USE OF TERRACED MARSH HABITATS BY ESTUARINE NEKTON IN SOUTHWESTERN LOUISIANA

A Thesis

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by
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for Papa

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ABSTRACT

A variety of techniques have been employed in attempts to mitigate the extensive wetland loss occurring in coastal Louisiana. Marsh terracing is a wetland restoration technique that has rapidly gained in popularity in recent years. Terraces are assumed to benefit coastal restoration by providing areas for emergent plant growth, reducing wave energies, and increasing edge habitat to support nekton communities. The objectives of this study were to: 1) determine the effect of marsh terraces on adjacent water quality and sediment characteristics, 2) compare nekton abundance, species richness, and diversity in edge and open water habitats within terraced and unterraced ponds, and 3) compare the condition of numerically dominant fishes between terraced and unterraced ponds as an indicator of habitat quality. Three study sites located in southwest Louisiana at Sabine National Wildlife Refuge and Rockefeller State Wildlife Refuge were selected for the study. Each study site consisted of a terraced pond and a nearby unterraced reference pond. Nekton was quantitatively sampled in four different habitat types at each study site with a 1-m^2 throw trap. The habitat types sampled were: 1) terraced marsh edge, 2) unterraced marsh edge, 3) open water within terraced ponds, and 4) open water within unterraced ponds. Nekton density ($P = 0.0004$), biomass ($P = 0.002$), species richness ($P = 0.0007$), and diversity (H', $P = 0.01$) (1-D, $P = 0.007$) were all significantly greater at terraced edge habitats (treatment) as compared to unterraced open water habitats (control). There was no significant difference in these variables between terraced and unterraced edge habitats. While terraced pond habitats were superior to pre-restoration conditions in terms of nekton habitat value, they lacked functional equivalency with comparable unterraced ponds in several areas: 1) nekton community composition differed

between terraced and unterraced edge habitats, and 2) several fish species were found to be in poorer condition in terraced ponds as compared to unterraced ponds. A lack of functional equivalency between terraced and unterraced habitats may be partially attributable to the relatively young age of the terraces studied, as many functions of created marshes may take years to develop.

INTRODUCTION

Background

Louisiana's coastal wetlands encompass an area of approximately 3 million acres (12,141 km^2) and provide valuable functions for the state's economy, culture, and natural environment. Louisiana's economy is linked to its wetlands. Commercial and recreational fishing in Louisiana is a $3.5 billion per year industry, supporting an estimated 40,000 jobs (Louisiana Coastal Wetlands Conservation and Restoration Task Force 2003). Approximately one third of the fish commercially harvested in the lower 48 states come from Louisiana's coastal zone (USDOC 2001). Wetlands also provide habitat for waterfowl and wildlife. Located at the southern terminus of the Mississippi Flyway, Louisiana's wetlands provide winter habitat for over 5 million migratory waterfowl as well as stopover habitat for neotropical migratory songbirds and other avian species (USACE 2004). Wetlands also serve as a buffer from storms, protecting the state's communities, ports, and oil and gas infrastructure from the damaging effects of hurricanes and tropical storms.

Over the past 7,000 years, successive patterns of land loss and gain have occurred in association with Mississippi River delta cycle processes with an average net gain in coastal wetlands (Boesch et al. 1994). However, over the last century Louisiana has experienced a net loss in coastal wetland area with more than 1.2 million acres (4,856 km^2) of land lost since the 1930's. The annual rate of land loss increased from 36 km^2/yr during the 1940s to over 100 km^2/yr during the 1970s (Boesch et al. 1994). While the rate of loss has declined in recent years (61.3 km^2/yr; Barras et al. 2003), Louisiana still accounts for over 80% of the total national coastal wetland loss despite the fact that only 40% of the nation's coastal wetlands are located in Louisiana (Boesch et al. 1994).

Coastal wetland loss results from a combination of natural processes such as subsidence, sea-level rise, storms, and geosynclinal downwarping combined with human-induced factors such as canal and levee construction and subsurface fluid withdrawal (Penland et al. 1990). In recent years, wetland loss has occurred predominantly through a combination of coastal erosion and marsh degradation (Penland et al. 1990, Barras et al. 2003). Coastal erosion involves the retreat of the exposed shorelines of large lakes, bays, and the Gulf of Mexico while marsh degradation often occurs through the development of small ponds within interior marshes, which is hypothesized to result from a lack of vertical accretion in the marsh, vegetation decline, and reduction of surface elevation, followed by further plant death and enlargement of these ponds (Delaune et al. 1994).

Wetland Restoration and Creation

Early recognition of the important role wetlands play in both the ecosystem and economy of Louisiana, the United States, and the world is evidenced by an international treaty focused solely on the conservation of wetlands (UNESCO 1971) as well as a U.S. policy of "no net loss" that has stimulated significant wetland restoration and creation activities. In general, wetland creation and restoration projects are designed to mitigate for the loss of ecosystem services resulting from the loss or degradation of wetlands (Kusler and Kentula 1990, Mitsch and Gosselink 2000, Craft et al. 2003). Specific project goals may include re-establishing natural hydrologic, geochemical, and ecological functions such as storm-water and nutrient retention and providing fish and wildlife habitat, and vary as a result of the type of wetland being restored and the reason for restoration (Simenstad and Thom 1996, Craft et al. 1999, Mitsch and Gosselink 2000). For example, the creation of smooth cordgrass *Spartina alterniflora* salt marshes has often been used with the goals of creating estuarine habitat, stabilizing dredge material, reducing shoreline erosion, and mitigating wetland loss (Craft et al. 1999).

Numerous techniques have been implemented in an attempt to mitigate wetland loss; vegetative plantings, creation of impoundments, use of dredge material, backfilling of canals, and re-distribution of spoil banks are a few of the more commonly used techniques (Boesch et al. 1994, Turner and Streever 2002). Success of these projects has been mixed as restoration is still considered to some extent an "art" and projects should be designed to be site-specific, taking into account such factors as marsh elevation, salinity regime, and soil characteristics (Steyer 1993, Delaney et al. 2000, Shafer and Streever 2000). In Louisiana, where the need for successful marsh restoration is vital to the state, managers are constantly testing new restoration techniques: thin-layer spray dredging, and freshwater and sediment diversions at both small and large scales are three restoration techniques that are currently popular in Louisiana. In wetlands suffering from significant marsh degradation as evidenced by large newly formed ponds, marsh terracing has become popular as a means of marsh habitat restoration in these interior ponds. The success or failure of many of these techniques in Louisiana is of prime importance to not only the state's ecosystems, but also its economy.

Measuring Success and Functional Equivalency

Despite the acknowledged importance of wetland systems, the critical need for protection and restoration of degraded systems, and the estimated millions of dollars invested in wetland restoration projects, evaluation of wetland restoration success nationwide has been limited because of a lack of long-term monitoring after construction (Race and Fonseca 1996, Zedler 2000). In many instances, determination of success is often based solely on establishment or creation of vegetated marsh areas without any quantitative measure of other metrics to assess "functional equivalency" (Moy and Levin 1991, Simenstad and Thom 1996, Zedler 2000, Windham et al. 2004). Functional equivalency refers to the fact that, rather than simply looking like natural wetlands, constructed or restored marshes should perform functions (e.g., improve water quality, protect shorelines, provide fish and wildlife habitat) similar to those performed by natural wetlands. While limited, there are quantitative studies that have examined aspects of restored wetlands in order to determine functional equivalency trajectories of restored wetlands (Simenstad and Thom 1996, Zedler and Callaway 1999, Morgan and Short 2002). Based on current information, research indicates that the various ecological processes responsible for these functions may develop at different speeds. For example, above and below-ground biomass of *Spartina* is often similar between restored and reference marshes within 3-5 yrs of construction (Broome et al. 1986, Craft et al. 1999): benthic invertebrate communities of restored marshes may take 10-15 yrs to reach equivalency with reference sites (Craft et al. 1999) while soil properties such as organic C and N, may take more than 30 yrs to develop (Craft et al. 1999).

Data on the ability of restored marshes to provide equivalent nekton habitat over time has been more mixed. Several studies comparing natural salt marshes and created salt marshes of various age have found that densities or abundances of nekton were generally lower in constructed marshes (Minello and Webb 1997, Minello 2000). In contrast, other studies have found comparable abundances of nekton between natural and constructed marshes (Williams and Zedler 1999, Rozas and Minello 2001, Havens et al. 2002, Jivoff and Able 2003, Able et al. 2004).

While nekton density, abundance and community composition represent means of measuring functional equivalency or perhaps habitat quality for nekton, another less frequently used approach involves the use of biotic indices of condition. Several studies have used condition indices to assess the fish health in different habitats (Mustafa 1978, Burke et al. 1993, Gilliers et al. 2004). Mustafa (1978) used Fulton's condition factor to compare the condition of pond and channel-dwelling populations of the common minnow *Esomus danricus*. Gilliers et al. (2004) used biochemical, morphometric, and recent growth indices to evaluate habitat quality of different nursery grounds for juvenile flatfish. Burke et al. (1993) used morphological indices of Atlantic croaker *Micropogonias undulatus* as indicators of habitat quality with respect to estuarine pollution. Assessing functional equivalency of restoration projects in coastal Louisiana is of primary importance to ensure the success of Louisiana's $14 billion proposed restoration plan. As marsh terracing is one of the more popular restoration techniques currently being used in southwestern Louisiana, we have a unique opportunity to measure the "equivalency" of terraced and unterraced ponds, and to provide guidance to managers in the design and continued construction of marsh terraces.

Marsh Terracing

Marsh terracing, sometimes called bay bottom terracing, is a restoration technique that was first employed in the United States at Sabine National Wildlife Refuge in Cameron Parish, Louisiana in 1990 (Underwood et al. 1991) and has gained in popularity since then, especially in the coastal wetlands of the Chenier Plain. At least 201 km of terraces had been constructed in Louisiana as of the spring of 2004 (Nyman pers. communication). Terraces are ridges or levees of discontinuous marsh that are created by excavating subtidal sediments on-site and forming them into narrow ridges that are created at marsh elevation to be flooded at high tide. The ridges are then planted with marsh vegetation, usually *Spartina alterniflora* (Fig. 1.1). Openings are left between terraces to allow for tidal exchange and the movement of nekton throughout the pond. Terrace fields are created in a variety of patterns meant to maximize intertidal edge and minimize fetch between terraces (Rozas and Minello 2001). Several spatial arrangements including checkerboard, linear, and duckwing designs have been used in coastal Louisiana.

Figure 1.1. Terraces under construction (left), recently planted (center), and completely vegetated (right) at Sabine National Wildlife Refuge.

Terraces are hypothesized to benefit restoration by reducing wave energy, decreasing turbidity, promoting deposition and retention of suspended sediments, creating marsh edge habitat, and increasing primary and secondary productivity (Underwood et al. 1991, Steyer 1993). Nekton are hypothesized to benefit from terracing through the creation of marsh edge habitat and the hypothesized regeneration of submerged aquatic vegetation (SAV). Both marsh edge and SAV habitat are associated with higher densities of nekton as compared to open water and non-vegetated shallow open water sites as a result of increased food and refuge provided by these habitats

(Zimmerman and Minello 1984, Boesch and Turner 1984, Rozas and Odum 1988, Baltz et al. 1993, Minello et al. 1994, Peterson and Turner 1994).

Four previous studies have examined some aspect of the success of marsh terracing in providing functions equivalent to natural marshes. As is often the case in restoration success studies, these studies focused on single terrace restoration projects, and results are thus site-specific. The first monitoring study of marsh terraces in Louisiana found that terracing increased the amount of marsh edge within the ponds, reduced wave heights, and increased primary productivity (Steyer 1993). Two more recent studies focusing on nekton abundances in terraced sites found significant differences in nekton densities between terraced and unterraced habitats (Rozas and Minello 2001, Bush Thom et al. 2004). Rozas and Minello (2001) concluded that increasing the proportion of marsh edge within terrace fields enhances the habitat value for fishery species, suggesting that the creation of marsh edge was enhancing the nekton value of the restored sites. Bush Thom et al. (2004) found significantly higher density, biomass, and diversity of nekton at terrace edge sites as compared to open water sites, but that the species composition of nekton communities at terrace edge were significantly altered compared to unmanaged marsh edge. Most recently, using GIS models, Rozas et al. (2005) concluded that terraced ponds supported high populations of blue crab *Callinectes sapidus*, brown shrimp *Farfantepenaeus aztecus*, and white shrimp *Litopenaeus setiferus* relative to pre-restoration conditions.

<u>Study Goals</u>

This study was designed to examine multiple terraced ponds and adjacent unterraced ponds in order to draw more complete conclusions as to the general effectiveness of marsh terraces in creating quality nekton habitat. The objectives of this study were to: 1) determine the effect of marsh terraces on adjacent water quality and sediment characteristics, 2) compare nekton abundance, species richness, and diversity in edge and open water habitats within terraced and unterraced ponds, and 3) compare condition of numerically dominant fishes between terraced and unterraced ponds as an indicator of habitat quality.

METHODS

<u>Study Area and Site Descriptions</u>

Data were collected in Louisiana's Chenier Plain, which is located along the southwestern coast from Vermillion Bay to the western boundary of the state. The Chenier Plain was formed by the deposition of fine-grained Mississippi River sediments carried by westward-moving currents along the coast (USACE 2004). The area is characterized by alternating shore-parallel ridges, or "cheniers," and mudflats (Penland and Suter 1989). Historically, land loss rates for the Chenier Plain have been lower than for the Deltaic Plain. Land loss in the Chenier Plain is due largely to the breakup of interior marshes, although shoreline erosion is a problem around large lakes and the Gulf of Mexico (Barras et al. 2003). Three hydrologically separate pairs of terraced and unterraced ponds located at Rockefeller State Wildlife Refuge (RWR) and Sabine National Wildlife Refuge (SWR) in Cameron Parish, Louisiana were sampled.

Sites 1 and 2 were located at RWR (Fig. 2.1), which is situated on the Gulf Coast Chenier Plain in Cameron and Vermillion Parishes and encompasses a 30,700-ha area managed by the Louisiana Department of Wildlife and Fisheries. Situated between Louisiana Highway 82 to the north and the Gulf of Mexico to the south, the refuge consists of 17 impoundments (units) as well as approximately 11,700 ha of unimpounded, tidally-influenced marsh (Wicker et al. 1983). Water levels in the impoundments are controlled by various water control structures such as flap gates, weirs, and gated culverts. Management activities at RWR include regular controlled burning of marsh vegetation and structural marsh management. Marsh types at RWR range from saline marsh along the Gulf of Mexico to intermediate marsh farther to the north.

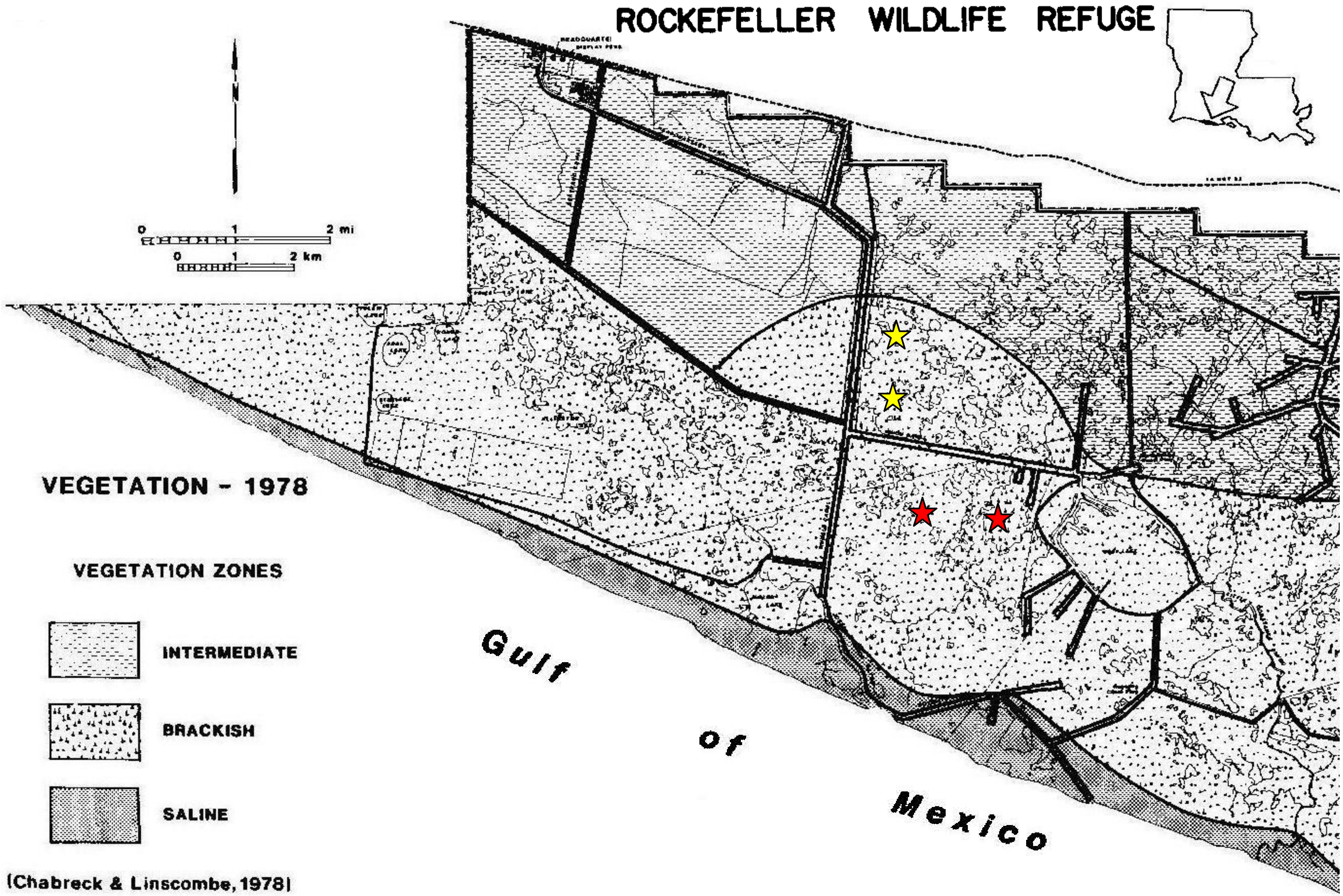


Figure 2.1. Rockefeller State Wildlife Refuge in southwest Louisiana. Stars represent location of study ponds for Site 1 (yellow) and Site 2 (red).

Site 1 – Site 1 was located in Unit 4 of RWR (Fig. 2.2), which is a 2,400-ha impoundment that is managed via 2 variable-crest flap-gated structures. The area is a brackish marsh dominated by saltmeadow cordgrass *Spartina patens* (Flynn et al. 1999). The terraced pond at site 1 was located near the western edge of the impoundment and is approximately 0.32 km^2 in size. The terraces, constructed in 2001, are arranged in a duck-wing or chevron-shaped pattern and were planted with *Spartina alterniflora*. The unterraced pond at site 1 is approximately 0.65 km^2 in size and is located to the south of the terraced pond.

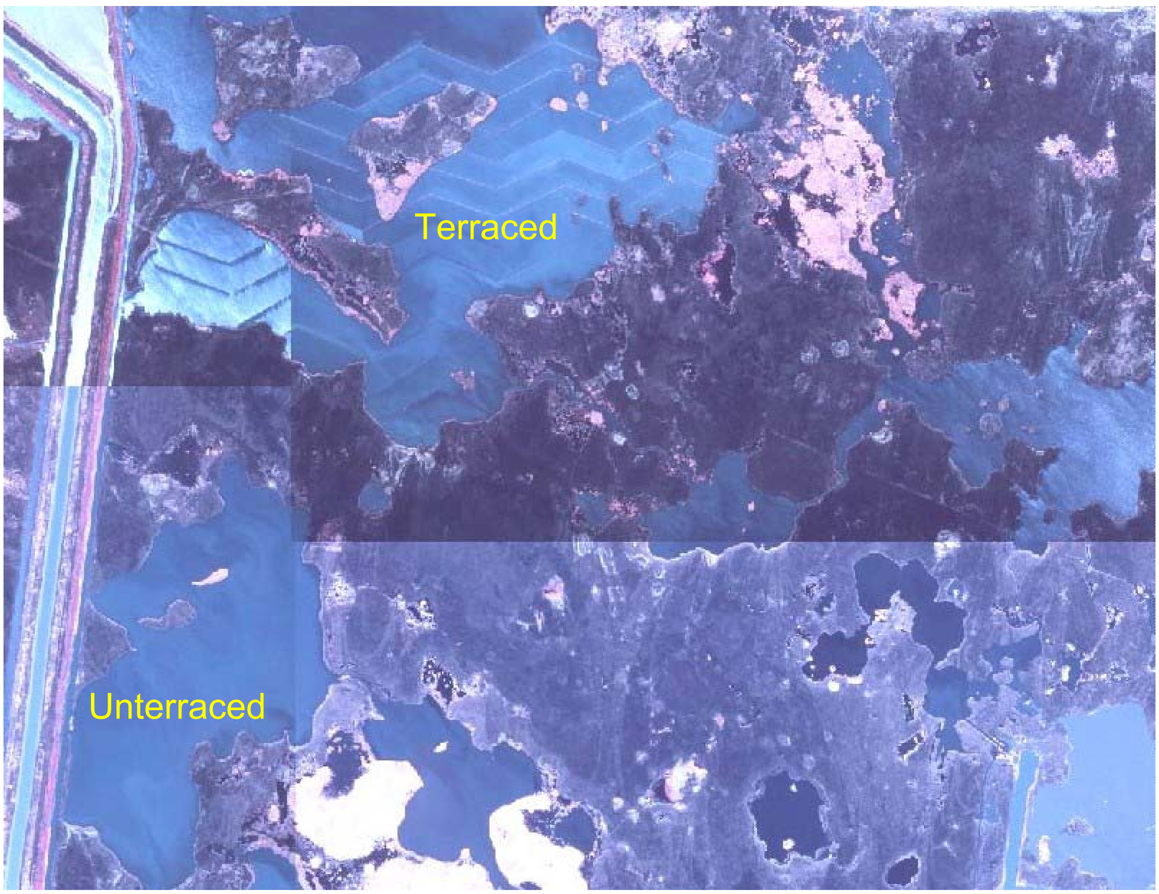


Figure 2.2. Site 1 located in western Unit 4 of Rockefeller Wildlife Refuge.

Site 2 – Site 2 was located in Unit 5 of RWR (Fig 2.3), which is a 1,982-ha impoundment directly south of Unit 4. The area is composed of brackish marsh dominated by *Spartina patens*. Levees are constructed around 3 sides of the impoundment while the southern end is open to the Gulf of Mexico. The terraced pond at site 2 is located in the northern portion of Unit 5 west of Deep Lake and is approximately 0.59 km^2 in size. The terraces were constructed in 2000 in a linear pattern. The terraces were planted with *Spartina alterniflora*. Since their construction, the Unit 5 terraces have degraded severely (Fig. 2.4). At the end of the study most of the emergent vegetation was no longer present and many of the terraces had eroded to the point that they were no longer visible above the water surface. Only 2 complete terraces along the western border of the pond and several terrace fragments throughout the pond were present at the completion of the study. The unterraced pond at site 2 is located to the west of the terraced pond and is approximately 0.51 km^2 in size.

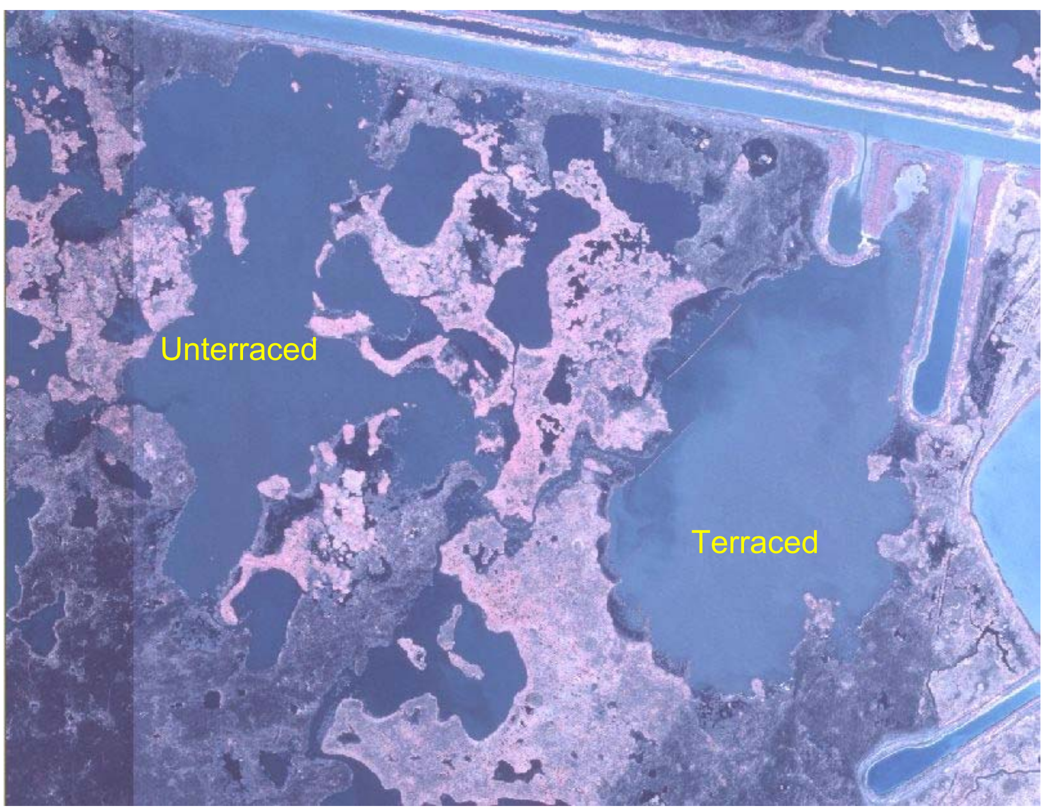


Figure 2.3. Site 2 located in northern Unit 5 of Rockefeller State Wildlife Refuge.

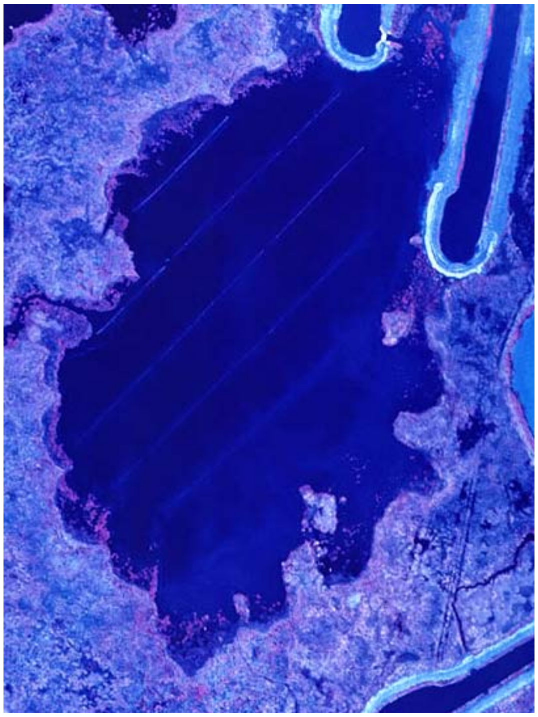



Figure 2.4. Terraced pond at Site 2. Note the degradation of the terraces between 2001 (left) and 2004 (right).

Site 3 was located at SWR (Fig. 2.6), which is located in Cameron Parish, Louisiana between Sabine and Calcasieu Lakes. The refuge encompasses 50,388 ha (USFWS, 2002) and is managed by the U.S. Fish and Wildlife Service. SWR is divided into management units by a system of canals and levees. Marsh types on the refuge range from saline to fresh. Management activities at SWR include prescribed burning, marsh impoundments, and water management for salinity control.

Site 3 –The terraced pond at site 3 is located in Unit 7 of SWR (Figure 2.5). The area is composed of brackish marsh dominated by *Spartina patens*. The terraced pond is approximately 4.82 km^2 in size. The Unit 7 terraces were constructed in 2001 in a duck-wing pattern and were planted with *Spartina alterniflora*. The unterraced pond is approximately 12.60 km^2 in size and is located in Unit 5 of SWR to the north of the terraced pond near the area known as Greens Lake. This area is also brackish marsh dominated by *Spartina patens*, but with small patches of common reed *Phragmites australis* interspersed throughout.

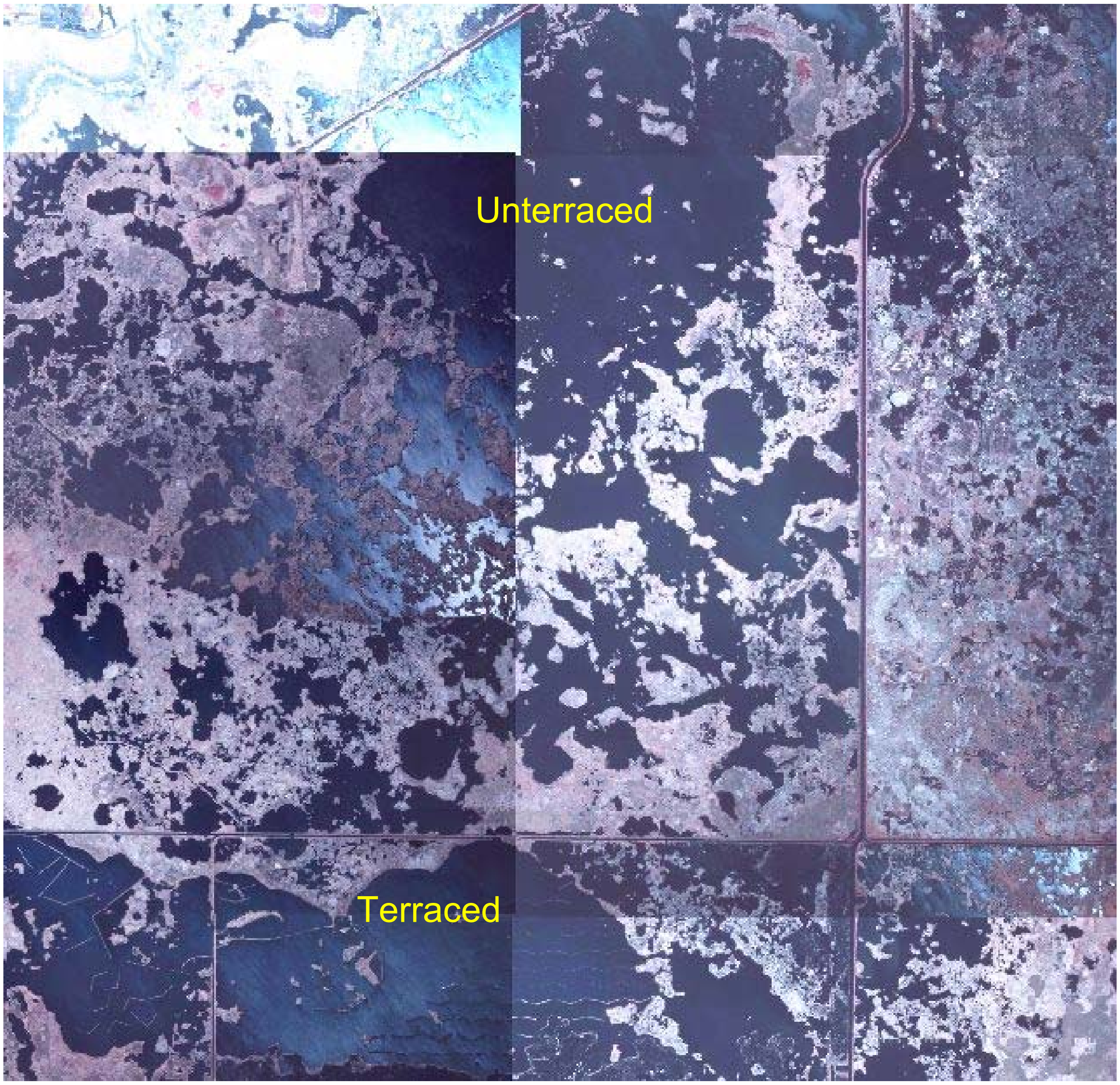


Figure 2.5. Site 3. The terraced pond is located in Unit 6 of SWR; the unterraced pond is located to the north in Unit 5.

Sampling Design

The three study sites each contained two hydrologic units, a terraced pond and a nearby unterraced reference pond. Sampling was conducted for four habitat types 1) terraced marsh edge, 2) unterraced marsh edge, 3) open water within terraced ponds, and

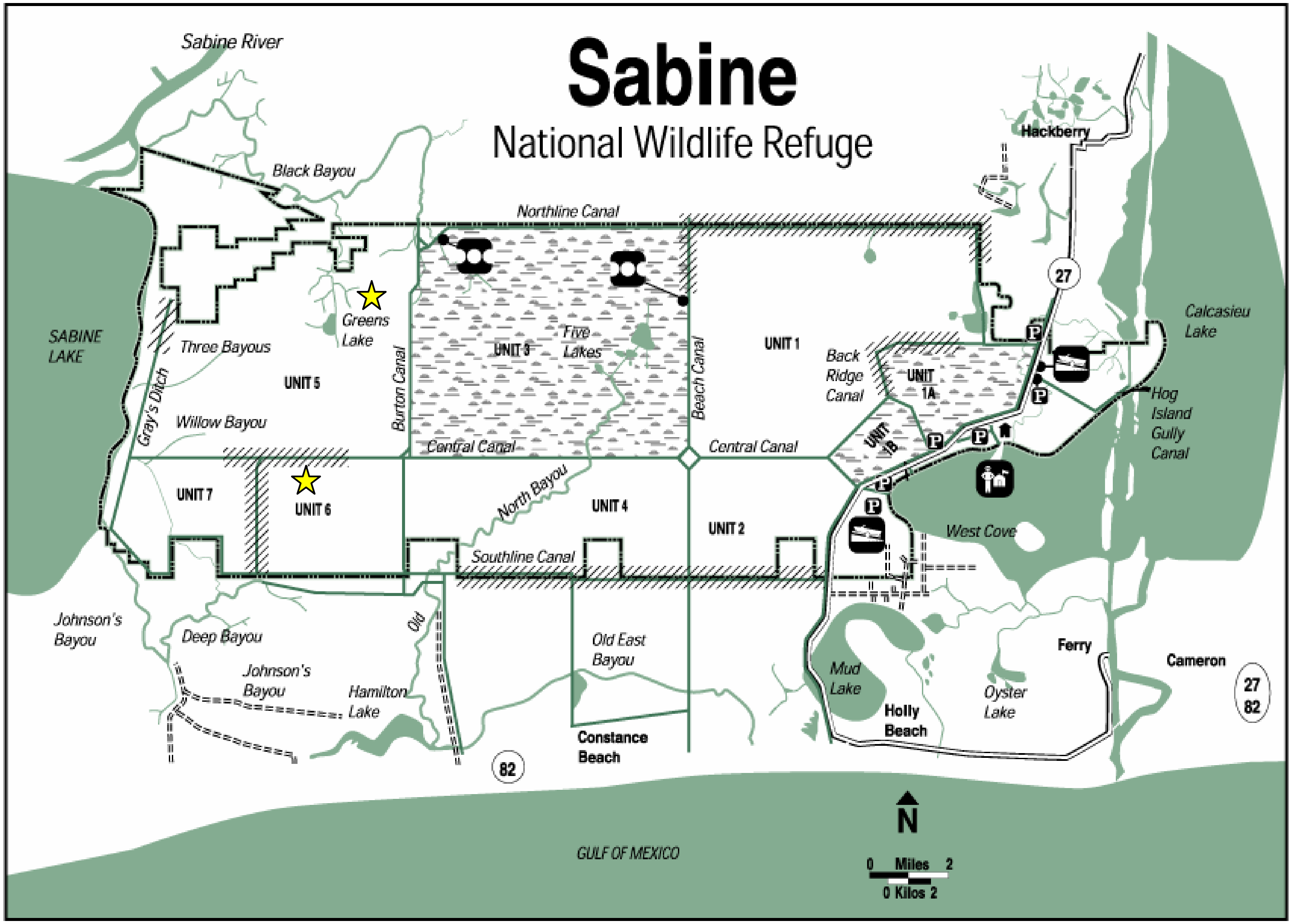


Figure 2.6. Sabine National Wildlife Refuge. Stars represent the location of study ponds for Site 3.

4) open water within unterraced ponds. Within each terraced pond, there were four sampling stations, two at randomly selected points along the terrace edge (defined as the waterward area within 1 m of emergent vegetation on the terrace), and two at randomly selected open water points (defined as any point greater than 50 m from the nearest terrace). Within each unterraced pond, there also were four sampling stations, two at randomly selected points along the natural marsh edge (defined as the waterward area within 1 m of the emergent vegetation on the marsh edge), and two open water points (defined as any point greater than 50 m from the marsh edge). Therefore, each site had 8 nekton samples (2 terraced edge, 2 unterraced marsh edge, 2 terraced open water, and 2 unterraced open water) per sample period, for a total of 24 samples for the 3 sites.

Sampling points were randomly selected by overlaying a grid over georeferenced Digital Orthophoto Quarter Quadrangle (DOQQ) aerial maps of the study ponds using ArcView 3.2 (ESRI, Redlands, CA). A random number generator was used to select numbers corresponding to squares within the grid. GPS coordinates obtained for each sampling point were used to locate points in the field.

Environmental Characteristics

Water quality data were collected along with each nekton sample. Salinity (ppt), conductivity (mS/cm), temperature (ºC), and dissolved oxygen (mg/L) were measured with a YSI model 556 multiparameter water quality meter (Yellow Springs Instruments, Yellow Springs, OH). Turbidity (NTU) was measured with a Turner Designs Aquafluor turbidimeter (Turner Designs, Sunnyvale, CA). Water depth was determined by calculating the mean of three depth measurements (cm) taken within each throw trap sample.

Soil cores were collected in May, 2005 to be analyzed for organic matter content. Eight 5-cm diameter soil cores were collected from the top 5 cm of soil at each sampling point (64 cores per site, 192 cores total). Cores were stored on ice until processing. Upon returning to the laboratory, the soil cores from each sampling point were homogenized into one composite sample. Composite samples were placed in crucibles and dried at 60ºC in a forced air drying oven to constant weight. The samples were then ground with a mortar and pestle and split into 5 sub-samples. The sub-samples were weighed to the nearest 0.001 g (initial dry weight), fired in a muffle furnace at 500ºC for 4 h, and weighed again (final dry weight). Percent organic matter was calculated as:

$$\% \text{ Organic Matter} = [1 - (\text{final dry weight} / \text{initial dry weight})] \times 100.$$

Nekton Sampling

Nekton was quantitatively sampled at each sample station with a 1-m^2 throw trap. The throw trap is well suited for sampling nekton in shallow estuarine habitats because of the quantitative nature of the samples and its effectiveness in areas with emergent and submerged aquatic vegetation (Jordan et al. 1997, Rozas and Minello 1997). The throw trap used in this study was constructed similar to the one described in Kushlan (1981). The trap consisted of a 1-m × 1-m × 0.66-m aluminum frame with 1.6-mm knotless nylon mesh sides. To facilitate sampling in water greater than 0.66 m deep the nylon mesh was extended above the frame to a total height of 1.25 m. A 1-m^2 PVC square was integrated into the top of the extended netting and buoyed by net floats. Sampling points were approached slowly by airboat to minimize disturbance to the site. The trap was thrown from the bow of the boat and immediately pressed into the substrate to prevent any captured nekton from escaping. The interior of the throw trap was swept with a 1-m wide bar seine (1.6-mm mesh) to clear all nekton from the trap. The trap was considered cleared when 5 consecutive sweeps of the bar seine yielded no organisms. Samples were placed on ice for transport to the laboratory, where they were frozen until processing.

Upon returning to the laboratory, samples were sorted, identified to species or lowest feasible taxon, measured, counted to determine density (individuals/m^2), and weighed to determine biomass (g/m^2). Fish and shrimp were measured to the nearest 0.1 mm total length and crabs were measured to the nearest 0.1-mm carapace width. All nekton were weighed to the nearest 0.001 g wet-weight using an Ohaus Adventurer model top-loading laboratory balance (Ohaus Corp., Pinebrook, NJ).

Sampling began in April 2004 and occurred bi-monthly over a 12 month period (4/2004, 6/2004, 8/2004, 10/2004, 12/2004, 2/2005, and 4/2005) for a total of 7 sample periods. Due to logistical problems, Site 3 was not sampled in 10/2004 or 12/2004. Therefore a total of 152 samples were collected (N = 152).

Submerged Aquatic Vegetation Sampling

All submerged aquatic vegetation (SAV) was collected from each throw trap sample. Prior to nekton removal all above-ground SAV was collected by hand. SAV samples were placed on ice for transport to the laboratory where they were sorted according to species, dried in a forced air drying oven at 60ºC to constant weight, and weighed to the nearest 0.001 g dry weight to determine SAV biomass (g/m^2).

Nekton Diversity

Shannon diversity index (H') and Simpson's diversity index (1-D) were calculated for each throw trap sample (Magurran 1988). Shannon diversity index was calculated as:

$$H' = -\Sigma p_i \ln p_i$$

where p_i is the proportion of individuals found in the *i*th species. Simpson's diversity index was calculated as:

$$D = \Sigma p_i^2$$

because as D increases diversity decreases, Simpson's diversity index was expressed as $1 - D$. Samples containing less than 3 organisms were excluded from the diversity analyses.

<u>Fish Condition Indices</u>

As indicators of habitat quality, condition indices were calculated for the 7 most numerically abundant fish species collected. Fulton's condition factor and Relative condition factor were calculated for each individual fish. Both of these condition factors use length-weight relationships to draw conclusions about the condition or well-being of a fish (Anderson and Neumann 1996). Fulton's condition factor has been widely used to assess fish condition, however it has also been criticized because of its assumption of isometric growth (i.e. shape does not change as the fish grows) and because it is size dependent. Therefore, it is generally most useful when fish of similar size are used. Relative condition factor compensates for the fact that many fishes do not grow isometrically.

Fulton's Condition Factor (K) is calculated with the following equation:

$$K = (W/L^3) \times 100{,}000$$

where W is weight (g), L is length (mm), and 100,000 is a scaling constant meant to convert small decimals to mixed numbers for easier interpretation (Anderson and Neumann 1996).

Relative Condition Factor (K_n) is calculated by comparing the actual weight of a given fish to the predicted weight for a fish of the same length within the study population. By performing a simple linear regression of the logarithmically transformed length-weight data for a particular species the following relationship can be determined:

$$\log_{10}(W) = a' + b \times \log_{10}(L),$$

where a' is the y-intercept of the regression line, and b is the slope. By taking the antilogarithm of a', the constant a can be determined and used in the following power function:

$$W' = aL^b$$

to predict the weight of a fish of a given length. Finally, Relative Condition Factor is calculated for each fish as:

$$K_n = (W/W'),$$

where W is the actual weight of the fish and W' is the length-specific mean weight for a fish in the study population (Anderson and Neumann 1996). Mean K_n for a population will always be one. A value of K_n greater than one indicates a fish in above average condition relative to the other fish in the population while K_n less than one indicates a fish in below average condition.

Statistical Analyses

Data analyses were based on a randomized block design, blocking by site. Three-way Analysis of Variance (ANOVA, Proc MIXED) was used to examine differences in density, biomass, species richness, diversity (Shannon, Simpson), and environmental variables (except soil organic matter) among treatments. To meet conditions of normality, nekton density and biomass were ln(x + 1) transformed. The treatments selected for analyses were pond type (terraced or unterraced), habitat type (edge or open water), and sampling date (4/2004, 6/2004, 8/2004, 10/2004, 12/2004, 2/2005, and 4/2005). Two *a priori* contrasts were included in the ANOVA to test for differences between specific habitat types. Terraced edge was compared to the control, unterraced open water, in order to determine if terrace edges support more nekton as compared to pre-terrace conditions: terraced and unterraced edge habitats were compared to one another in order to determine if terraced edge habitat was equivalent, in terms of nekton support to "target" unterraced edge habitat. Simple Linear Regression (SLR, Proc REG) was used to determine the relationship between SAV biomass and nekton density. To meet conditions of normality, SAV biomass data were ln(x + 1) transformed. A Chi-square test was used to test for differences in nekton species composition by functional group (crustaceans, demersal fishes, benthopelagic fishes, and pelagic fishes) at each of the 4 habitat types sampled: 1) terraced edge, 2) terraced open water, 3) unterraced edge, and 4) unterraced open water. A t-test was used to compare soil organic matter content between terraced and unterraced edge habitats.

Length-weight relationships were calculated using all fish of a particular species regardless of the pond or habitat where the fish was collected. T-tests (Proc TTEST) were used to compare fish condition (K, K_n) for the seven most numerically abundant fish species by pond type. All condition indices were log transformed except for K for *Cyprinodon variegatus* and *Gambusia affinis*, and K_n for *Microgobius gulosis*. An alpha level of 0.05 was considered significant for all analyses. All statistical analyses were conducted with SAS statistical analysis software for PC (version 9.0; SAS Institute, Cary, North Carolina). All results presented are mean ± standard error.

RESULTS

Environmental Characteristics

Water quality characteristics were typical of brackish marsh environments (Table 3.1). No significant differences were found between terraced and unterraced ponds for depth, temperature, dissolved Oxygen, or turbidity. Salinity and conductivity were significantly lower in terraced ponds as compared to unterraced ponds. Seventy percent of throw trap samples contained SAV (107 of 152 samples) with SAV biomass significantly greater in terraced ponds as compared to unterraced ponds. Soil organic matter content was significantly lower at terraced edge as compared to unterraced edge. In comparing edge and open water habitats, depth was significantly lower at edge habitats (35.0 ± 1.7 cm) as compared to open water habitats (45.8 ± 2.3 cm) ($P < 0.0001$).

Table 3.1. Low, high, and mean (SE) values for environmental characteristics by pond type. Significant p-values are in bold type. * Soil organic matter includes only edge habitats.

Variable	Terraced			Unterraced			
	low	high	mean	low	high	mean	Pr>F
Depth (cm)	6.0	85.7	39.9 (2.1)	6.0	85.3	41.1 (2.1)	0.69
Salinity (ppt)	0.5	11.6	4.5 (0.4)	0.7	13.6	5.4 (0.5)	**0.01**
Conductivity (mS/cm)	1.0	19.5	8.0 (0.7)	1.3	22.1	9.5 (0.8)	**<0.0001**
Temperature (°C)	10.9	32.2	21.9 (0.6)	13.2	35.6	23.5 (0.7)	0.06
Dissolved Oxygen (mg/L)	2.6	12.0	9.0 (0.3)	4.1	12.6	8.0 (0.4)	0.25
Turbidity (NTU)	0.5	79.0	25.7 (2.4)	0.3	95.0	28.2 (3.4)	0.23
SAV biomass (g/m^2 dry wt.)	0.0	160.2	19.7 (5.5)	0.0	16.4	1.3 (0.4)	**0.003**
Soil Organic Matter (%)*	2.5	32.2	11.9 (1.7)	25.3	60.0	35.6 (2.1)	**<0.0001**

Nekton Species Composition

A total of 3,544 organisms were collected representing 25 taxa (Table 3.2). Total catch consisted of 57 % fish (2,033 individuals, 20 sp.) and 43 % crustaceans (1,511 individuals, 5 sp.). The most frequently collected fish species were rainwater killifish

Lucania parva (n = 465), sailfin molly *Poecilia latipinna* (n = 392), and inland silverside *Menidia beryllina* (n = 387). The most frequently collected crustacean species were daggerblade grass shrimp *Palaemonetes pugio* (n = 1,171), blue crab *Callinectes sapidus* (n = 150), and white shrimp *Litopenaeus setiferus* (n = 93).

The composition of total catch differed between terraced and unterraced ponds (Fig. 3.1). In terraced ponds, total catch consisted of 65 % fishes (1,061 individuals, 17 spp.) and 35 % crustaceans (562 individuals, 5 spp.). In unterraced ponds, total catch consisted of 51 % fishes (972 individuals, 14 spp.) and 49 % crustaceans (949 individuals, 5 spp.).

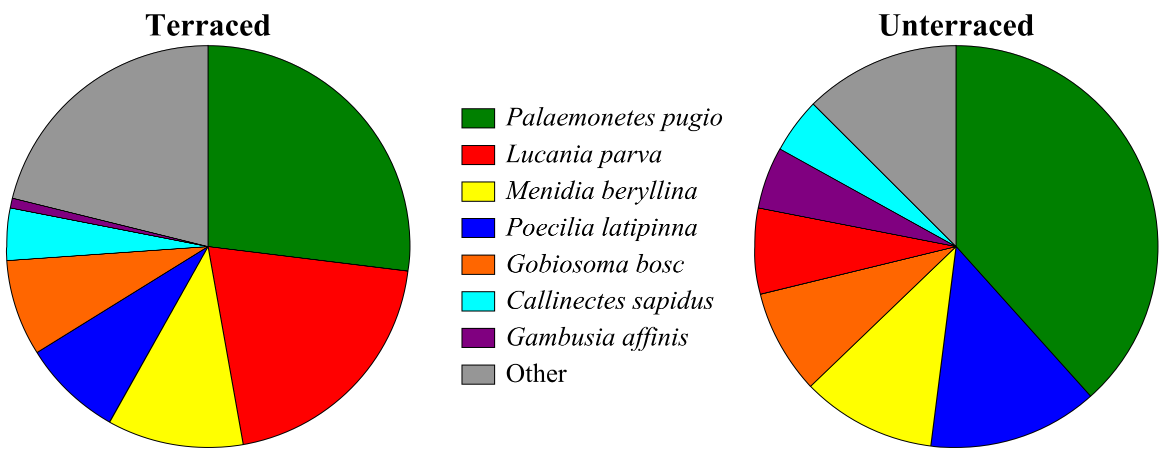


Figure 3.1. Species composition of samples collected in terraced (left) and unterraced (right) study ponds. Species with a total catch fewer than 100 individuals were placed in the "other" category.

<u>Nekton Species Composition – Functional Groups</u>

Distribution of nekton functional groups differed significantly among the four habitat types (Chi-sq: $P < 0.0001$) (Fig 3.2). Percent of total catch of crustaceans was higher at edge (>48%) as compared to open water habitats (<22%), and highest at unterraced (62%) as compared to terraced edge (41%). Percent total catch of pelagic fishes was higher in terraced (>28%) as compared to unterraced ponds (<21%), and highest at terraced open water habitats (46%). Percent total catch of benthopelagic fishes was highest at unterraced open water habitats (46%). Demersal fishes made up a relatively consistent proportion of total catch across habitat types (8-17%).

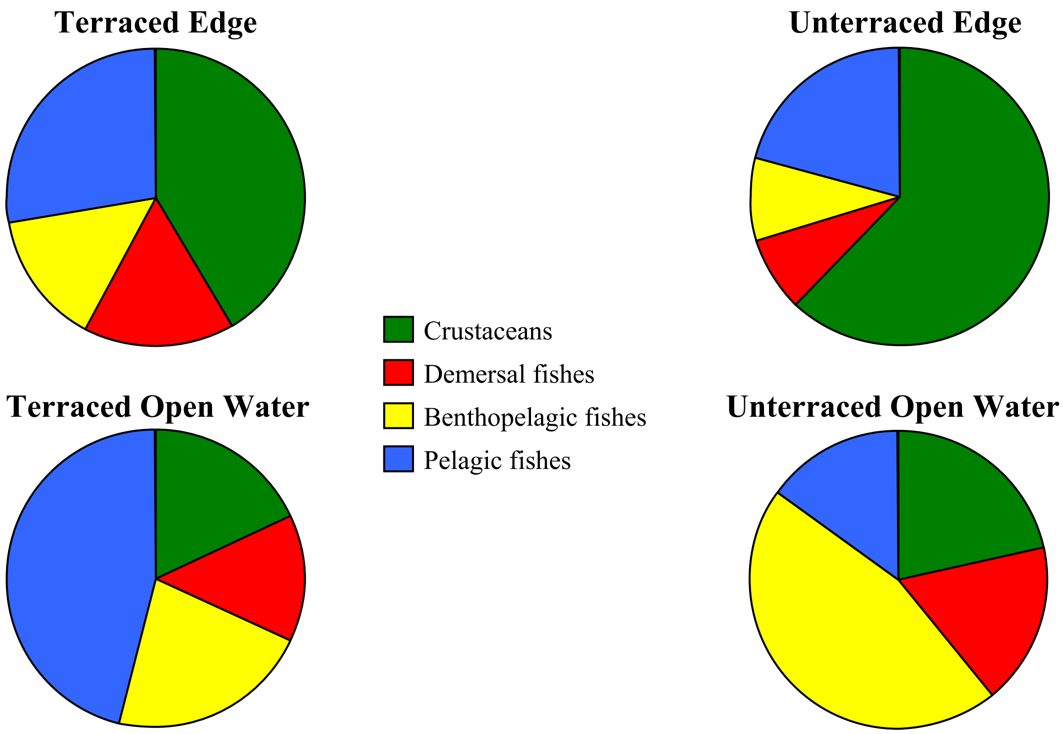


Figure 3.2. Total catch of crustaceans (*Palaemonetes pugio, Callinectes sapidus, Litopenaeus setiferus, Xanthidae spp., Farfantepenaeus aztecus*), demersal fishes (*Gobiosoma bosc, Microgobius gulosis, Micropogonias undulatus, Syngnathus scovelli, Bairdiella chrysoura, Lagodon rhomboides, Archosargus probatocephalus, Prionotus rubio, Lepisosteus oculatus*), benthopelagic fishes (*Poecilia latipinna, Gambusia affinis, Cyprinodon variegatus, Fundulus pulvereus, Lepomis spp.*), and pelagic fishes (*Anchoa mitchilli, Brevoortia patronus, Lucania parva, Menidia beryllina, Alosa chrysochloris*) by habitat type (Chi-sq: $P < 0.0001$, n = 3,544)

Table 3.2. Total catch of each species collected during the study period by pond type and habitat type. A total of 152 samples were collected, 38 at each of the 4 habitat types.

Common Name	**Scientific Name**	**Study Total**	**Terraced**			**Unterraced**		
			Total	**Edge**	**Open water**	**Total**	**Edge**	**Open water**
Crustaceans								
Daggerblade grass shrimp	*Palaemonetes pugio*	1171	436	380	56	735	628	107
Blue crab	*Callinectes sapidus*	150	67	49	18	83	71	12
White shrimp	*Litopenaeus setiferus*	93	27	16	11	66	55	11
Mud crab	*Xanthidae* spp.	90	28	28	0	62	61	1
Brown shrimp	*Farfantepenaeus aztecus*	7	4	4	0	3	2	1
Fishes								
Rainwater killifish	*Lucania parva*	465	330	183	147	135	104	31
Sailfin molly	*Poecilia latipinna*	392	131	68	63	261	28	233
Inland silverside	*Menidia beryllina*	387	177	119	58	210	153	57
Naked goby	*Gobiosoma bosc*	285	127	114	13	158	77	81
Western mosquitofish	*Gambusia affinis*	113	14	9	5	99	59	40
Clown goby	*Microgobius gulosus*	84	50	25	25	34	16	18
Sheepshead minnow	*Cyprinodon variegatus*	74	37	12	25	37	31	6
Sunfish	*Lepomis* spp.	73	73	67	6	0	0	0
Gulf pipefish	*Syngnathus scovelli*	70	62	46	16	8	7	1
Bay anchovy	*Anchoa mitchilli*	42	24	11	13	18	14	4
Silver perch	*Bairdiella chrysoura*	13	12	0	12	1	0	1
Bluegill	*Lepomis macrochirus*	11	11	8	3	0	0	0
Gulf menhaden	*Brevoortia patronus*	8	8	8	0	0	0	0
Atlantic croaker	*Micropogonias undulatus*	8	2	2	0	6	1	5
Skipjack herring	*Alosa chrysochloris*	2	0	0	0	2	2	0
Pinfish	*Lagodon rhomboides*	2	0	0	0	2	2	0
Sheepshead	*Archosargus probatocephalus*	1	0	0	0	1	1	0
Bayou killifish	*Fundulus pulverous*	1	1	0	1	0	0	0
Spotted gar	*Lepisosteus oculatus*	1	1	0	1	0	0	0
Blackwing searobin	*Prionotus rubio*	1	1	1	0	0	0	0
Total		3544	1623	1150	473	1921	1312	609

Nekton Density and Biomass

Nekton density and biomass were significantly greater at edge as compared to open water habitats, and specifically at terraced edge as compared to unterraced open water habitats (Table 3.3, Fig. 3.3, 3.4). There was no significant difference in nekton density or biomass between terraced and unterraced ponds, or between terraced and unterraced edge habitats. The effect of sampling date was not significant.

Nekton Species Richness and Diversity

Pearson's correlation coefficient indicated a strong correlation between Shannon diversity index (H') and Simpson's diversity index (1-D) (Prob > | r | = 0.91, $P < 0.0001$), therefore only species richness and H' will be reported. Species richness and H' were significantly greater at edge as compared to open water habitats, and specifically at terraced edge as compared to unterraced open water habitats (Table 3.3, Fig. 3.5, 3.6). Species richness and H' were significantly higher at terraced edge habitats as compared to unterraced open water habitats. There was no significant difference in species richness or H' between terraced and unterraced ponds, or between terraced edge and unterraced edge habitats. The effect of sampling date was not significant.

Submerged Aquatic Vegeation

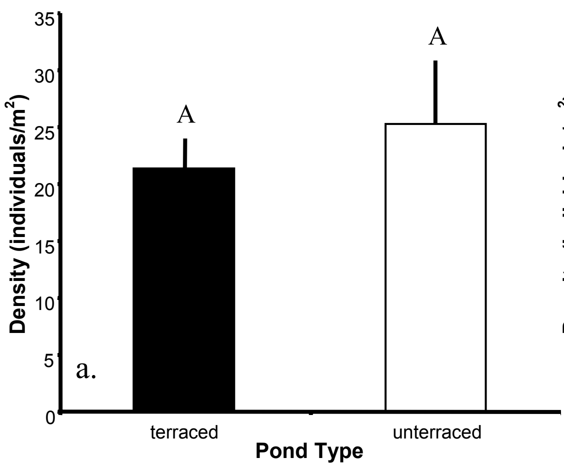
Nekton density was positively related to SAV biomass ($P < 0.0001$) (Fig. 3.7). This relationship can be described by the following regression equation:

$$\text{Nekton density} = 1.72 + 0.45(\text{SAV biomass}).$$

The explained variance (R^2) for the model was 18.2 %.

Table 3.3. Mean (SE) values of density, biomass, species richness, and diversity of nekton. Terraced and unterraced include both habitat types. Edge and open water include both pond types.

Variable	Terraced	Unterraced	Edge	Open water	Terraced edge	Unterraced edge	Unterraced open water
Density (individuals/m^2)	21.3 (3.5)	25.3 (6.2)	32.4 (4.7)	14.2 (5.0)	30.2 (6.0)	34.5 (7.4)	16.0 (9.7)
Biomass (g/m^2)	6.8 (1.4)	7.1 (1.9)	9.6 (1.7)	4.3 (1.6)	10.2 (2.5)	9.0 (2.2)	5.2 (3.1)
Species Richness (species/m^2)	3.1 (0.2)	2.3 (0.3)	3.5 (0.2)	1.9 (0.2)	3.6 (0.3)	3.3 (0.3)	1.4 (0.3)
Shannon (H')	0.93 (0.06)	0.78 (0.07)	0.92 (0.05)	0.77 (0.09)	0.98 (0.07)	0.86 (0.07)	0.60 (0.16)
Simpson (1-D)	0.56 (0.03)	0.47 (0.04)	0.55 (0.03)	0.46 (0.05)	0.57 (0.03)	0.53 (0.05)	0.34 (0.08)



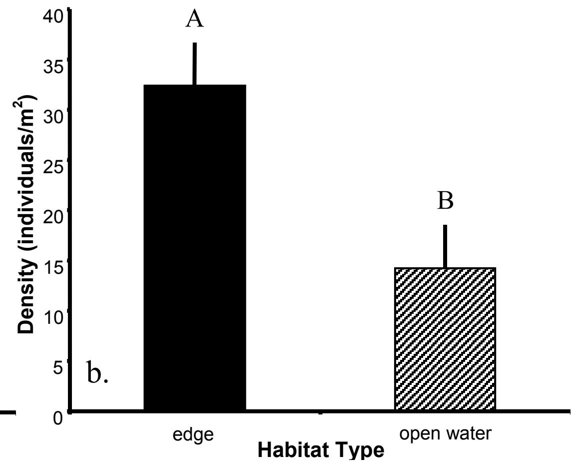


Figure 3.3a and 3.3b. Nekton density (individuals/m^2) by pond type (a) and habitat type (b). Results are not weighted for amount of edge and open water habitat within ponds. Different letters indicate significant difference ($P < 0.05$).

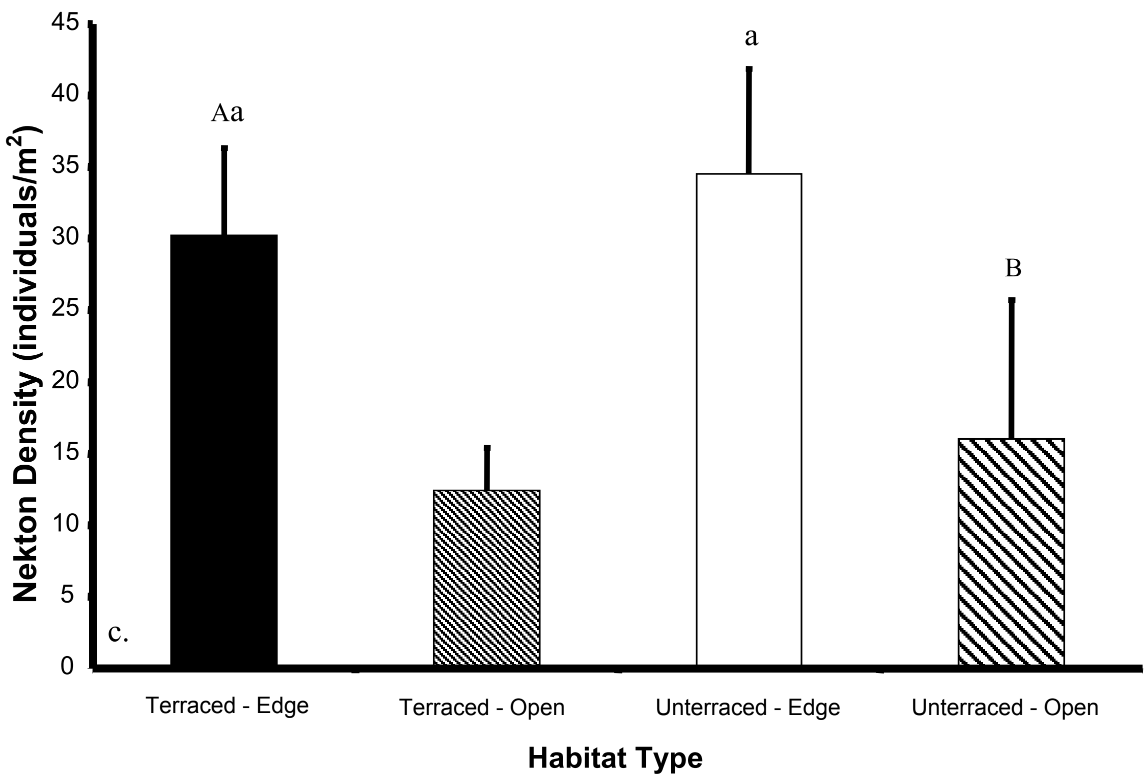


Figure 3.3c. Nekton density (individuals/m^2) by habitat type. Different letters indicate a significant difference ($P < 0.05$) in two *a priori* contrasts: terraced edge vs. unterraced open water (uppercase letters), terraced edge vs. unterraced edge (lowercase letters). Other comparisons were not of interest in this study and were thus not compared post-ANOVA.

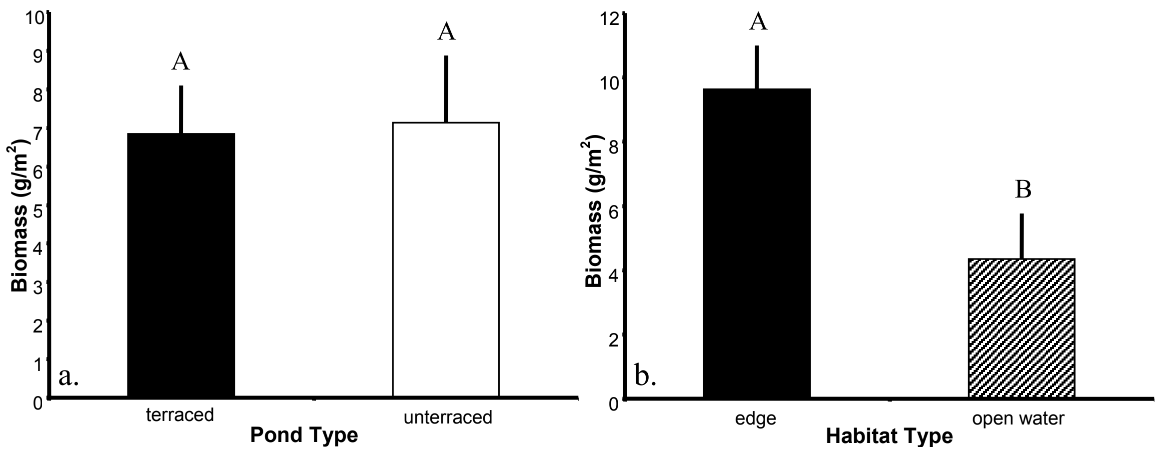


Figure 3.4a and 3.4b. Nekton biomass (g/m^2) by pond type (a) and habitat type (b). Results are not weighted for amount of edge and open water habitat within ponds. Different letters indicate significant difference (P < 0.05).

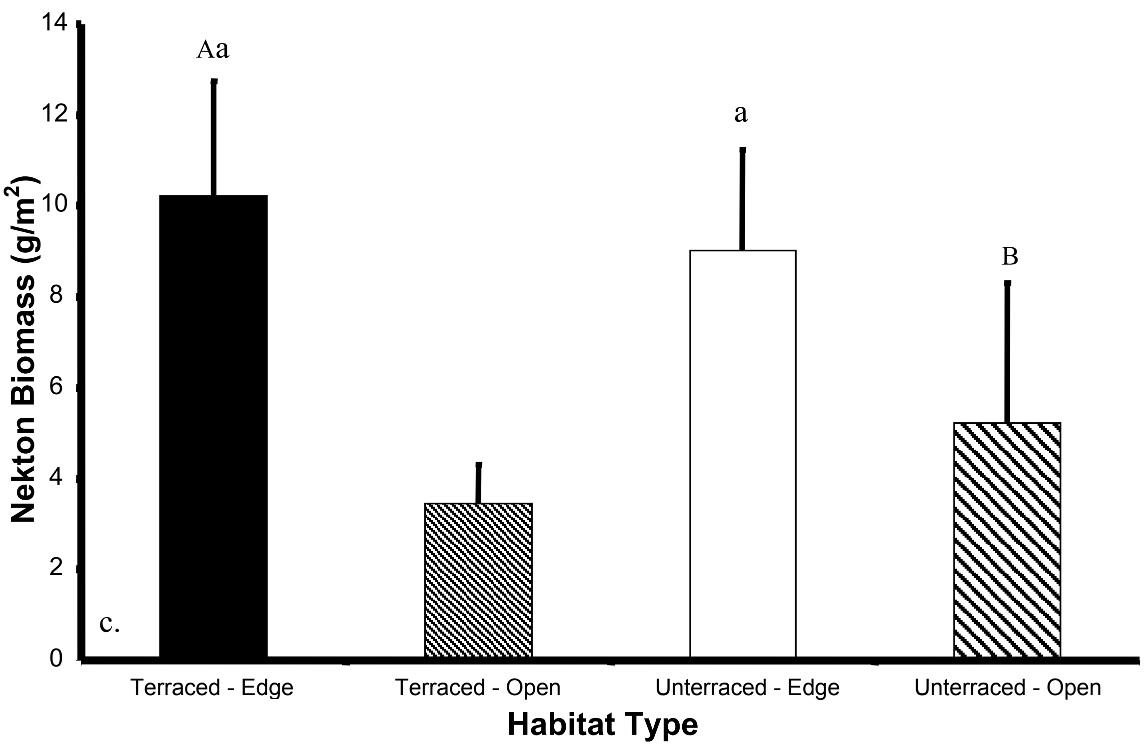


Figure 3.4c. Nekton biomass (g/m^2) by habitat type. Different letters indicate a significant difference (P < 0.05) in two *a priori* contrasts: terraced edge vs. unterraced open water (uppercase letters), terraced edge vs. unterraced edge (lowercase letters). Other comparisons were not of interest in this study and were thus not compared post-ANOVA.

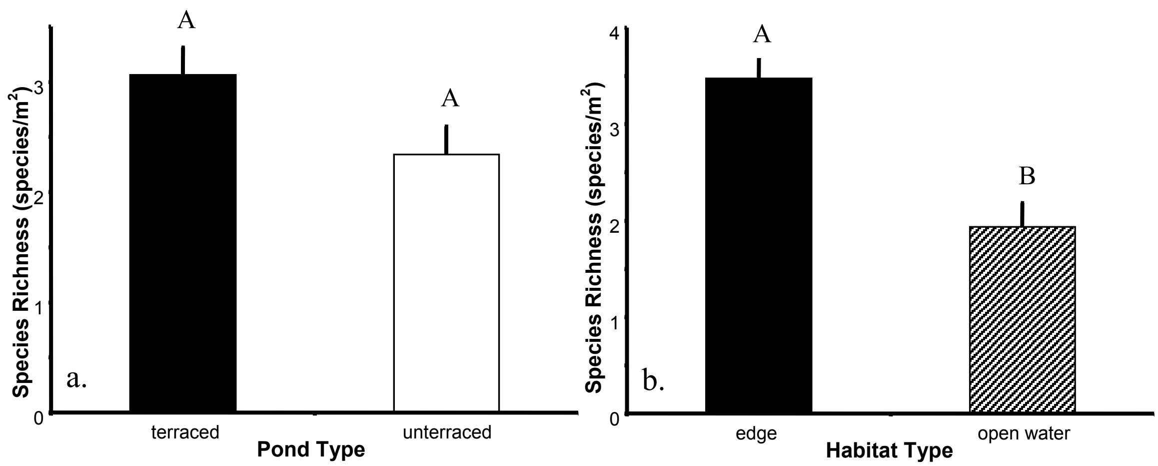


Figure 3.5a and 3.5b. Nekton species richness (species/m^2) by pond type (a) and habitat type (b). Results are not weighted for amount of edge and open water habitat within ponds. Different letters indicate significant difference ($P < 0.05$).

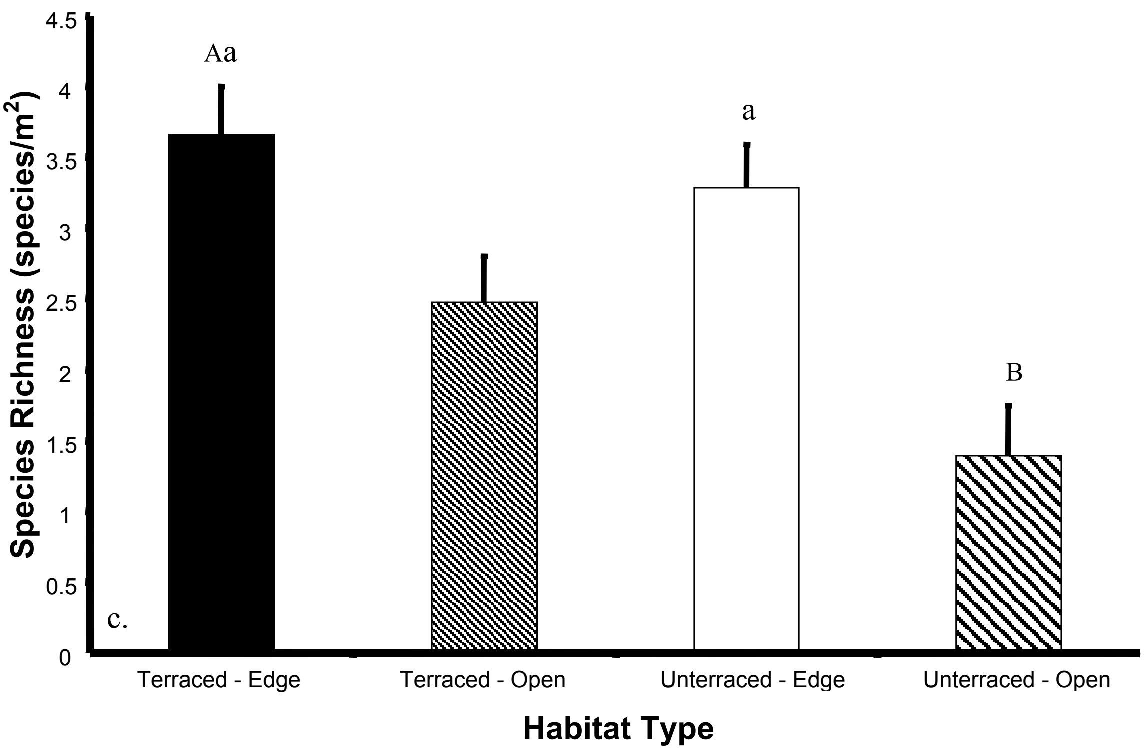


Figure 3.5c. Nekton species richness (species/m^2) by habitat type. Different letters indicate a significant difference ($P < 0.05$) in two *a priori* contrasts: terraced edge vs. unterraced open water (uppercase letters), terraced edge vs. unterraced edge (lowercase letters). Other comparisons were not of interest in this study and were thus not compared post-ANOVA.

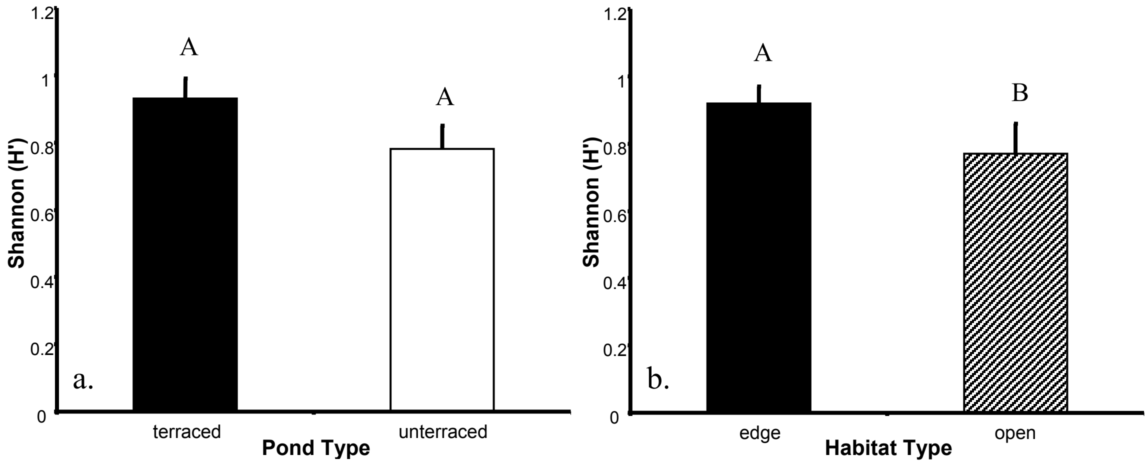


Figure 3.6a and 3.6b. Shannon diversity index (H') by pond type (a) and habitat type (b). Results are not weighted for amount of edge and open water habitat within ponds. Different letters indicate significant difference ($P < 0.05$).

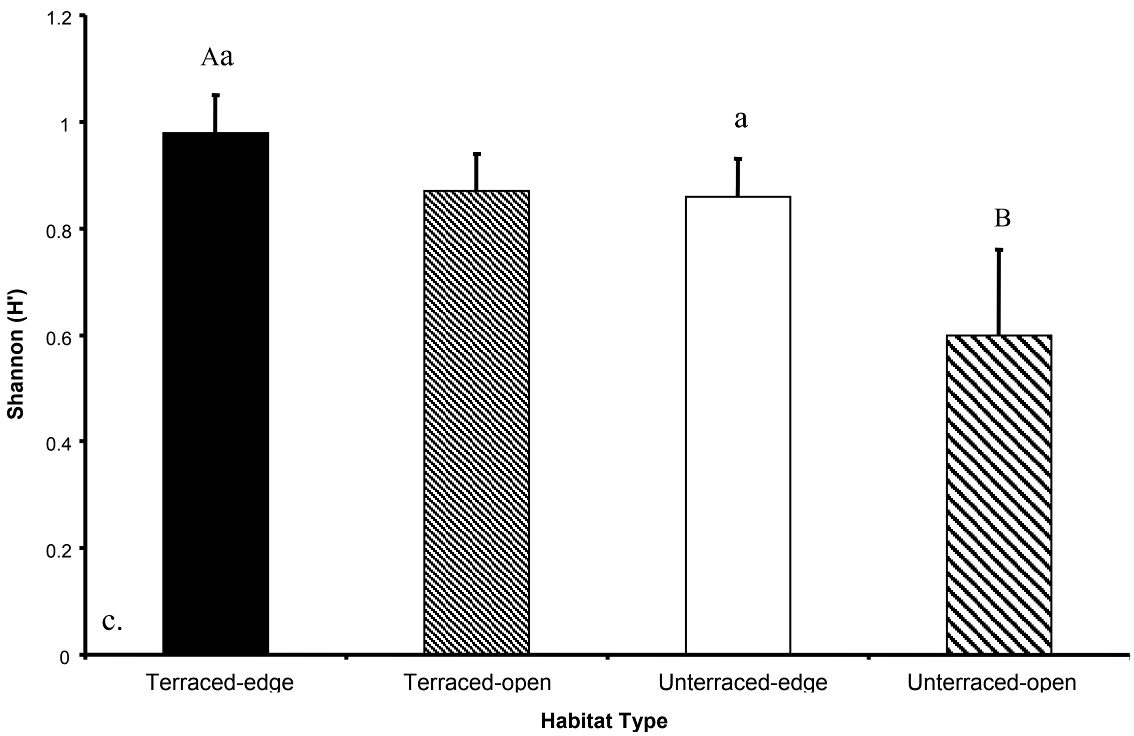


Figure 3.6c. Shannon diversity index (H') by habitat type. Different letters indicate a significant difference ($P < 0.05$) in two *a priori* contrasts: terraced edge vs. unterraced open water (uppercase letters), terraced edge vs. unterraced edge (lowercase letters). Other comparisons were not of interest in this study and were thus not compared post-ANOVA.

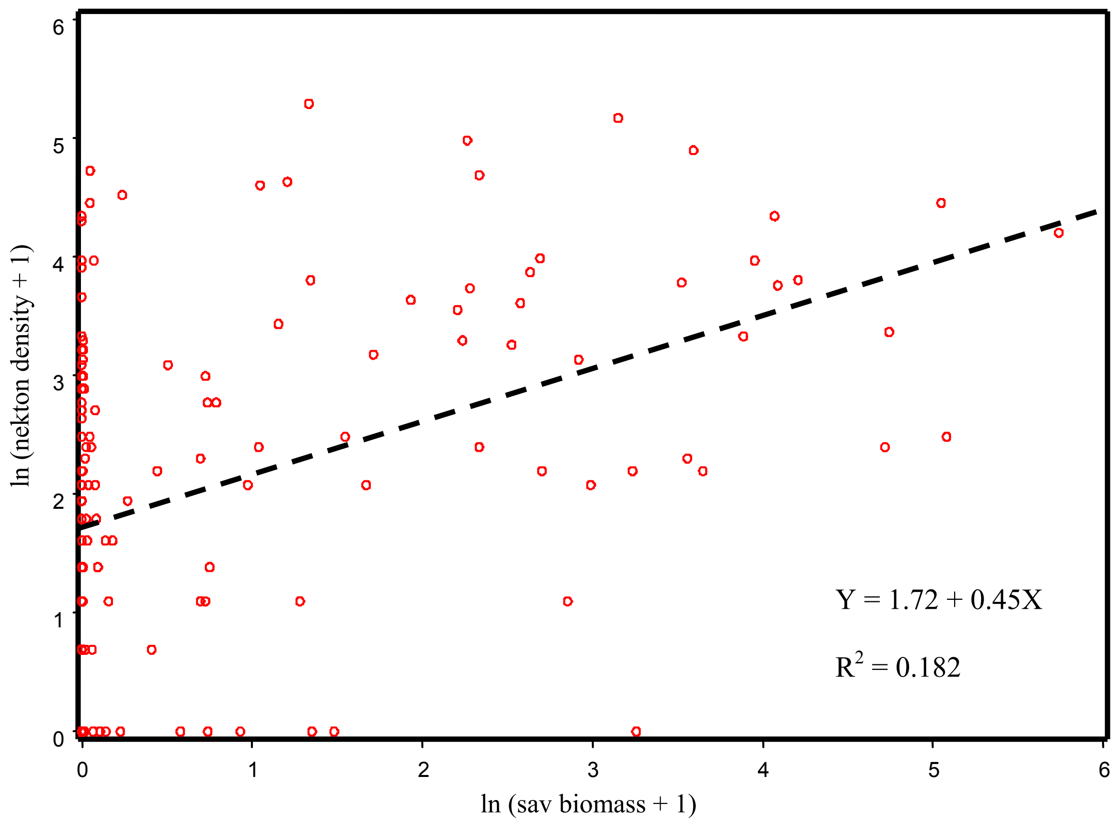


Figure 3.7. Relationship between SAV biomass (ln(x+1) transformed) and nekton density (ln(x+1) transformed). Regression line indicated by dashed line.

Fish Condition Indices

Simple Linear Regression was used to determine the length-weight relationships of the seven most frequently collected fish species (Table 3.4, Appendix A).

Fulton's condition factor (K) and Relative condition factor (K_n) were calculated for the seven most frequently collected fish species (Table 3.5). For *Poecilia latipinna,* K and K_n were significantly higher for fish collected in terraced ponds as compared to those collected in unterraced ponds. For *Menidia beryllina* and *Cyprinodon variegatus,* K and K_n were significantly lower for fish collected in terraced ponds as compared to fish collected in unterraced ponds. Both K and K_n were lower for

Microgobius gulosus collected in terraced ponds as compared to those collected in unterraced ponds, however only K_n was significant. There was no significant difference in K or K_n between terraced and unterraced ponds for *Lucania parva*, *Gobiosoma bosc*, or *Gambusia affinis*.

Table 3.4. Length-weight relationships of seven species of fish expressed as regression equations and power functions

Species	**n**	$\log_{10}W = a' + b \times \log_{10}L$	$W = aL^b$
Lucania parva	465	$\log_{10}W = -1.90 + 2.93 \times \log_{10}L$	$W = 0.0127L^{2.9253}$
Poecilia latipinna	392	$\log_{10}W = -1.76 + 2.96 \times \log_{10}L$	$W = 0.0173L^{2.9552}$
Menidia beryllina	387	$\log_{10}W = -2.06 + 2.80 \times \log_{10}L$	$W = 0.0086L^{2.8014}$
Gobiosoma bosc	285	$\log_{10}W = -1.82 + 2.81 \times \log_{10}L$	$W = 0.0153L^{2.8072}$
Gambusia affinis	113	$\log_{10}W = -1.90 + 3.08 \times \log_{10}L$	$W = 0.0126L^{3.0809}$
Microgobius gulosus	84	$\log_{10}W = -1.92 + 2.72 \times \log_{10}L$	$W = 0.0121L^{2.7239}$
Cyprinodon variegatus	74	$\log_{10}W = -1.78 + 3.28 \times \log_{10}L$	$W = 0.0166L^{3.279}$

Table 3.5. Mean K and K_n by pond type for the seven most frequently collected fish species. Significant t-test results are in bold type.

Species	**Fulton's Condition Factor (*K*)**			**Relative Condition Factor (*Kn*)**		
	Terraced	**Unterraced**	**Pr > |t|**	**Terraced**	**Unterraced**	**Pr > |t|**
Lucania parva	1.20 ± 0.01	1.17 ± 0.01	0.11	1.01 ± 0.01	1.00 ± 0.01	0.36
Poecilia latipinna	1.75 ± 0.02	1.65 ± 0.01	**< 0.0001**	1.05 ± 0.01	1.00 ± 0.01	**0.0004**
Menidia beryllina	0.70 ± 0.01	0.75 ± 0.01	**< 0.0001**	1.00 ± 0.01	1.04 ± 0.01	**0.008**
Gobiosoma bosc	1.31 ± 0.02	1.31 ± 0.02	0.78	1.00 ± 0.01	1.02 ± 0.01	0.55
Gambusia affinis	1.38 ± 0.05	1.36 ± 0.03	0.81	1.02 ± 0.04	1.02 ± 0.02	0.88
Microgobius gulosus	0.89 ± 0.02	0.93 ± 0.03	0.21	0.98 ± 0.02	1.05 ± 0.02	**0.01**
Cyprinodon variegatus	1.84 ± 0.08	2.32 ± 0.05	**< 0.0001**	0.94 ± 0.03	1.08 ± 0.03	**0.0004**

DISCUSSION

The results of this study indicate that terracing improves the habitat value of degrading marsh ponds for estuarine nekton. Nekton density, biomass, species richness, and diversity are all increased through the conversion of shallow open water habitat to marsh edge. While terraced pond habitats were superior to pre-restoration conditions in terms of nekton habitat value, they lacked functional equivalency with comparable unterraced ponds in several areas: 1) nekton community composition differed between terraced and unterraced edge habitats, and 2) several fish species were found to be in poorer condition in terraced ponds as compared to unterraced ponds. A lack of functional equivalency between terraced and unterraced habitats may be partially attributable to the relatively young age of the terraces studied, which may not have allowed for the long-term development of some environmental variables, such as soil organic matter, which is linked to benthic infaunal community diversity and density, needed for a marsh to reach full functional equivalency (Moy and Levin 1991, Sacco et al. 1994, Morgan and Short 2002, Craft et al. 2003).

Value of Terraced Edge Habitat

The value of marsh edge as habitat for nekton is well documented (Boesch and Turner 1984, Baltz et al. 1993, Minello et al. 1994, Peterson and Turner 1994). Resident and transient nekton species use the marsh edge as habitat and highest densities of nekton can be found less than 3 m from the marsh edge (Peterson and Turner 1994). Decapod crustaceans such as white shrimp, brown shrimp, and daggerblade grass shrimp have shown a strong affinity for marsh edge (Minello et al. 1994, Peterson and Turner 1994). Marsh edge habitat also serves as a seasonal nursery for many estuarine-dependent species. The early life history stages of many economically important fishes are found in significant densities among the shallow habitats associated with marsh edge (Boesch and Turner 1984, Baltz et al. 1993).

Because marsh terracing converts areas of open water to marsh edge, comparisons of these two habitat types (open water, terraced marsh edge) have been used to evaluate the success of terracing projects (i.e. Rozas and Minello 2001, Bush Thom et al. 2004). While these past studies each examined one specific terrace field, this study, encompassing three terrace fields utilizing different spatial arrangements, was able to corroborate their primary findings that terraced edge habitats support greater nekton density, biomass, species richness, and diversity as compared to unterraced open water habitats. Marsh terracing converts areas of lower quality open water habitat to higher quality edge habitat and increases the proportion of edge throughout the pond. These findings strongly suggest that to maximize the impact of a marsh terracing project on nekton habitat, the amount of marsh edge created should be maximized.

In previous studies of nekton in terraced marsh ponds, actual nekton densities have varied considerably. Bush Thom et al. (2004) reported mean nekton densities of 3.3 individuals/m^2 along the marsh edge and 1.3 individuals/m^2 in unmanaged open water. Rozas and Minello (2001) found much higher densities of up to 110.3 individuals/m^2 (107.2 crustaceans, 3.1 fish) along the terrace edge and 14.7 individuals/m^2 (10.9 crustaceans, 3.8 fish) in reference pond open water. While the mean density value for open water was similar in our study to that of Rozas and Minello (2001) (16.0 individuals/m^2), the mean density values for terraced edge (30.2 individuals/m^2) fell between the densities reported in the two previous site specific studies. A review utilizing data from 22 studies in estuarine areas of the northern Gulf of Mexico reported mean nekton densities ranging from 36.0 to 83.5 crustaceans/m^2 and 7.7 to 14.9 fish/m^2 at marsh edge habitats, depending on the type of emergent vegetation present, and 5.5 crustaceans/m^2 and 10.5 fish/m^2 at shallow unvegetated bottom habitats. (Minello 1999). In another study from a Louisiana estuary, nekton densities ranged from 12.4 to 32.4 individuals/m^2 at the marsh edge (Baltz et al. 1993). As indicated by the above literature, nekton densities can vary greatly, which may be attributable to specific properties of the sites, sampling gear, time of day, or time of year. For the terracing studies, although the actual densities of nekton varied by study, the general trend in nekton distribution among habitats within terraced and reference ponds remained consistent, with greater densities at both terraced and unmanaged edge as compared to open water habitats within terraced and unterraced ponds.

<u>Nekton Community Composition</u>

While terraced marsh edge supported densities of nekton similar to unterraced marsh edge, nekton community composition differed significantly. Crustaceans made up the greatest proportion of total catch at edge habitats with a greater proportion at unterraced edge (62%) as compared to terraced edge (41%). Similarly, Rozas and Minello (2001) found significantly lower densities of daggerblade grass shrimp, brown shrimp, and blue crab at terraced marsh as compared to natural marsh and Bush Thom et al. (2004) found that terraced edge supported a high percentage of pelagic fishes while unterraced edge supported high percentages of benthic fishes and crustaceans. Differences in abundance of crustacean species at terraced and unterraced edge habitats could be caused by differences in the availability of suitable benthic prey. Decapod crustaceans are known to feed on benthic infauna and epifauna (Kneib 1985, Hunter and Feller 1987). Minello and Zimmerman (1992) found a positive correlation between density of benthic infauna and density of decapod crustaceans, and found that diversity of infauna was significantly greater in natural as compared to created salt marshes. In this study, fishes were more abundant at terraced edge with all three functional groups (demersal, benthopelagic, and pelagic) accounting for a larger proportion of total catch as compared to unterraced edge habitats. Oftentimes, generalist species with less specific habitat and prey requirements are more abundant in restored or created marshes (Minello and Webb 1997, Williams and Zedler 1999, Bush Thom et al. 2004) although this does not seem to be the case with this study because demersal fishes were more abundant at terraced edge as compared to unterraced edge.

Fish Condition

Condition of fishes can be used to draw conclusions as to the quality of the habitats from which those fishes were collected. For example, condition indices have been used to compare the condition of juvenile white seabream *Diplodus sargus* from rocky coastal habitats and sandy coastal habitats (Lloret and Planes 2003), to compare the condition of common minnows *Esomus danricus* in flowing stream channels and stagnant pools (Mustafa 1978), and to examine the response of Atlantic croaker *Micropogonias undulatus* to estuarine pollution (Burke et al. 1993). Despite the use of condition indices in the above mentioned studies, we are aware of no studies that have used condition indices to compare fish condition in restored or created marshes to reference marshes. This study used two condition indices (K and K_n) as indicators of habitat quality in order to determine if there was a detectable difference in the condition of numerically dominant fishes between terraced and unterraced ponds.

In comparing terraced and unterraced ponds, condition (K and K_n) of fishes varied by species. Of the 7 species examined, condition was higher in terraced ponds for 1 species, lower in terraced ponds for 3 species, and similar between pond types for the 3 remaining species. It is important to note that differences in condition between pond types do not necessarily indicate that the fish from one pond type are in "good" condition while those from the other pond type are in "poor" condition, rather that a difference in condition exists between the treatments.

There are several possible explanations for the higher condition of the terraced pond *Poecilia latipinna*. While it is possible that *P. latipinna* were simply faring better in the terraced ponds, the higher condition is more likely due to the fact that 84 % of the *P. latipinna* were collected in one sample from an unterraced pond (219 of 261 fish) during a period of low water when edge habitats were dry and nekton seemed concentrated in the shallow open water. These fish were likely in poor condition due to stress caused by the environmental conditions at the time. The higher condition of *P. latipinna* in terraced ponds may also be explained by the higher SAV biomass present in the terraced ponds which provided a more suitable habitat for *P. latipinna*.

The poorer condition of *Cyprinodon variegatus*, *Microgobius gulosus*, and *Menidia beryllina* from terraced ponds can likely be explained by differences in habitat quality between terraced and unterraced ponds. Terraced edge habitats had significantly less organic matter in the soil than unterraced edge habitats. Differences in soil organic matter between natural and constructed marshes have been linked to differences in benthic infaunal communities (Moy and Levin 1991, Sacco et al. 1994, Levin et al. 1996). Salt marsh infauna, located at the base of the estuarine food web, are an important link between primary production in the marsh and the adjacent waters (Sacco et al. 1994). In gut content analysis of mummichogs *Fundulus heteroclitus* collected in North Carolina, major differences were observed in the diets of fish collected in natural and constructed marshes (Moy and Levin, 1991). There were differences in the composition of meiofauna and macrofauna consumed by *F. heteroclitus* as well as plant detritus. Fish from the natural marshes consumed more plant detritus than fish from the constructed marsh.

There was no clear explanation for the differences in condition based on functional groupings. The three species with poorer condition in the terraced ponds included one demersal, one benthopelagic, and one pelagic species. Likewise, the three species with no difference in condition between ponds included one demersal, one benthopelagic, and one pelagic species. It is promising though that three species showed no difference in condition between ponds, suggesting that terraced ponds provide equivalent habitat, at least for some species, as unterraced ponds.

<u>Functional Equivalency</u>

The terraced ponds in the study were not functionally equivalent to the unterraced ponds in several categories. Organic matter content in terraced edge soil was less than that of unterraced edge soil. Although nekton density, biomass, species richness, and diversity were similar between terraced and unterraced edge habitats, the composition of the nekton communities differed. Also, condition indices for three species of fish were found to be lower in terraced ponds as compared to unterraced ponds, suggesting that habitat functions may have not yet developed.

The three terraced ponds investigated were all 3-4 years of age at the time of the study. Some of the ecological functions of constructed marshes can take much longer than this to develop to levels similar to reference areas. In a study of constructed marshes ranging from 1-28 years of age, Craft (2003) found that ecological attributes linked to heterotrophic processes (invertebrate density, C mineralization) were strongly tied to levels of organic C in the top 10 cm of soil. Most ecological processes reached equivalence to natural marshes after 5-15 years when C and N reached critical levels in the soil. Soil organic matter at terraced edge habitats was significantly less than at unterraced edge habitats. Due to the importance of soil organic matter in the ecology of marshes and the young age of the terraces, it is likely that other habitat functions have not developed to the levels of the unterraced ponds. It is possible that habitat functions will develop as these terrace fields age, providing habitats of similar quality to those found in unterraced ponds.

Terrace Design and Future Research

This and previous studies all strongly suggest that maximizing the amount of marsh edge within ponds will maximize the habitat value for nekton. Future terracing projects should incorporate this concept into their design, perhaps by reducing the amount of space between terraces. This may also be helpful in achieving one of the other goals of terracing: the reduction of turbidity. There was little indication from this study that terracing had any effect on water quality. Most of the water quality variables measured were similar between ponds. Turbidity in the terraced ponds was slightly lower, although the difference was not significant. Likewise, Rozas and Minello (2001) and Bush Thom et al. (2004) found no significant effect of terraces on turbidity. Lower salinity and conductivity in the terraced ponds was likely due to the physical location of the ponds rather than a treatment effect of the terraces.

Future studies of terraces are warranted because there are still important considerations that have not been addressed. Future studies should include sampling gears that are suited to the capture of adult fish, as enclosure samplers tend to exclude larger individuals (see Appendix B). Sampling of benthic invertebrates as well as gut content analysis for fishes could help to explain differences in condition. Finally, future studies should include older terrace fields to determine if the functional differences observed in past studies will become more similar over time. Until recently this has not been possible because terracing was not introduced as a restoration technique in Louisiana until 1990. Soon however, terrace fields 10-15 years of age will become increasingly common, allowing researchers to study their development over time.

LITERATURE CITED

Able, K. W., D. M. Nemerson, and T. M. Grotheus. 2003. Evaluating salt marsh restoration in Delaware Bay: analysis of fish response at former salt hay farms. Estuaries 27:58-69

Anderson, R. O. and R.M. Neumann. 1996. Length, weight, and associated structural indices. pp. 447-476 In: Murphy, B.R. and Willis D.W. editors. Fisheries techniques. 2nd ed. Bethesda, Maryland: American Fisheries Society;

Baltz, D. M., C. Rakocinski, and J. W. Fleeger. 1993. Microhabitat use by marsh-edge fishes in a Louisiana estuary. Environmental Biology of Fishes 36:109-126.

Barras, J., S. Beville, D. Britsh, S. Hartley, S. Hawes, J. Johnston, P. Kemp, Q. Kinler, A. Martucci, J. Porthouse, D. Reed, K. Roy, S. Sapkota, and J. Suhayda. 2003. Historical and projected coastal Louisiana land changes: 1978-2050. USGS Open File Report 03-334, 39 p. (Revised Jan. 2004).

Boesch, D. F., M. N. Josselyn, J. M. Ashish, J. T. Morris, W. K. Nuttle, C. A. Simenstad, and D. J. P. Swift. 1994. Scientific Assessment of Coastal Wetland Loss, Restoration and Management in Louisiana. Journal of Coastal Research. Special Issue no. 20. 103 pp.

Boesch, D. F., and R. E. Turner. 1984. Dependence of fishery species on salt marshes: the role of food and refuge. Estuaries 7:460-468

Broome, S. W., E. D. Seneca, and W. W. Woodhouse Jr. 1986. Long-term growth and development of transplants of the salt marsh grass *Spartina alterniflora*. Estuaries 9:63-74.

Burke, J. S., D. S. Peters, and P. J. Hanson. 1993. Morphological indices and otolith microstructure of Atlantic croaker, *Micropogonias undulatus*, as indicators of habitat quality along an estuarine pollution gradient. Environmental Biology of Fishes 36:25-33.

Bush Thom, C. S., M. K. La Peyre, and J. A. Nyman. 2004. Evaluation of nekton use and habitat characteristics of restored Louisiana marsh. Ecological Engineering 23:63-75.

Craft, C., J. Reader, J. N. Sacco, and S. W. Broome. 1999. Twenty-five years of ecosystem development of constructed *Spartina alterniflora* (Loisel) marshes. Ecological Applications 9:1405-1419.

Craft, C., P. Megonigal, S. Broome, J. Stevenson, R. Freese, J. Cornell, L. Zheng, and J. Sacco. 2003. The pace of ecosystem development of constructed *Spartina alterniflora* marshes. Ecological Applications 13:1417-1432.

Delaney, T. P., J. W. Webb, and T. J. Minello. 2000. Comparison of physical characteristics between created and natural estuarine marshes in Galveston Bay, Texas. Wetlands Ecology and Management 8:343-352.

Delaune, R. D., J. A. Nyman, W. H. Patrick Jr. 1994. Peat collapse, ponding, and wetland loss in a rapidly subsiding coastal marsh. Journal of Coastal Research 10:1021-1030.

Flynn, K. M., I. A. Mendelssohn, and B. J. Wilsey. 1999. The effect of water level management on the soils and vegetation of two coastal Louisiana marshes. Wetlands Ecology and Management 7:193-218.

Gilliers, C., R. Amara, J. P. Bergeron, and O. Le Pape. 2004. Comparison of growth and condition indices of juvenile flatfish in different coastal nursery grounds. Environmental Biology of Fishes 71:189-198.

Havens, K. J., L. M. Varnell, and B. D. Watts. 2002. Maturation of a constructed tidal marsh relative to two natural reference tidal marshes over 12 years. Ecological Engineering 18:305-315.

Hunter, J. and R. J. Feller. 1987. Immunological dietary analysis of two penaeid shrimp species from a South Carolina tidal creek. Journal of Experimental Marine Biology and Ecology 107:61-70.

Jivoff, P. R. and K. W. Able. 2003. Evaluating salt marsh restoration in Delaware Bay: the response of blue crabs, *Callinectes sapidus*, at former salt hay farms. Estuaries 26:709-719.

Jordan, F., S. Coyne, and J. C. Trexler. 1997. Sampling fishes in vegetated habitats: effects of habitat structure on sampling characteristics of the 1-m^2 throw trap. Transactions of the American Fisheries Society 126:1012-1020.

Kneib, R. T. 1985. Predation and disturbance by grass shrimp, *Palaemonetes pugio* Holthius, in soft-substratum benthic invertebrate assemblages. Journal of Experimental Marine Biology and Ecology 93:91-102.

Kushlan, J. A. 1981. Sampling characteristics of enclosure fish traps. Transactions of the American Fisheries Society 110:557-562.

Kusler, J. A. and M. E. Kentula. 1990. Wetland Creation and Restoration: The Status of the Science. Island Press, Washington, D.C

Levin, L. A., D. Talley, and G. Thayer. 1996. Succession of macrobenthos in a created salt marsh. Marine Ecology Progress Series 141:67-82.

Lloret, J. and S. Planes. 2003. Condition, feeding and reproductive potential of white seabream *Diplodus sargus* as indicators of habitat quality and the effect of reserve protection in the northwestern Mediterranean. Marine Ecology Progress Series 248:197-208.

Louisiana Coastal Wetlands Conservation and Restoration Task Force. 2003. The 2003 Evaluation Report to the U.S. Congress on the Effectiveness of Coastal Wetland Planning, Protection and Restoration Act Projects. Baton Rouge, LA: Louisiana Department of Natural Resources. 27 pp.

Magurran, A. E. 1988. Ecological Diversity and Its Measurement. Princeton University Press, Princeton, New Jersey 179 pp.

Minello, T. J. 1999. Nekton densities in shallow estuarine habitats of Texas and Louisiana and the identification of essential fish habitat. American Fisheries Society Symposium 22:43-75.

Minello, T. J. 2000. Temporal development of salt marsh value for nekton and epifauna: utilization of dredged material marshes in Galveston Bay, Texas, USA. Wetlands Ecology and Management 8:327-341.

Minello, T. J., R. J. Zimmerman, and R. Medina. 1994. The importance of edge for natant macrofauna in a created salt marsh. Wetlands 14:184-198.

Minello, T. J. and J. W. Webb, Jr. 1997. Use of natural and created Spartina alterniflora salt marshes by fishery species and other aquatic fauna in Galveston Bay, Texas, USA. Marine Ecology Progress Series 151:165-179.

Minello, T. J., and R. J. Zimmerman. 1992. Utilization of natural and transplanted Texas salt marshes by fish and decapod crustaceans. Marine Ecology Progress Series 90:273-285.

Mitsch, W. J., and J. G. Gosselink. 2000. Wetlands. Van Norstrand Reinhold Company Inc., New York, New York, USA. 920 pp.

Morgan, P. A. and F. T. Short. 2002. Using functional trajectories to track constructed salt marsh development in the Great Bay estuary, Maine/New Hampshire, USA. Restoration Ecology 10:461-473.

Moy, L. D. and L. A. Levin. 1991. Are *Spartina* marshes a replaceable resource? A functional approach to evaluation of marsh creation efforts. Estuaries 14:1-6.

Mustafa, S. 1978. Length-weight relationship and condition factor of common minnow, *Esomus danricus* (Ham.) from different freshwater environments. Zoological Journal of the Linnean Society 62:349-354.

Nyman, J. A. Assistant Professor. School of Renewable Natural Resources, Louisiana State University Agricultural Center, Louisiana State University.

Penland, S., H. H. Roberts, S. J. Williams, A. H. Sallenger Jr., D. R. Cahoon, D. W. Davis, and C. G. Groat. 1990. Coastal land loss in Louisiana. Transactions of the Gulf Coast Associations of Geological Societies 90:685-699.

Penland, S. and J. R. Suter. 1989. Geomorphology of the Mississippi River Chenier Plain. Marine Geology 90:231-258.

Peterson, G. W. and R. E. Turner. 1994. The value of salt marsh edge vs. interior as a habitat for fish and decapod crustaceans in a Louisiana tidal marsh. Estuaries 17:235-262.

Race, M.S. and M. S. Fonseca. 1996. Fixing compensatory mitigation: what will it take? Ecological Applications 6:94-101.

Rozas, L. P. and W. E. Odum. 1988. Occupation of submerged aquatic vegetation by fishes: testing the roles of food and refuge. Oecologica 77:101-106.

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

Rozas, L. P. and T. J. Minello. 2001. Marsh terracing as a wetland restoration tool for creating fishery habitat. Wetlands 21:327-341.

Rozas, L. P., P. Caldwell, and T. J. Minello. 2005. The fishery value of salt marsh restoration projects. Journal of Coastal Restoration Special Issue No. 40:37-50.

Sacco, J. N., E. D. Seneca, and T. R. Wentworth. 1994. Infaunal community development of artificially established salt marshes in North Carolina. Estuaries 17:489-500.

Shafer, D. and W. J. Streever. 2000. A comparison of 28 natural and dredged material salt marshes in Texas, with an emphasis on geomorphological variables. Wetlands Ecology and Management 8:353-366.

Simenstad, C. A. and R. M. Thom. 1996. Functional equivalency trajectories of the restored Gog-Le-Hi-Te estuarine wetland. Ecological Applications 6:38-56.

Steyer, G. D. 1993. Annual Monitoring Report Sabine Terracing Project. Louisiana Department of Natural Resources, Coastal Monitoring Division. 94 pp.

Turner, R. E. and W. Streever. 2002. Approaches to Coastal Wetland Restoration: Northern Gulf of Mexico. SPB Academic Publishing bv, The Hague, The Netherlands.

Underwood, S. G., G. D. Steyer, B. Good, and D. Chambers. 1991. Bay bottom terracing and vegetative planting: an innovative approach for habitat and water quality enhancement. p. 164-173 in F. J. Webb (ed.) Annual Conference on Wetlands Restoration and Creation. Hillsborough Community College, Tampa, FL, USA.

United Nations Educational, Scientific and Cultural Organization (UNESCO). 1971. Convention on wetlands of international importance especially as waterfowl habitat. Ramsar, Iran. 2 February 1971.

U.S. Army Corps of Engineers. 2004. Louisiana Coastal Area (LCA), Louisiana Ecosystem Restoration Study. U.S. Army Corps of Engineers, New Orleans District

U.S. Department of Commerce, National Oceanographic and Atmospheric Administration and National Marine Fisheries Service (USDOC). 2001. Fisheries of the United States. Washington D.C. Internet URL: http://www.st.nms.gov/st1/fus/fus01/index.html.

U.S. Fish and Wildlife Service. 2002. Sabine National Wildlife Refuge general brochure. 7 pp.

Wicker, K.M., D. Davis, and D. Roberts. 1983. Rockefeller State Wildlife Refuge and Game Preserve: evaluations of wetland management techniques. Coastal Management Division, Louisiana Department of Natural Resources, Baton Rouge, Louisiana

Williams, G. D. and J. B. Zedler. 1999. Fish assemblage composition in constructed and natural tidal marshes of San Diego Bay: relative influence of channel morphology and restoration history. Estuaries 22:702-716.

Windham, L., M. S. Laska, and J. Wollenberg. 2004. Evaluating urban wetland restorations: case studies for assessing connectivity and function. Urban Habitats 2:130-146.

Zimmerman, R. J. and T. J. Minello. 1984. Densities *of Penaeus aztecus*, *P. setiferus* and other natant macrofauna in a Texas salt marsh. Estuaries 7:421-433.

Zedler, J. B. 2000. Progress in wetland restoration ecology. Trends in Ecology and Evolution 15:402-407.

Zedler, J. B. and J. C. Callaway. 1999. Tracking wetland restoration: do mitigation sites follow desired trajectories? Restoration Ecology 7:69-73.

APPENDIX A

GRAPHS OF LENGTH-WEIGHT RELATIONSHIPS FOR SEVEN SPECIES OF ESTUARINE FISH

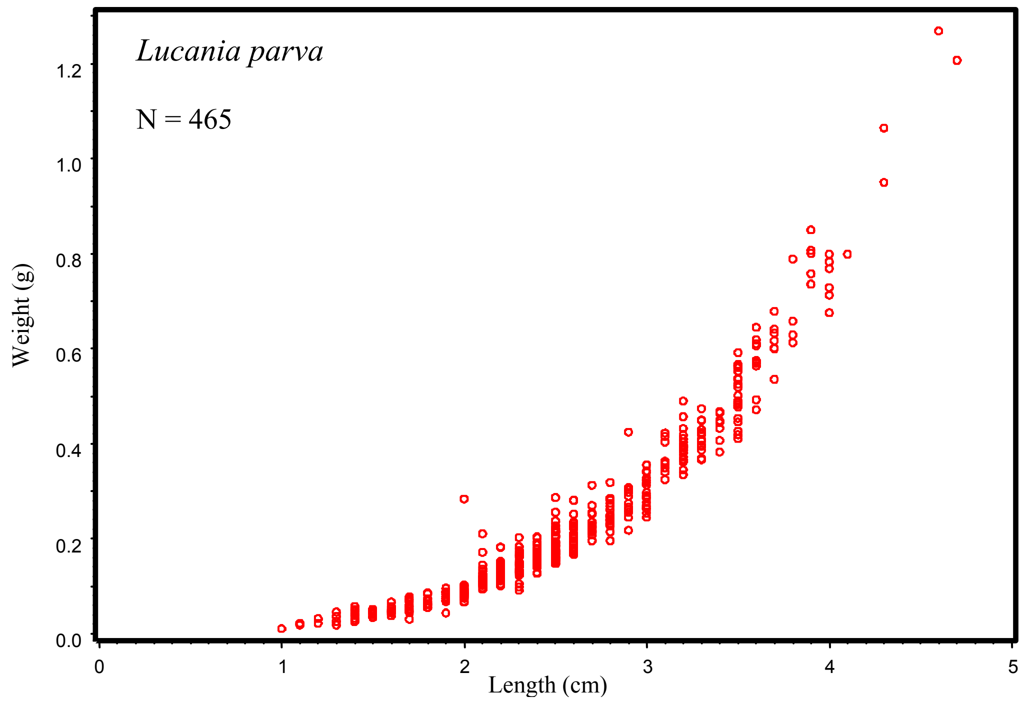


Figure A.1a. Plot of raw length-weight data for all rainwater killifish collected during study period.

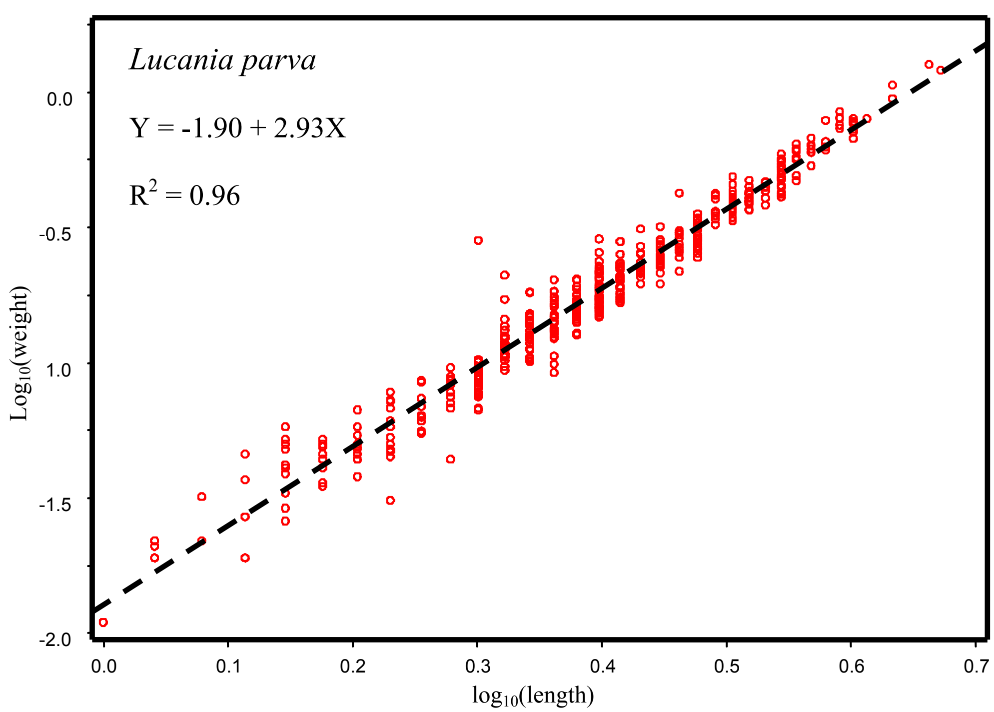


Figure A.1b. Simple Linear Regression on logarithmically transformed length-weight data for rainwater killifish.

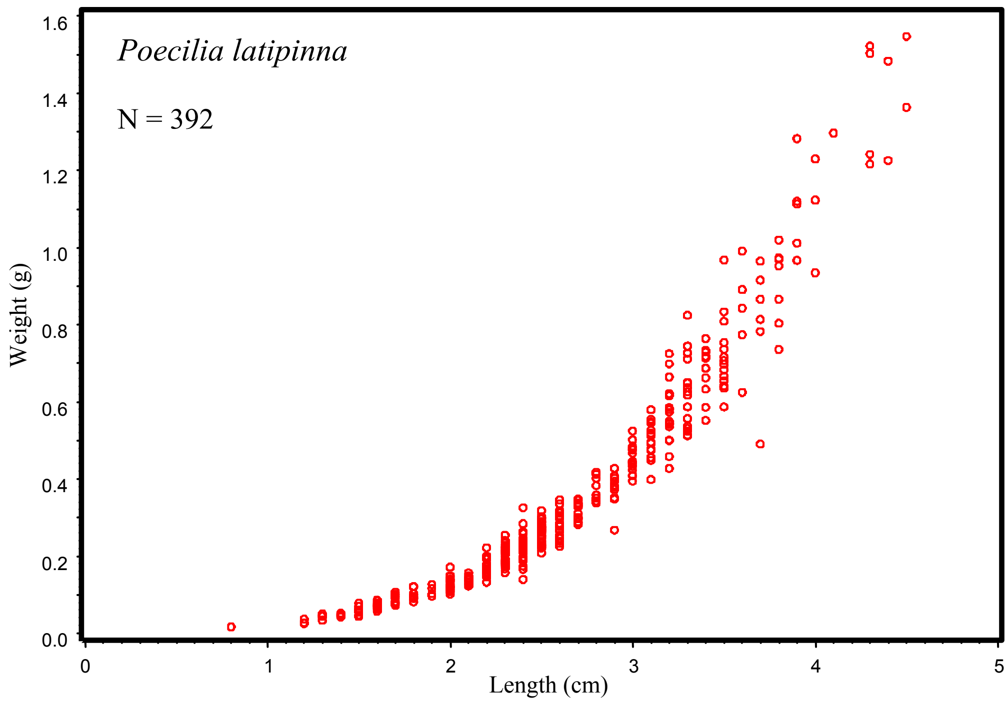


Figure A.2a. Plot of raw length-weight data for all sailfin mollies collected during study period.

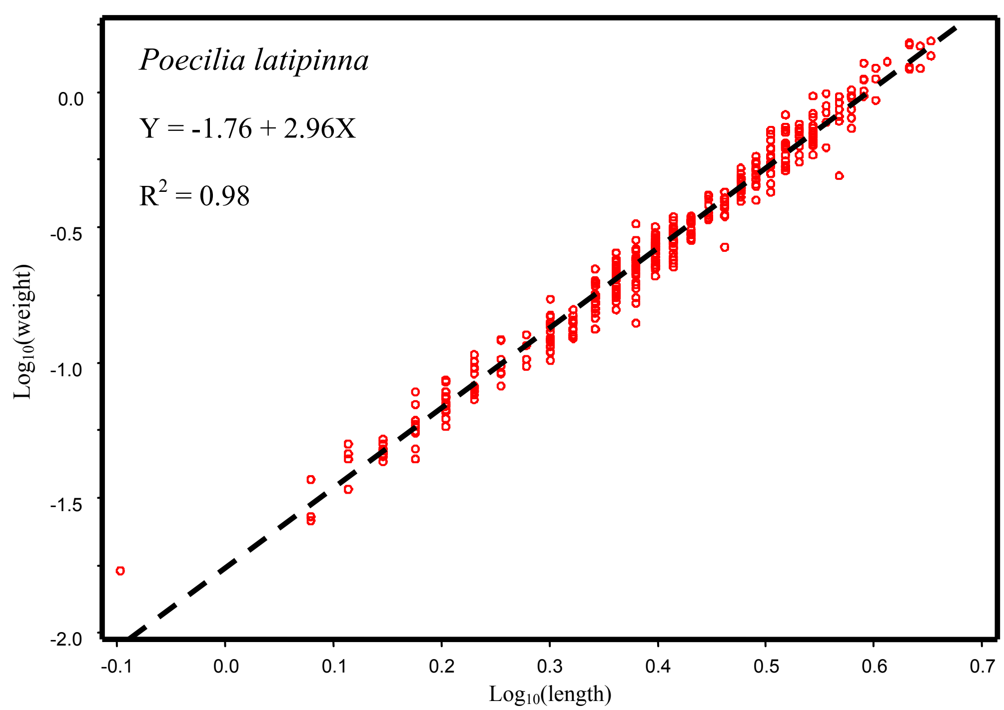


Figure A.2b Simple Linear Regression on logarithmically transformed length-weight data for sailfin mollies.

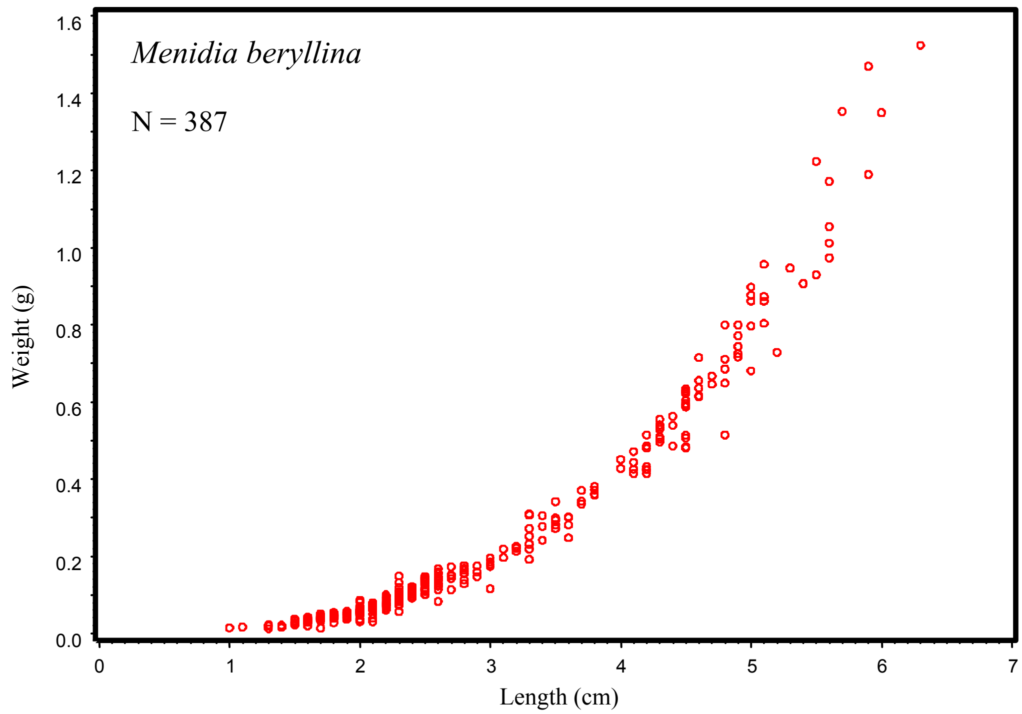


Figure A.3a. Plot of raw length-weight data for all inland silversides collected during study period.

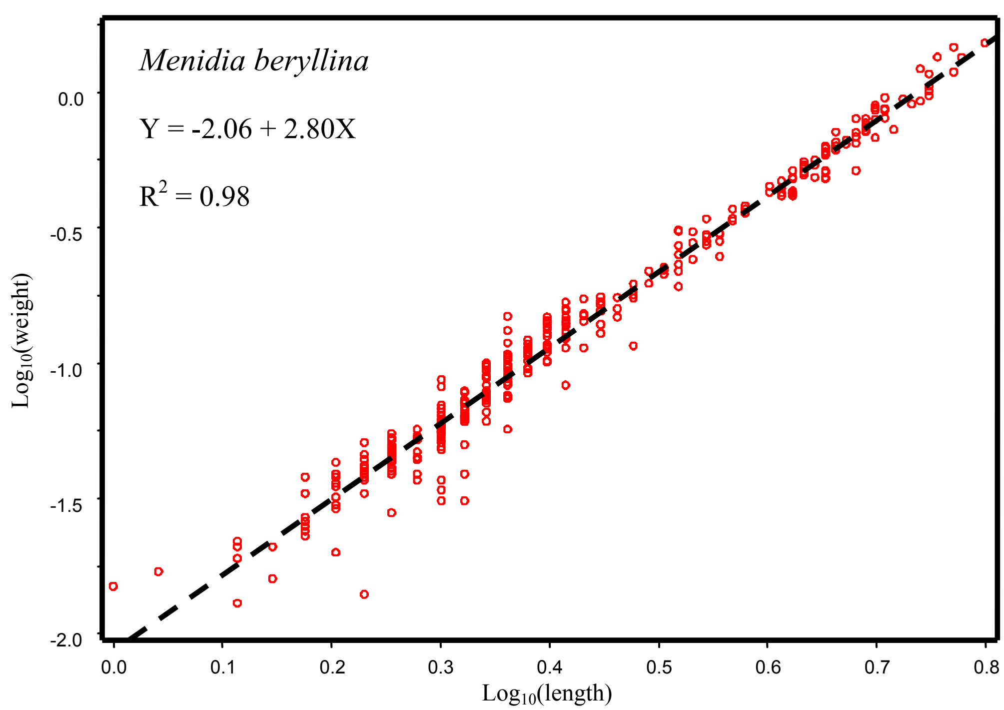


Figure A.3b. Simple Linear Regression on logarithmically transformed length-weight data for inland silversides.

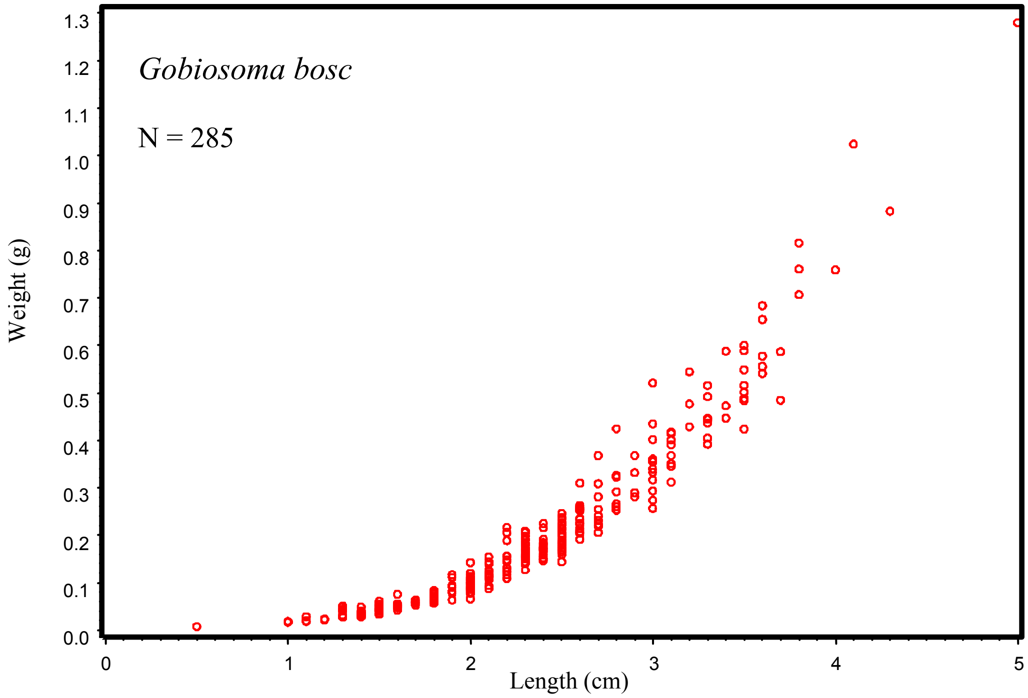


Figure A.4a. Plot of raw length-weight data for all naked gobies collected during study period.

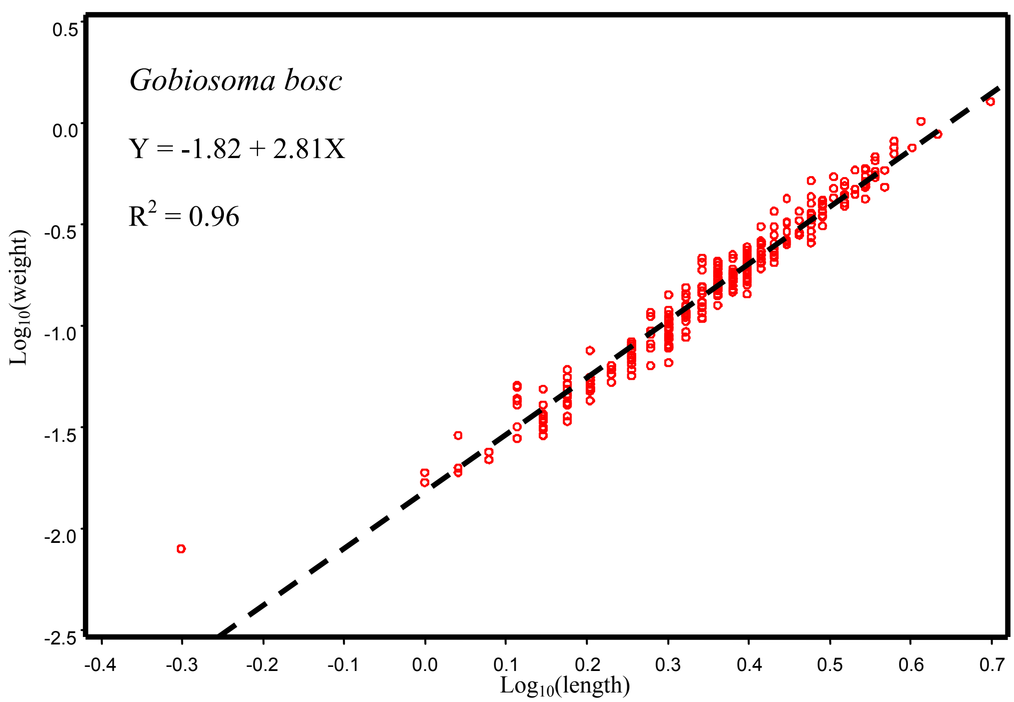


Figure A.4b. Simple Linear Regression on logarithmically transformed length-weight data for naked goby.

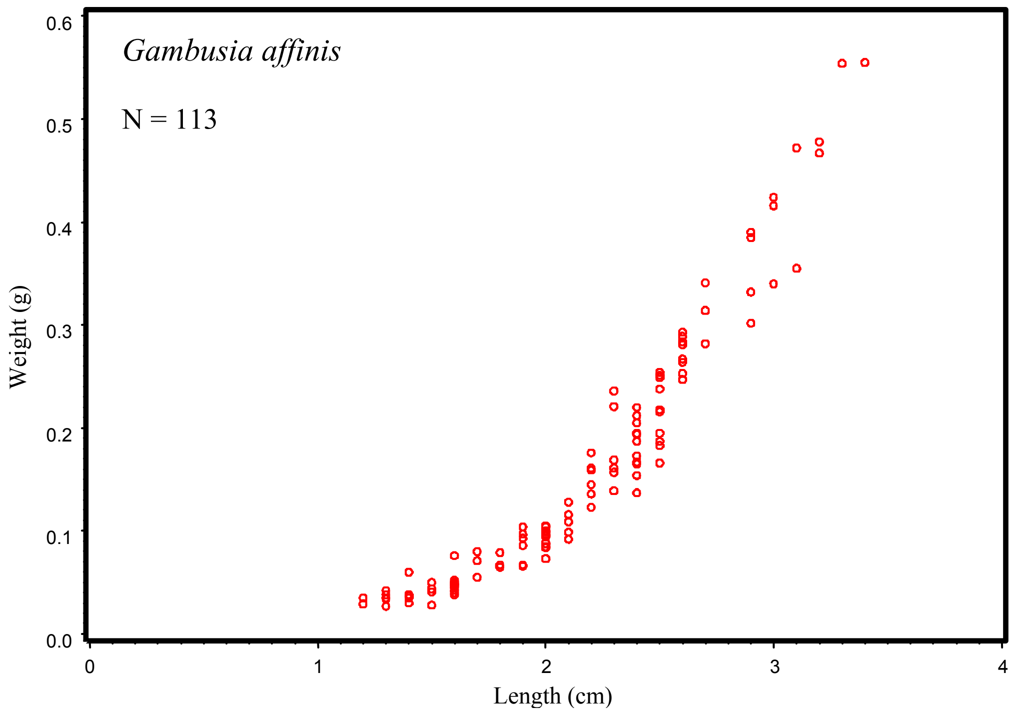


Figure A.5a. Plot of raw length-weight data for all western mosquitofish collected during study period.

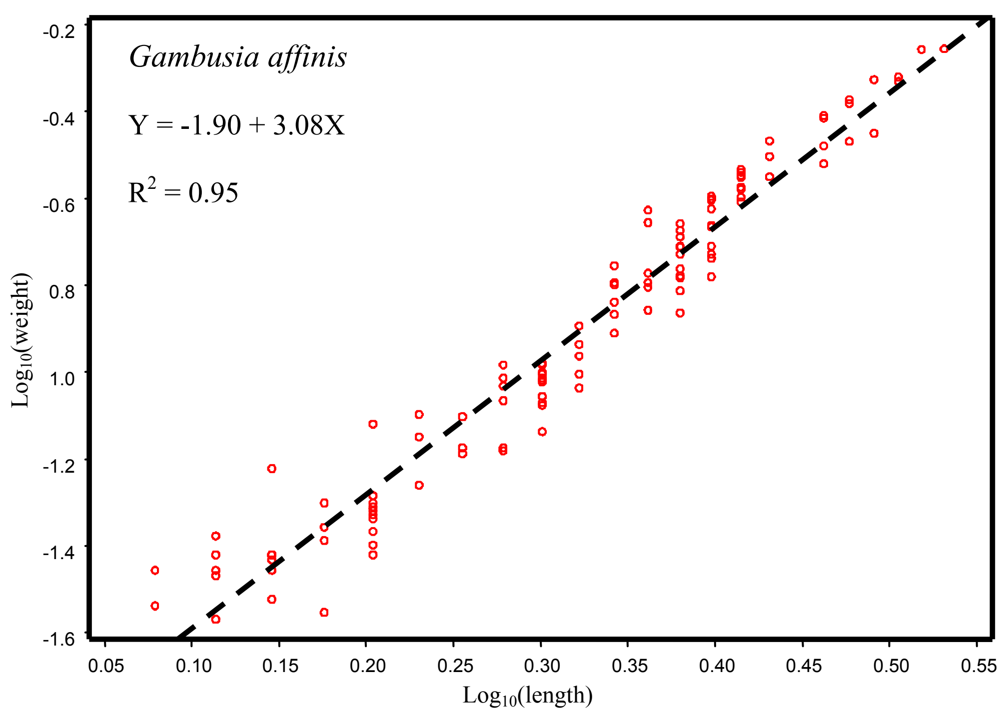


Figure A.5b. Simple Linear Regression on logarithmically transformed length-weight data for western mosquitofish.

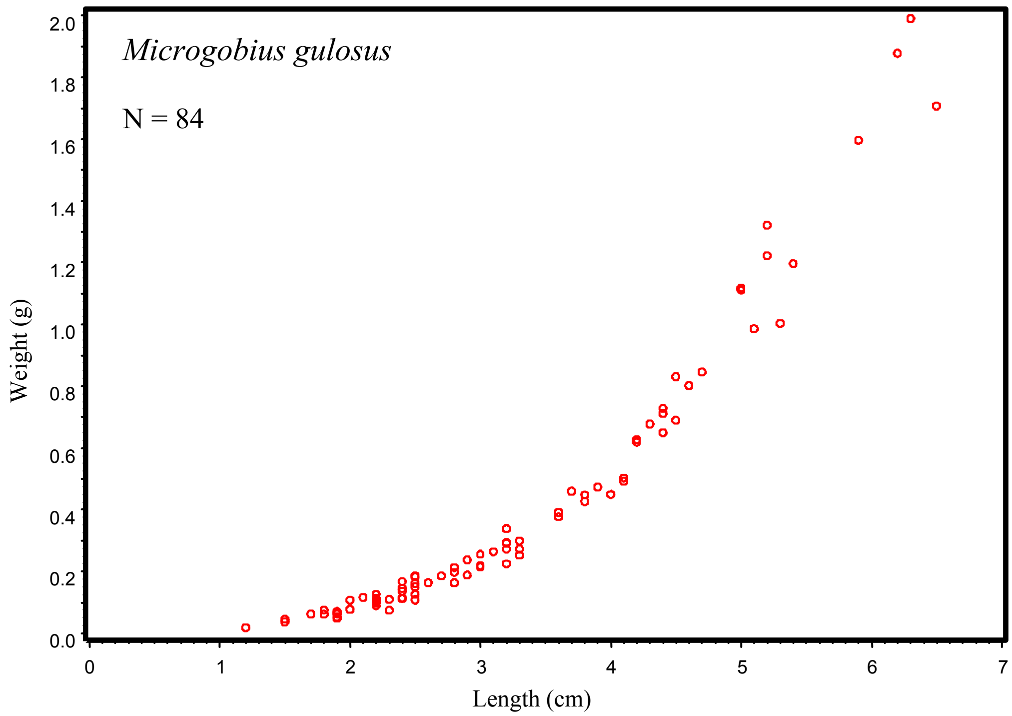


Figure A.6a. Plot of raw length-weight data for all clown gobies collected during study period.

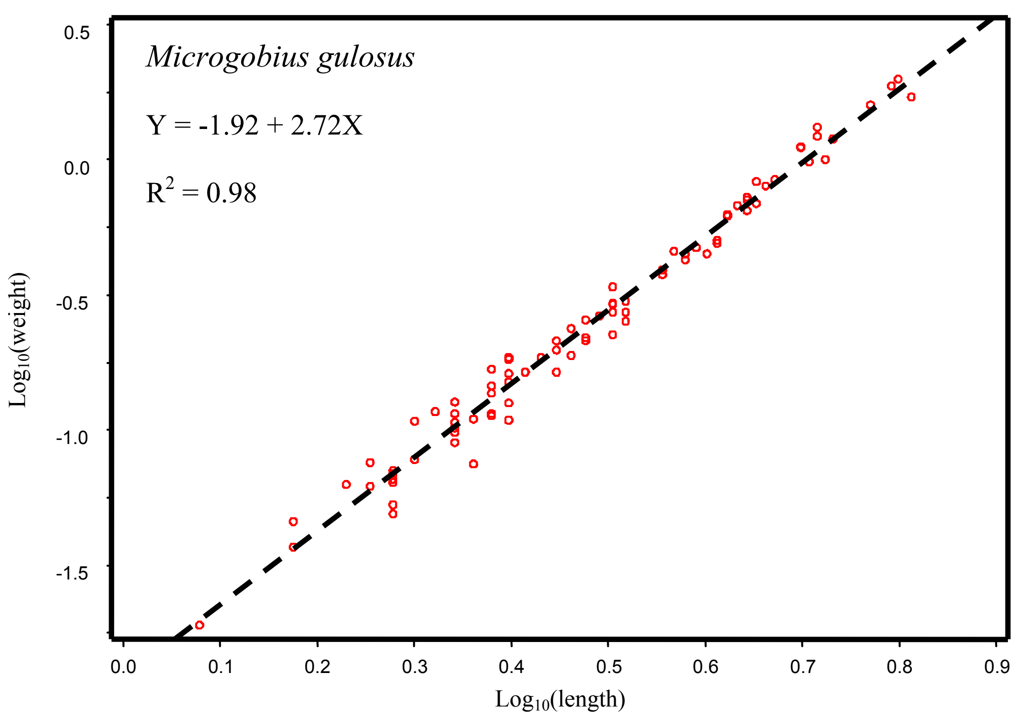


Figure A.6b. Simple Linear Regression on logarithmically transformed length-weight data for clown gobies.

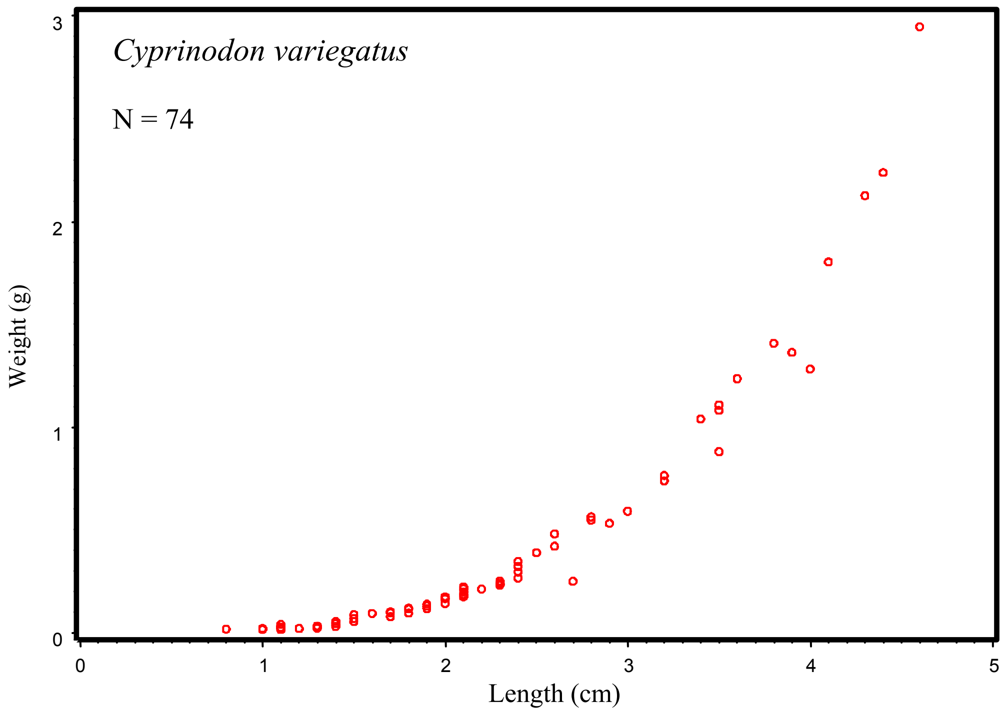


Figure A.7a. Plot of raw length-weight data for all sheepshead minnows collected during study period.

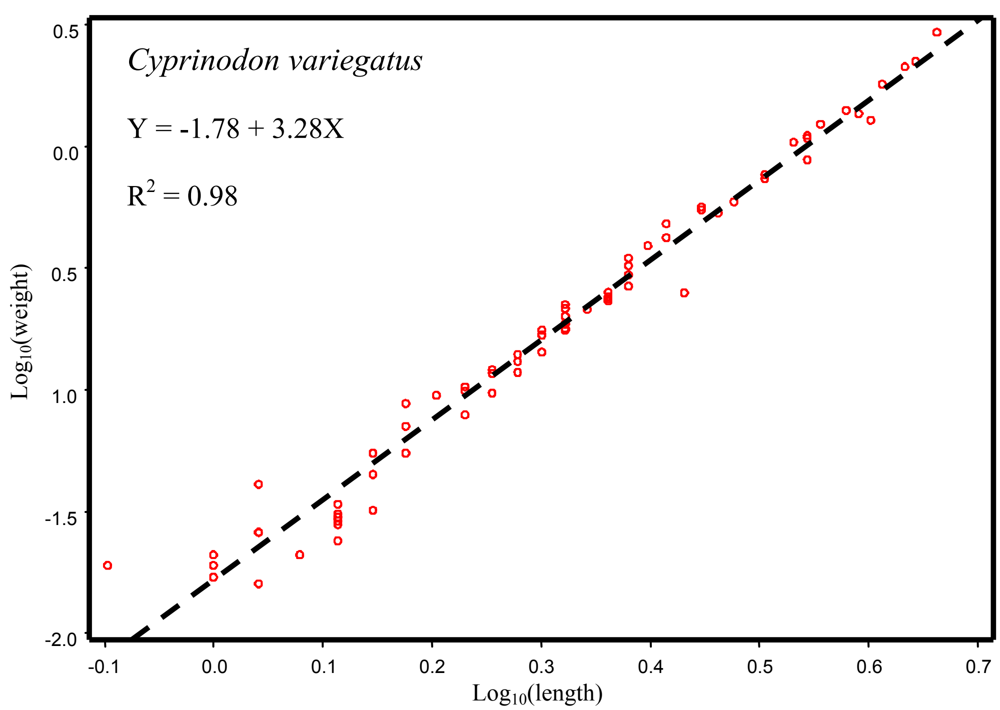


Figure A.7b. Simple Linear Regression on logarithmically transformed length-weight data for sheepshead minnow.

APPENDIX B

PROPOSED ADULT FISH STUDY

A supplemental study using gill nets as sampling gear was planned for May and August, 2005. The throw trap may be biased against some larger fishes because of their ability to avoid capture (Kushlan, 1981). Because the throw trap is selective for small nekton species and juvenile fishes, gill net sampling was necessary to determine which adult fishes were present in the terraced and unterraced study ponds.

Due to logistical problems, the August, 2005 samples were not collected. Because of the small sample size I have decided to exclude this study from my thesis, however I will present the preliminary data from the May, 2005 samples.

METHODS

Fish Sampling

The same study sites and sampling points used in the main study were used in the adult fish study. Each study site was sampled twice in May, 2005 for a total of 48 samples (N=48). The gill nets selected for the study were 10 m in length and consisted of four 2.5-m panels of 1.5, 2.0, 2.5, and 3.0-in stretch mesh (3.8, 5.1, 6.4, and 7.6-cm). Nets were set perpendicular to the marsh edge and because of the length of the gill nets, it was necessary to change our definition of edge to include the waterward area within 10 m of the emergent vegetation on the marsh or terrace. The gill nets were set within 2 h of sunrise and were left undisturbed for approximately 6 h before they were retrieved. Upon retrieval of the nets, individual fish were identified to species, measured to the nearest cm, weighed to the nearest 0.01 kg with a Berkley hanging electronic scale (Berkely, Spirit Lake, IA), and released alive when possible.

Environmental Characteristics

Water quality data was collected for each gill net sample at the time of net deployment. Salinity (ppt), conductivity (mS/cm), temperature (ºC), and dissolved oxygen (mg/L) were measured with a YSI model 556 multiparameter water quality meter (Yellow Springs Instruments, Yellow Springs, OH). Turbidity (NTU) was measured with a Turner Designs Aquafluor turbidimeter (Turner Designs, Sunnyvale, CA). Water depth was determined by calculating the mean of three depth measurements (cm) taken at both ends and the center of the gill net.

RESULTS

A total of 47 fish were collected representing 6 species (Table 6.1). The most frequently collected species of fish were black drum *Pogonias cromis* (16), spotted gar *Lepisosteus oculatus* (12), and alligator gar *Atractosteus spatula* (10).

Table B.1. Total catch and catch by pond type of fish collected during the gill net study.

Common name	**Scientific name**	**Study total**	**Terraced**	**Unterraced**
Black drum	*Pogonias cromis*	16	8	8
Spotted gar	*Lepisosteus oculatus*	12	6	6
Alligator gar	*Atractosteus spatula*	10	1	9
Striped mullet	*Mugil cephalus*	6	1	5
Gizzard shad	*Dorosoma cepedianum*	2	2	0
Blue catfish	*Ictalurus furcatus*	1	1	0
Total Fish		**47**	**19**	**28**

Mean catch per unit effort (CPUE, number of fish per gill net set) for terraced ponds was 0.8 ± 0.2 while CPUE for unterraced ponds was 1.2 ± 0.4. Mean CPUE was 1.0 ± 0.4 for edge habitats and 1.0 ± 0.3 for open water habitats.

Environmental and water quality characteristics were typical of brackish marsh environments (Table 6.2).

Table B.2. Water quality values (mean ± standard error) observed during the gill net study.

Variable	Overall	Terraced	Unterraced
Depth (cm)	37.5 ± 2.8	35.6 ± 2.8	39.5 ± 2.8
Salinity (ppt)	11.7 ± 1.1	12.4 ± 1.1	11.9 ± 1.2
Conductivity (mS/cm)	18.8 ± 1.7	19.8 ± 1.7	18.9 ± 1.8
Temperature (^{o}C)	23.9 ± 0.2	24.0 ± 0.2	23.9 ± 0.2
Dissolved O_2 (mg/L)	4.7 ± 0.2	4.6 ± 0.2	4.6 ± 0.2
Turbidity (NTU)	14.3 ± 1.7	14.3 ± 1.8	14.5 ± 1.9

DISCUSSION

There is little evidence to suggest any differences in catch per unit effort or environmental characteristics between pond types or habitat types, however it would be imprudent to speculate due to the limited sample size. Previous studies (Bush Thom 2004, Rozas and Minello 2001) have not addressed the issue of the presence of adult fish in terraced and unterraced marsh habitats due to limitations of the sampling gear used. Because of the incomplete nature of this study, there are still no studies that address the issue of adult fish. Future studies should include multiple sampling gears suited to catching nekton of various age and size classes in order to draw more complete conclusions as to the structure of nekton communities associated with various terraced and unterraced marsh pond habitats.

VITA

Bryan Paul Gossman was born in Pensacola, Florida. He graduated from Pensacola Catholic High School in 1996. Bryan attended the University of Florida, where he earned a Bachelor of Arts in environmental science in 2000. After graduation, he worked for the South Florida Water Management District in West Palm Beach, Florida, participating in field and greenhouse research to benefit Everglades restoration. Bryan came to Baton Rouge, Louisiana in the spring of 2004 to pursue a Master of Science in fisheries at Louisiana State University under the direction of Dr. Megan La Peyre. Following graduation, Bryan will remain at Louisiana State University to work as a Research Associate in the School of Renewable Natural Resources.