Only the species that contributed up to 90% of the total assemblage change are shown. Values in the Contribution percentage column represent the average dissimilarity each species contributed to the overall dissimilarity between assemblages.

SIMPER Analysis Results

2008 : 2009

Species	2008 Mean Abundance	2009 Mean Abundance	Mean Dissimilarity	Contribution %
Menidia beryllina	2.8	19.62	34.4	39.83
Lepomis macrochirus	2.02	2.02	16.3	18.87
Lucania parva	5.19	0.08	9.38	10.86
Gobiosoma bosc	0.27	2.01	6.28	7.27
Poecilia latipinna	4.02	0.08	4.87	5.64
Gambusia affinis	0.28	0.85	4.79	5.55
Micropterus salmoides	0.37	0.32	3.36	3.89

Table 34. Similarity Percentages (SIMPER) showing the species that contributes up to 90% of the differences between 2008 and 2010. Only the species that contributed up to 90% of the total assemblage change are shown. Values in the Contribution percentage column represent the average dissimilarity each species contributed to the overall dissimilarity between assemblages.

SIMPER Analysis Results

2008 : 2010

Species	2008 Mean Abundance	2010 Mean Abundance	Mean Dissimilarity	Contribution %
Menidia beryllina	2.8	13.26	27.63	31.63
Lepomis macrochirus	2.02	3.88	18.73	21.45
Lucania parva	5.19	0.47	9.36	10.72
Micropterus salmoides	0.37	1.52	7.67	8.78
Gambusia affinis	0.28	1.64	7.25	8.29
Poecilia latipinna	4.02	0.28	4.98	5.7
Gobiosoma bosc	0.27	0.43	3.97	4.54

Table 35. Similarity Percentages (SIMPER) showing the species that contributes up to 90% of the differences between 2009 and 2010. Only the species that contributed up to 90% of the total assemblage change are shown. Values in the Contribution percentage column represent the average dissimilarity each species contributed to the overall dissimilarity between assemblages.

SIMPER Analysis Results

2009 : 2010

Species	2009 Mean Abundance	2010 Mean Abundance	Mean Dissimilarity	Contribution %
Menidia beryllina	19.62	13.26	38.79	47.56
Lepomis macrochirus	2.02	3.88	16.16	19.81
Gambusia affinis	0.85	1.64	7.43	9.11
Micropterus salmoides	0.32	1.52	6.53	8
Gobiosoma bosc	2.01	0.43	5.3	6.49

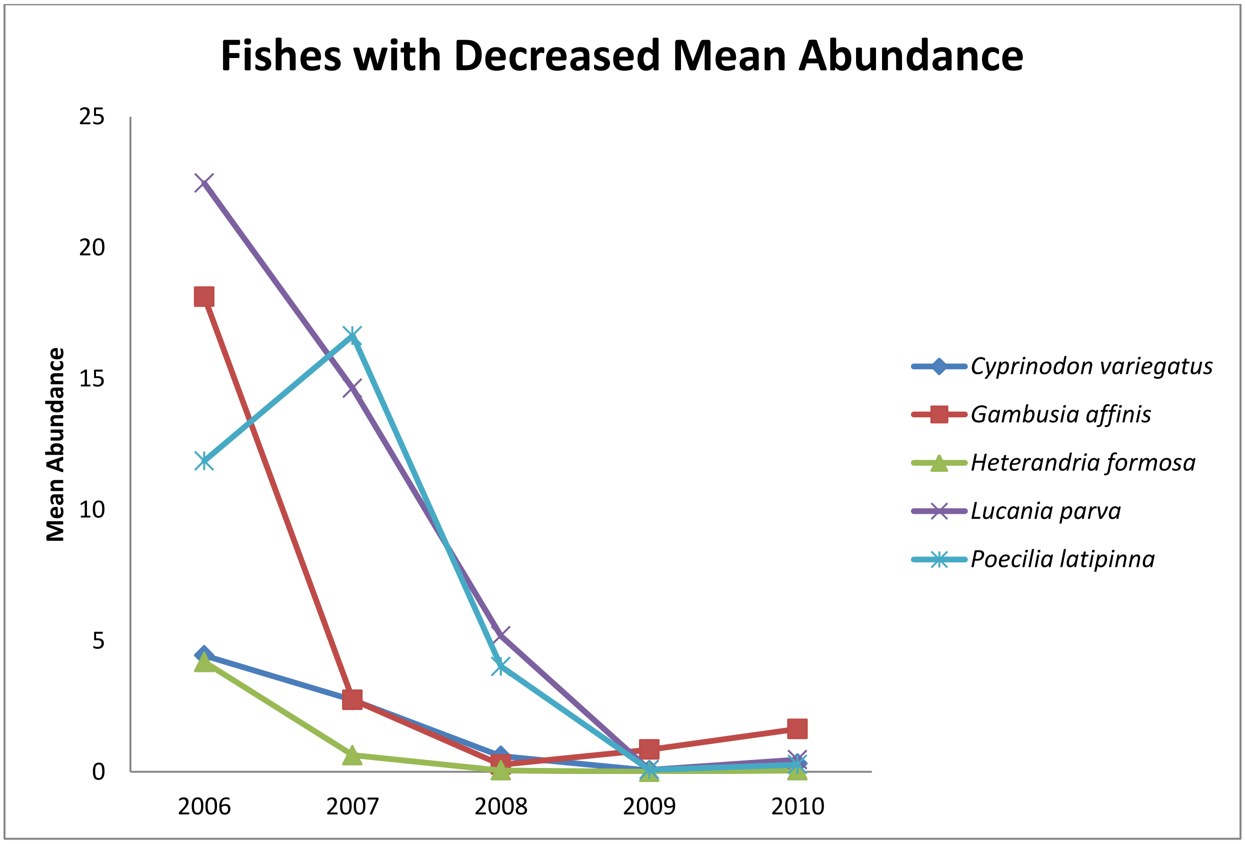

Figure 26: Chart of line graphs representing five species of fish within the Order Cyprinodontiformes whose mean abundances decreased across years (2006 – 2010) from a seining survey.

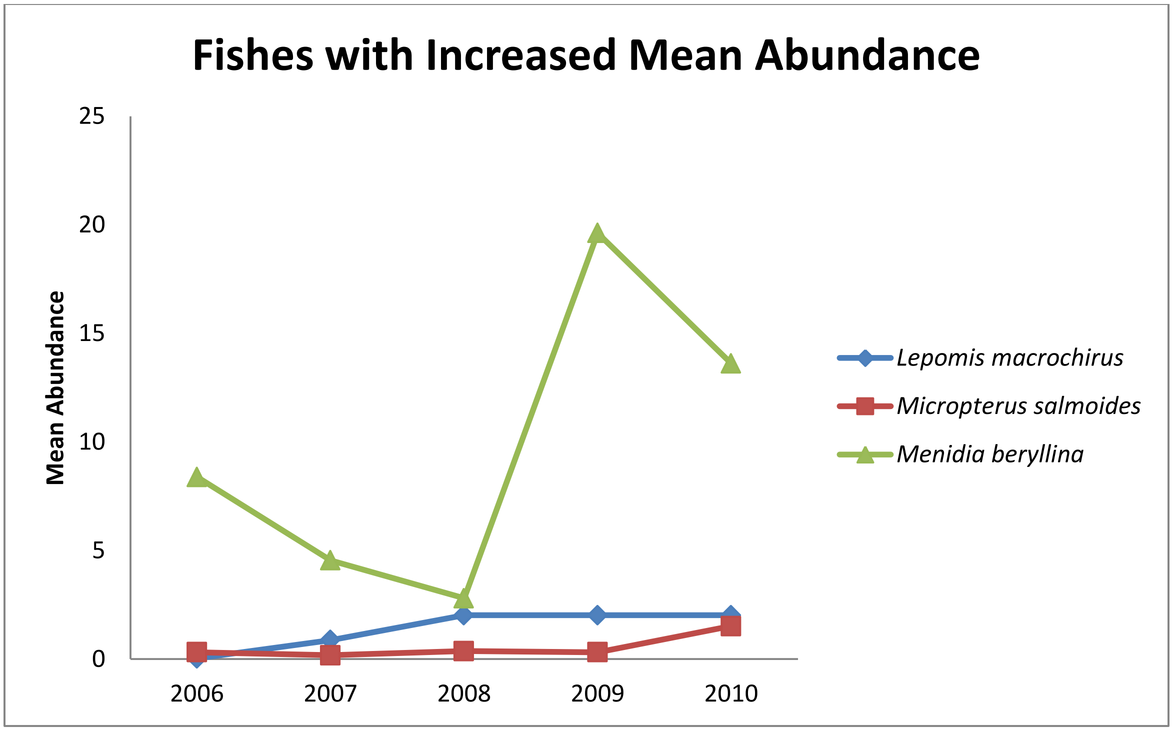

Figure 27: Chart of line graphs representing four species of fish whose mean abundances increased across years (2006 – 2010) from a seining survey.

Discussion

Based on my comparisons of taxonomic distinctness, fish assemblages in CPLL appear stable except during the years immediately following Hurricane Katrina. Before the 2006 – 2010 seining survey, only the variation in taxonomic distinctness during the electrofishing survey in 2002 was significantly higher than expected. A higher than expected variation in Λ+ suggests that the difference in taxonomic distinctness varies markedly more than expected among the species from this survey. This would suggest that the electrofishing survey in 2002 found a wider variety of species, indicating a more taxonomically diverse assemblage than expected.

The only survey in City Park before 2002 was a shoreline survey using multiple gears from 1971 – 1972. Its levels of Λ+ and Δ+ were not significant, suggesting a taxonomically diverse assemblage. Combining this with the shoreline and pelagic electrofishing survey from

2002 suggests one of two things. Either the pelagic zone in City Park has maintained a high level of taxonomic diversity from 1971 to 2002 or the variation in taxonomic distinctness over multiple habitats (pelagic and shoreline) increased in this same time period. Interpretation of both Δ+ and Λ+ does not indicate any significantly higher taxonomic diversity for any other survey. Therefore, it seems most likely that the pelagic assemblage was more taxonomically diverse, across years.

In the years following Hurricane Katrina, results suggest that taxonomic distinctness decreased in the shoreline habitats of CPLL. For the first two years (2006 - 2007) of the shoreline seining survey, average taxonomic distinctness was significantly lower than expected. In 2006, significantly higher Λ+ values were found, suggesting the species from this survey had a low average diversity that varied more than expected. In 2007, a significantly lower Δ+ was found. This is probably the result of impacts from Hurricane Katrina. Over all studies, lower salinity values have been documented in City Park than BSJ (Cali, 1972; O'Connell, unpublished data). This suggests that a more freshwater assemblage may have naturally evolved in CPLL. It seems likely that an initial drop in taxonomic diversity would occur in areas with historically low salinities following 2 to 3 weeks of saltwater inundation following Hurricane Katrina. Significantly reduced freshwater assemblages have been seen in other, more natural areas within the Lake Pontchartrain Basin (Van Vrancken and O'Connell, 2010), and in other regulated ecosystems (Atchafalaya River basin; Perret *et al.*, 2010) following Hurricane Katrina.

For all the years following Hurricane Katrina, average taxonomic distinctness was significantly lower than expected for shoreline habitats in Bayou St. John, suggesting lower than expected taxonomic diversity. Since higher salinities are typically found in BSJ, it would seem that it would be more capable at handling the effects of Hurricane Katrina. One of the two pre-

Katrina shoreline surveys also indicates reduced taxonomic distinctness for BSJ. During the survey from 1971-1972 lower than expected average taxonomic distinctness values were found, suggesting that shoreline habitats may have historically and currently been less taxonomically diverse. However, a significant value for average taxonomic distinctness was not found in the 1981-1982 seining survey. This study is the only study that also sampled outside of the impoundment between Lake Pontchartrain and BSJ (Ward, 1982). The lack of a lower than expected average taxonomic distinctness seen here may be a result of sampling waters outside of impounded BSJ. The area outside of the impoundment could be considered either Lake Pontchartrain or BSJ. When comparing studies over a similar temporal period, many more species were collected in Lake Pontchartrain than BSJ (O'Connell *et al*., 2004, current study). Overall, results suggest that the shoreline habitats in BSJ have been and still exhibit low taxonomic diversity. Unlike the shoreline habitats in BSJ, pelagic surveys did not show reduced taxonomic diversity. Both pelagic surveys, pre-Katrina in 1982 and post-Katrina in 2010-2011, did not indicate significantly different average taxonomic distinctness or variation in taxonomic distinctness values. These results suggest that shoreline habitat appears to affect the diversity of fishes in BSJ more than any other factor. Concrete stabilization of banks along with nearby road traffic may be causes of this.

There was not a significant difference in dissimilarity based on a nested two-way ANOSIM between Areas (CPLL and BSJ) using Sites as subgroups during the seining survey from 2006 to 2010. This suggests homogeneity between these groups following Hurricane Katrina. However, significant differences were found among sites across years and across months. This suggests that the shoreline habitats at each site support different groups of fishes while the overall fish assemblage between areas does not. The difference among Sites with no difference between Areas suggests that the 3 to 4 week period in which CPLL and BSJ were essentially one body of water may have had lasting effects. Since no standardized assemblage driven surveys exist before this event, it is difficult to determine if Hurricane Katrina caused this or these Areas already exhibited similar fish assemblages.

Pairwise ANOSIM tests between sites across Years indicate significant differences between each Site. However, the pairwise test across Months for Pontchartrain Lagoon and Metairie Bayou does not indicate a significant difference. This evidence does seem to go against the theory that Hurricane Katrina created one homogeneous fish assemblage with different groups of fishes found among sites. However, these sites are the closest sites geographically (0.70 km) and this may be the reason why there is no significant difference between these sites across months. A Global R value close to 0 was generated, and this also indicates that these assemblages are similar.

SIMPER analysis indicated that only a few of the twenty-seven species were responsible for the majority of the change observed between years. The overall trends among these species are five species from the Order Cyprinodontiformes decreased steadily across years, while average abundances of *M. beryllina*, *L. macrochirus*, and *M. salmoides* increased across years. Poeciliids have been shown to have cyclical abundance patterns, but a decline across years in the three (*P. latipinna*, *G. affinis*, and *H. formosa*) most abundant species of the family is probably not the result of a natural population cycle (Shoemaker, 1944; Silliman, 1948; Rose, 1959). Following Hurricane Katrina, *M. salmoides* were stocked in CPLL and BSJ. This stocking could have resulted in increased mean abundance. No evidence supporting cyclicity of *C. variegatus* and *L. parva* abundances was found by the author. Therefore, the decrease of these organisms is difficult to explain. Also, no records of stocking *L. macrochirus* were made available to the author and stocking of *M. beryllina* seems unlikely. Increases in these two organisms are also difficult to understand. One of the possible causes of this could be a response to the invasive *Herichthys cyanoguttatus* (Rio Grande Cichlid) that was sampled throughout this survey. Its effects were not analyzed in taxonomic distinctness, ANOSIM, or SIMPER analyses because the focus of this study was to understand the native fish assemblage.

Conclusions

City Park Lakes and Lagoons have maintained relatively stable taxonomic diversity across all surveys, except for the shoreline assemblage following Hurricane Katrina. In 2006 and 2007 lower than expected average taxonomic distinctness values were observed suggesting that impacts from the Hurricane may have negatively affected the shoreline assemblages in this area. Any surveys from before Hurricane Katrina and after indicated that there was no reduced diversity. Differences in shoreline fish groups were seen for these sites across months and years, except for two close sites. This suggests difference in microhabitats support different assemblages.

Bayou St. John's shoreline fish assemblage exhibited reduced taxonomic diversity across years. All surveys on pelagic habitats in BSJ suggest they are healthy. These data suggest that Hurricane Katrina may not have affected BSJ in the same way as CPLL. BSJ's lack of appropriate shoreline habitat makes determination of the effects of Hurricane Katrina on BSJ fish assemblage difficult. Significant differences among all sites across years and months suggest microhabitats within BSJ support different assemblages.

From 2006-2010, I measured a reduction in the mean diversity of five Cyprinodontiformes and an increase in *M. beryllina*, *L. macrochirus*, and *M. salmoides*. These results are puzzling and may in part be due to stocking practices, the cyclicity of Poeciliids, reduced habitat variation, the invasive *H. cyanoguttatus*, or some combination of these four reasons.

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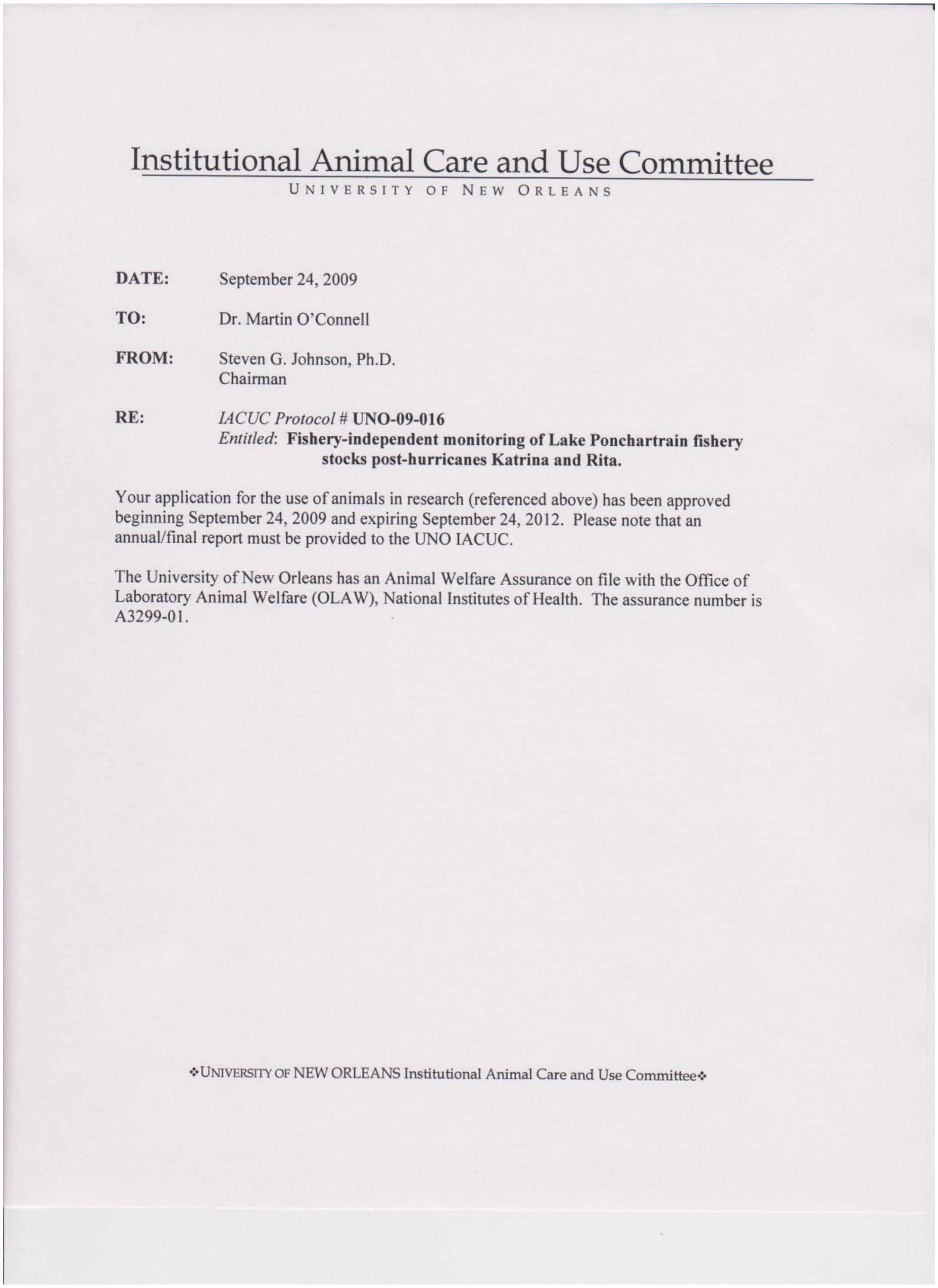
Appendix I – IACUC Approval



Institutional Animal Care and Use Committee
UNIVERSITY OF NEW ORLEANS

DATE: September 24, 2009

TO: Dr. Martin O'Connell

FROM: Steven G. Johnson, Ph.D.
Chairman

RE: *IACUC Protocol #* **UNO-09-016**
Entitled: **Fishery-independent monitoring of Lake Ponchartrain fishery stocks post-hurricanes Katrina and Rita.**

Your application for the use of animals in research (referenced above) has been approved beginning September 24, 2009 and expiring September 24, 2012. Please note that an annual/final report must be provided to the UNO IACUC.

The University of New Orleans has an Animal Welfare Assurance on file with the Office of Laboratory Animal Welfare (OLAW), National Institutes of Health. The assurance number is A3299-01.

❖UNIVERSITY OF NEW ORLEANS Institutional Animal Care and Use Committee❖

Vita

Patrick William Smith was born on 6 February 1985 in Newnan, Georgia and grew up in Augusta, GA. He received his Bachelors of Science degree in Biology from Augusta State University in May 2008. In 2009, Patrick worked as a Research Scientist with the Southeastern Natural Sciences Academy, and married his wife, Megan Greer in October. He was accepted into Martin T. O'Connell's Nekton Research Lab at the University of New Orleans in January 2010.