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Nekton use and growth in three brackish marsh pond microhabitats

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**NEKTON USE AND GROWTH IN THREE BRACKISH MARSH POND
MICROHABITATS**

A Thesis

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

in

The School of Renewable Natural Resources

by

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ABSTRACT

With continued marsh break-up and loss in Louisiana, small interior ponds are created, increasing areas of shallow water habitats. These shallow water habitats are potential sites for submerged aquatic vegetation (SAV) establishment. It is important to characterize nekton community composition, density, biomass, and growth within brackish marsh pond microhabitats because SAV is often cited as essential fish habitat (EFH). Three microhabitat types were investigated: (1) inner-pond SAV (> 1 m from edge); (2) near marsh-edge SAV (< 1 m from edge) and (3) nonvegetated bottom. We tested the null hypotheses that nekton community composition, density, and biomass were not related to microhabitat type and characteristics. Ninety-six quantitative samples were taken with a 1-m² throw trap between September 2001 and July 2002. The two vegetated microhabitats were characterized by monotypic stands of widgeon grass *Ruppia maritima* and contained similar biomass. Nekton community composition, density, and biomass did not differ between vegetated microhabitats, but differed significantly from the nonvegetated microhabitat ($p < 0.0001$). Therefore, SAV appears to be a dominant factor influencing nekton distribution within ponds. Submerged aquatic vegetation beds may also provide nekton with better growth environments by providing better quality or quantities of food for nekton than nonvegetated habitats. We also tested the null hypothesis that nekton growth was similar between vegetated and nonvegetated habitat types to determine if SAV provided a greater food resource than nonvegetated bottom. An *in situ* field experiment was conducted that compared growth rates of Atlantic croaker *Micropogonias undulatus* between vegetated and nonvegetated habitats to investigate the role of SAV in supporting nekton growth. We detected no statistically significant difference in nekton growth between vegetated and nonvegetated habitats ($p = 0.125$).

CHAPTER I.

GENERAL INTRODUCTION

A variety of different habitat types are essential during the lifetime of estuarine dependent nekton. In 1996, to help conserve and protect these essential fish habitats (EFH), Congress signed into law the Sustainable Fisheries Act (SFA). Congress defined EFH as "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity" (NOAA 1996). The SFA requires cooperation between the Federal and State governments to conserve, protect, and enhance EFH. The conservation of EFH, and thereby estuarine habitat types, is an important part of maintaining sustainable commercial and recreational fisheries.

Conserving and protecting EFH requires extensive information regarding all estuarine habitats and their influences on fisheries production. The National Marine Fisheries Service has developed four basic levels of information required for the identification of EFH. These include: species presence or absence and estimates of species abundance (level 1), evidence of habitat association (level 2), information on growth, survival, mortality, and trophic dynamics (level 3), and estimates of how each habitat type contributes to year class strength (level 4) (NMFS 1997).

Submerged aquatic vegetation (SAV) habitats are frequently cited as EFH. The physical structure of SAV provides nekton refuge and camouflage from predators and SAV supports abundant food resources that are thought to enhance nekton production. Nekton density, abundance, and diversity are often greater in SAV than in nonvegetated bottom habitats (Arrivillaga and Baltz 1999, Minello et al. 2003), possibly due to SAV contributing greater habitat value than nonvegetated habitats. Greater habitat value is thought to be related to the refuge and high quality growth environment provided by SAV. Because vegetated habitats support greater infaunal communities (benthic and epiphytic algae, detritus, and infaunal

organisms) than nonvegetated habitats (Phillips and McRoy 1980) food is more available in vegetated habitats and therefore, hypothesized to be a high quality growth environment.

Although several studies comparing nekton communities in SAV and nonvegetated habitat types have been conducted in Texas (Petrik et al. 1999, Stunz et al. 2002b, Minello et al. 2003) in Louisiana, the functional role of SAV in sustaining secondary production has not been well documented (Duffy and Baltz 1998, Castellanos and Rozas 2001).

As coastal marsh loss continues in Louisiana, emergent marshes are being converted to shallow open water habitats. These newly created shallow open water habitats become potential sites for SAV establishment. Thus, in Louisiana with continued marsh loss the contribution of SAV habitats to nekton production will likely become increasingly important. Given the importance of Louisiana's estuaries in supporting commercial and recreational fisheries, a better understanding of the potential role of SAV in supporting secondary production is needed. This thesis addresses that need. The primary goal was to examine the relative value of SAV for supporting nekton communities. Secondary goals included identifying characteristics of SAV beds (SAV biomass and location) that are important in supporting nekton communities, and examining the functional role by which SAV may support higher densities of nekton and diverse nekton communities.

The following chapters investigate the relative value of SAV for supporting nekton communities to address levels 1 and 2 (chapter 2) and level 3 (chapter 3) information needed for the identification, conservation, and protection of EFH. The objective of chapter 2 was to compare nekton communities (species, biomass, density, and diversity) among different habitat types (near-marsh edge SAV, inner-pond SAV, and nonvegetated bottom) within selected study ponds. Relationships of nekton community composition, nekton densities, nekton abundance,

nekton biomass, SAV biomass, and environmental variables were examined among vegetated and nonvegetated habitat types. Chapter 3 describes an investigation of the functional role of SAV in supporting nekton communities. Nekton growth was investigated to determine whether SAV habitat types support faster nekton growth than nonvegetated habitat due to increased food availability and supply. Chapters 2 and 3 were written as independent manuscripts. Each chapter has a separate Introduction, Methods, Results, and Discussion section. Chapter 4 is a synthesis of the previous chapters. Literature citations are repeated throughout the chapters, therefore a comprehensive literature cited section follows chapter 4.

CHAPTER II.

NEKTON USE OF THREE BRACKISH MARSH MICROHABITATS AT MARSH ISLAND WILDLIFE REFUGE, LOUISIANA

Introduction

Estuaries and tidal salt marshes are among the most productive ecosystems in the world (Mitsch and Gosselink 2000). Coastal areas provide essential habitat for 75% of the United States' commercial fish and shellfish landings. Estuaries in the northern Gulf of Mexico comprise habitat types (e.g., oyster reefs, submerged aquatic vegetation, salt marshes, marsh creeks, and mangroves) that help support the United States' second largest commercial fishery and substantial recreational fisheries (Louisiana Coastal Wetlands Conservation and Restoration Task Force 2001). These habitat types provide areas of increased production that estuarine dependent nekton (fishes and decapod crustaceans) use as nursery grounds to feed, spawn, and take refuge from predators (Boesch and Turner 1984, Zimmerman and Minello 1984, Peterson and Turner 1994, Levin et al. 1997).

A variety of different habitat types are essential as estuarine dependent nekton develop and mature. In 1996, to help conserve and protect these essential fish habitats (EFH), Congress signed into law the Sustainable Fisheries Act (SFA). Congress defined EFH as "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity" (NOAA 1996). The SFA requires cooperation between the Federal and State governments to conserve, protect, and enhance EFH. The conservation of EFH, and thereby estuarine habitat types, is an important part of maintaining sustainable commercial and recreational fisheries.

Conserving and protecting EFH requires extensive information regarding all estuarine habitats and their influences on fisheries production. The National Marine Fisheries Service developed four basic levels of information required for the identification of EFH. These include:

species presence or absence and estimates of species abundance (level 1), evidence of habitat association (level 2), information on growth, survival, mortality, and trophic dynamics (level 3), and estimates of how each habitat type contributes to year class strength (level 4) (NMFS 1997).

This study addresses levels 1 and 2 with regard to submerged aquatic vegetation (SAV), because it is often cited as EFH. The roles of SAV in supporting nekton communities have been well studied by comparing nekton densities among habitat types. Nekton densities can be influenced by numerous factors (e.g., recruitment, survival, emigration) therefore, nekton density can be an important indicator of nursery habitat value (Minello 1999). Comparing nekton densities among specific habitat types provides information on the relative habitat value of each habitat type. Submerged aquatic vegetation is assumed to be valuable habitat because it provides refuge, habitat complexity, and food for resident and estuarine dependent nekton.

Refuge habitat is essential for juvenile nekton while they are growing because they are small and can not move quickly to avoid predation. The structural heterogeneity of SAV habitats provides camouflage or hiding places for juvenile nekton to take refuge from predators (Rozas and Odum 1988). The structure of SAV also reduces maneuverability and catch efficiency of larger pelagic visual predators, thereby benefiting survival of smaller individuals (Wyda et al. 2002).

Food availability for nekton is generally greater in SAV habitats than in nonvegetated habitats (Phillips and McRoy 1980, Orth et al. 1984). Submerged aquatic vegetation habitats provide food for nekton by three major trophic pathways: direct herbivory of living plant material, secondary contributions to the detrital food web by decaying vegetation, and export of living and detrital biomass to the adjacent ecosystems (Heise and Bortone 1999). Submerged aquatic vegetation supports high densities and species diversity of benthic invertebrates because

the detrital cycle is a major food resource for benthic invertebrates. Consequently, high densities of benthic invertebrates present in SAV beds provide rich food sources for epibenthic nekton (Summerson and Peterson 1984). The physical structure of SAV also facilitates greater densities of benthic invertebrates because SAV inhibits predation on benthic invertebrates and the baffling action of vegetation blades trap invertebrate larvae and food (Tegner and Dayton 1981). The increased food supply within SAV habitats may provide nekton with a growth advantage allowing them to become larger individuals over a short period of time. Larger nekton are more mobile and thus able to avoid predation to a greater extent than smaller individuals. Therefore, the greater food availability within SAV habitats could increase nekton survival.

It is important to understand the functional role SAV plays in sustaining secondary production because it has not been well documented in Louisiana (Duffy and Baltz 1998, Castellenos and Rozas 2001). The contribution of SAV habitats to nekton production will likely become increasingly important with continued land loss in Louisiana. As land loss and marsh degradation continue, emergent marsh is converted to open water habitat. Intermediate stages of marsh degradation are likely beneficial to fisheries production due to increased edge habitat and increased amounts of suitable SAV habitat (Chesney et al. 2000). Given the potential importance of Louisiana's remaining SAV to recreational and commercial fisheries and to coastal restoration a better understanding is needed of the role of SAV in supporting secondary production.

Research Objectives

The primary goal was to examine the relative habitat value of SAV and to identify the characteristics of SAV habitats (density, biomass, and location) that enable them to support nekton communities. The objectives of this study were to compare nekton assemblages

(diversity, abundance, density, and biomass) among different habitat types: 1) near marsh edge SAV, 2) inner-pond SAV, and 3) nonvegetated bottom. This study tested the null hypotheses 1) nekton community composition was not related to habitat type and habitat characteristics; 2) nekton density and biomass were not related to habitat type and habitat characteristics.

Methods

Study Area

This study was conducted within Marsh Island Wildlife Refuge (MIWR) located in Vermilion Bay, Louisiana. The island consists of 28,300 ha of tidally influenced brackish marsh. The Refuge has lost approximately 2,700 ha of emergent marsh due to erosional processes since originally deeded to the state. Since 1949, emergent vegetation types have changed from saline and brackish species (e.g., oyster grass *Spartina alterniflora*, black rush *Juncus roemerianus*, and wiregrass *Spartina patens*) to brackish and intermediate species in 1997 (e.g., *Spartina patens*, three-cornered grass *Schoenoplectus pungens*, and sawgrass *Cladium jamaicense*) (Linscombe et al. 1998). The emergent marsh vegetation is currently dominated by marshhay cordgrass *Spartina patens*. The Louisiana Department of Wildlife and Fisheries manages areas within MIWR for increased SAV abundance to support overwintering waterfowl, nesting shorebird habitat, and revegetation to counteract marsh loss. The study was conducted within the unmanaged areas of MIWR to avoid issues related to nekton access into managed areas.

Monospecific stands of widgeon grass *Ruppia maritima* were common in unmanaged areas. Widgeon grass is a submerged aquatic vegetation species with worldwide distribution. *Ruppia maritima* tolerates a wide range of environmental conditions. This species is eurythermic and it survives temperatures ranging from 7 to 39°C. Temperatures between 20 and 25°C are optimum for growth (Thursby 1984). *Ruppia maritima* is typically classified as a brackish water

species occurring most frequently below 25 ppt (Brock 1979), but this species can tolerate fresh to hypersaline waters (Pip 1979). It is most abundant in shallow brackish water with low turbidity and relatively stable water depths. The seeds, leaves, and stems are important food for overwintering waterfowl on the Louisiana coast (Chabreck and Condrey 1979).

Sampling Design

The study was conducted using a stratified random sampling design. Samples were collected from eight randomly selected ponds (Figure 2.1). Study ponds included a range of pond sizes and numbers of tidal channels that represented the natural variability of the system. Large and small tidal channels are common throughout MIWR connecting lakes to a complex system of ponds. The study ponds ranged from 0.20 ha to 1.87 ha, and one to five tidal channels connected each pond to the system.

In each of the eight ponds, three habitat types were randomly sampled: inner-pond SAV (IP: SAV > 1 m from emergent marsh edge), near-marsh edge (NME: SAV < 1 m from emergent marsh edge), and nonvegetated bottom (NB) for a total of 24 samples per collection period (8 ponds by 3 habitat types). Each of the eight ponds was sampled four times (9/2001, 12/2001, 3/2002, 7/2002) for a total of 96 samples (24 samples per collection by 4 time periods).

All sampling occurred when the emergent marsh was not flooded and nekton was concentrated in marsh ponds. A 1-m² aluminum-sided throw trap similar to that described in Kushlan (1981) was used to quantitatively sample the nekton community. Sample sites were randomly selected within the three microhabitat types (NME, IP, NB). Sweeps with a 1-m wide bar seine (3 mm mesh size) were made to remove nekton from the trap. Five consecutive sweeps without collecting organisms were completed before the trap was considered free of nekton. Nekton was frozen and returned to the laboratory where it was sorted and identified to

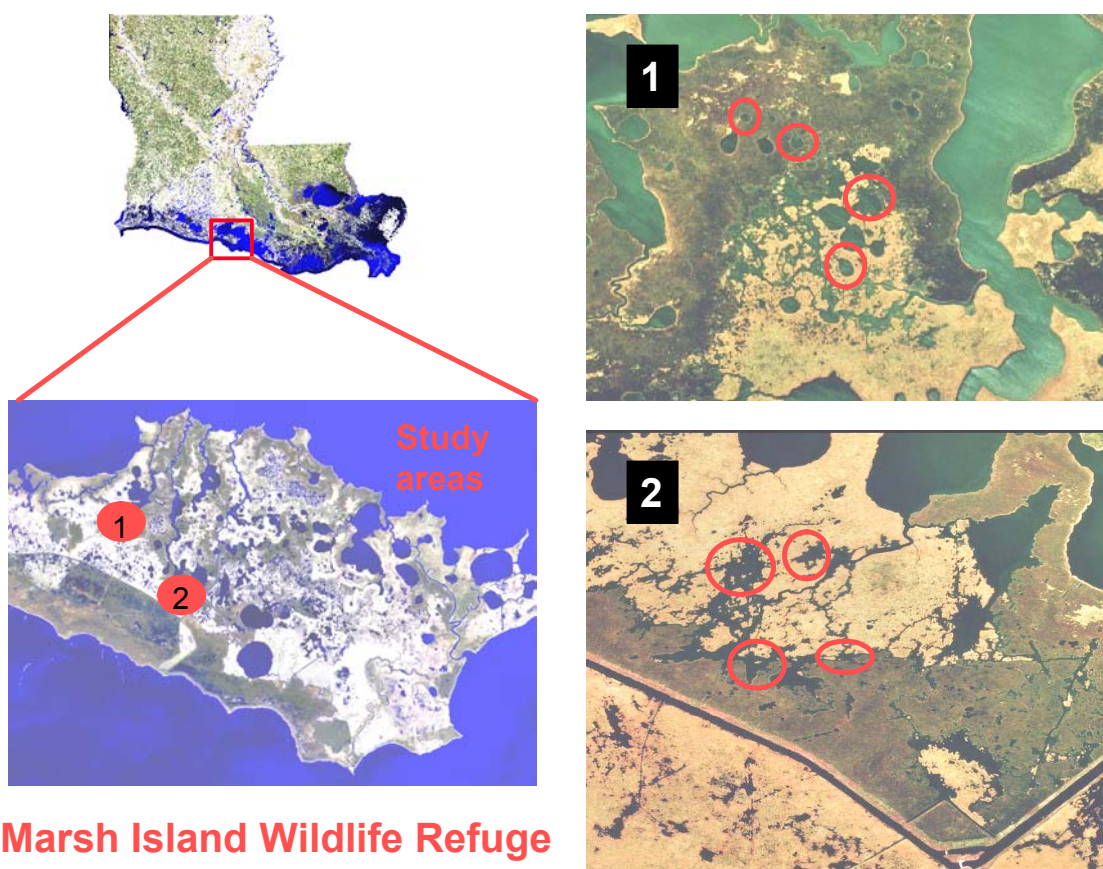


Figure 2.1. Study areas at Marsh Island Wildlife Refuge, Vermilion Bay, Louisiana.

species or the lowest possible taxon. Total length (mm) of fish and shrimp and carapace width (mm) of crabs were measured. All nekton were weighed to the nearest 0.001 g wet-weight to determine biomass (g wet wt m⁻²). Density (individuals m⁻²) was calculated for each throw trap sample. In addition, nekton community diversity was determined using Shannon Wiener diversity index (H'). The Shannon Wiener diversity index was calculated from the following equation: $H' = -\sum p_i \ln p_i$, where p_i is the proportion of individuals found in the i th species (Magurran 1988).

In throw trap samples containing SAV, the vegetation was removed prior to removal of the nekton. Vegetation was removed from within the throw trap by hand and rake, carefully removing both above and below ground vegetation. The SAV was refrigerated and returned to

the laboratory for processing. The vegetation was dried to a constant weight at 65°C and weighed to the nearest 0.001 g dry-weight to determine biomass (g dry wt m⁻²).

Emergent vegetation adjacent to NME sampling sites was sampled with a 0.25-m² quadrat. Triplicate samples were collected at random from each study pond. Above ground vegetation was clipped to within 1 cm of the substrate. In the laboratory, vegetation was sorted taxonomically to species. Stems were enumerated and vegetation was dried to a constant weight at 65°C and weighed to the nearest 0.01 g dry-weight to determine biomass (g dry wt m⁻²).

Environmental variables were measured for each sample, including mean water depth (cm), distance to the emergent marsh edge (m), and Secchi disk depth (cm). Other environmental variables measured, with a YSI Model 556 water quality monitor, included water temperature (°C), salinity (ppt), and dissolved oxygen (mg L⁻¹). All water quality measurements were taken within each sample location following throw trap deployment.

Statistical Analyses

Multivariate analysis of variance (MANOVA) was used to test the null hypotheses that nekton biomass, nekton density, nekton diversity, and SAV biomass, examined together, were equal among ponds, months, habitats, and month by habitat interactions (SAS Institute 1989). Following significant MANOVA results, separate univariate analysis of variance (ANOVA, Proc GLM) with *a posteriori* Tukey tests was used to test the null hypotheses that individual variables (i.e. nekton biomass, nekton density, nekton diversity, SAV biomass) were equal among ponds, months, and habitats, and month by habitat interactions. Analysis of covariance (ANCOVA, Proc GLM) was used to test the null hypotheses that nekton biomass and SAV biomass were equal among ponds, months, habitats, distance, and month by habitat interactions. Simple linear regression (SLR, Proc REG) was used to examine the potential relationship between SAV

biomass (independent predictor variable) and nekton biomass (dependent response variable). Prior to statistical analyses, nekton biomass data were $\log(x + 1)$ transformed to address the assumptions of normality and equal variance. An alpha level of 0.05 was used to determine significance for all statistical analyses.

Results

Study Site

Environmental Variables

Water temperatures ranged from 16.0°C in March 2002 to 33.0°C in July 2002. Mean water temperature throughout the study was $24.52 \pm 0.57^\circ\text{C}$ (Table 2.1). Dissolved oxygen within study ponds varied from hypoxic to normoxic conditions. Mean dissolved oxygen was lowest in September 2001 (2.03 mg L^{-1}) and highest in December 2001 (7.16 mg L^{-1}). Overall mean dissolved oxygen for all sampling periods was $5.03 \pm 0.26 \text{ mg L}^{-1}$. Salinities were typical of brackish marsh systems. Salinity ranged from 1.3 ppt in July 2002 to 6.2 ppt in March 2002. Throughout the study, mean salinity was 4.10 ± 0.21 ppt. Water depths taken within the throw trap samples ranged from 8.6 cm to 53.0 cm averaging 31.77 ± 1.09 cm. Distance of the throw trap samples to the emergent marsh edge ranged from 0.5 m to 55.0 m. Mean distance to the emergent marsh edge was 8.06 ± 0.04 m. Secchi disk depths ranged from 4.0 cm to 44.3 cm with a mean Secchi depth of 19.35 ± 0.87 cm.

TABLE 2.1. Environmental variables at Marsh Island Wildlife Refuge between September 2001 and July 2002. Maximum and minimum values with overall study means \pm standard errors.

Date	Water temp. (°C)	DO (mg L^{-1})	Salinity (ppt)	Water depth (cm)	Distance to marsh edge (m)	Secchi (cm)
Sept. 2001	29.1-32.2	1.6-2.3	3.4-3.7	32.0-53.0	0.5-50.0	25.0-43.5
Dec. 2001	19.0-24.2	2.9-14.7	4.9-5.2	12.0-44.3	0.5-55.0	12.0-44.3
March 2002	16.0-21.7	2.3-6.3	5.4-6.2	8.6-49.6	0.5-32.0	4.0-19.6
July 2002	28.1-33.0	2.2-6.3	1.3-1.7	14.3-50.6	0.6-20.0	13.0-30.0
Mean \pm SE	24.52 ± 0.57	5.03 ± 0.26	4.10 ± 0.21	31.77 ± 1.09	8.06 ± 0.04	19.35 ± 0.87

Water temperature, dissolved oxygen, salinity, water depth, distance to marsh edge, and Secchi depth were significantly different among habitat types (Wilks' lambda = 0.20, $F_{12, 108} = 11.13$; $p < 0.0001$), months (Wilks' lambda = 0.00, $F_{18, 153} = 95.37$; $p < 0.0001$), and ponds (Wilks' lambda = 0.28, $F_{45, 257} = 1.88$; $p = 0.0016$), but there was no interaction of month and habitat (Wilks' lambda = 0.63, $F_{24, 190} = 1.11$; $p = 0.34$). Based on analysis of variance, habitat types differed significantly for three environmental variables: water depth (cm), distance to marsh edge (m), and Secchi depth (cm). Vegetated habitats were significantly shallower (NME = 27.29 ± 2.12 cm, IP = 31.45 ± 1.68 cm) than nonvegetated bottom (36.57 ± 1.49 cm) ($p = 0.0006$) (Figure 2.2). The vegetated habitats were also significantly closer to the emergent marsh edge (NME = 0.82 ± 0.04 m, IP = 3.91 ± 1.10 m) than nonvegetated habitat (20.18 ± 2.33 m) ($p < 0.0001$) (Figure 2.2). Secchi depth was significantly less in the NME (15.67 ± 1.23 cm) habitat than in the IP (21.14 ± 1.36 cm) and nonvegetated bottom (20.80 ± 1.73 cm) habitats ($p = 0.0357$) (Figure 2.2). No significant differences in water temperature ($^{\circ}\text{C}$), dissolved oxygen (mg L^{-1}), and salinity (ppt) were detected among habitat types ($p > 0.05$) (Table 2.2).

Water temperature, dissolved oxygen, salinity, water depth, distance to marsh edge and secchi depth differed significantly by sampling month ($p < 0.0001$, Proc GLM, SAS).

Water temperature, dissolved oxygen, water depth, distance to marsh edge, and Secchi depth were similar among ponds ($p > 0.05$, Proc GLM, SAS). However, salinity differed statistically among ponds ($p = 0.0003$). Differences among ponds could not be explained by variation in pond size, number of tidal channels, or amount of emergent marsh edge.

Emergent Vegetation

Total Abundance

A total of 13 species of emergent vegetation was collected adjacent to the study ponds (Table 2.3). The most abundant species were marshhay cordgrass *Spartina patens* (67.0%),

three-cornered grass *Schoenoplectus americanus* (18.4%), and salt grass *Distichlis spicata* (10.3%).

Number of Stems

The mean number of stems was significantly greater ($p = 0.0005$) in March (202.80 ± 34.48 stems) than in September (62.36 ± 18.25 stems), December (89.57 ± 16.16 stems), and July (81.63 ± 18.31 stems) (Figure 2.3).

Biomass

Biomass, for 0.25 m^2 quadrats, was significantly greater in July 2002 ($98.20 \pm 22.15 \text{ g dry wt m}^{-2}$) than in September 2001 ($29.91 \pm 7.59 \text{ g dry wt m}^{-2}$). However, emergent vegetation biomass was similar December 2001 ($47.52 \pm 7.72 \text{ g dry wt m}^{-2}$) and March 2002 ($67.65 \pm 17.67 \text{ g dry wt m}^{-2}$) and did not differ from September 2001 ($p = 0.02$) (Figure 2.4).

Nekton Biomass and SAV Biomass

Analysis of covariance indicated nekton biomass ($\log(x + 1)$ transformed) did not differ significantly with distance from the emergent marsh edge ($p = 0.3969$). Likewise, SAV biomass did not differ significantly by distance from the emergent marsh edge ($p = 0.3242$).

Simple linear regression showed a significant relationship between nekton biomass ($\log(x + 1)$ transformed) and SAV biomass ($r^2 = 0.46$; $p < 0.0001$) (Figure 2.5). Nekton biomass and SAV biomass were positively related (nekton biomass = $1.1208 + 0.053 \text{ SAV biomass}$).

Nekton

Total Nekton Abundance

A total of 5041 organisms (28 species) were collected from 96 throw trap samples. Fishes represented the majority of the catch (77.4% of the total) and 24 species. Four decapod crustacean species were collected composing 22.6% of the total abundance (Table 2.4). The majority of nekton species were more abundant within vegetated habitats (NME and IP) than

over nonvegetated bottom. Four species were more abundant in nonvegetated bottom; skipjack herring *Alosa chrysochloris*, bay anchovy *Anchoa mitchilli*, Atlantic croaker *Micropogonias undulatus*, and red drum *Sciaenops ocellatus*, however these species only accounted for 1.7% of the total nekton abundance. Bay anchovy *Anchoa mitchilli*, Gulf menhaden *Brevoortia patronus*, and blue crab *Callinectes sapidus* were similarly abundant among all habitat types (Table 2.4).

Total Nekton Biomass

Total nekton biomass was greatest in July 2002 (731.21 g wet wt m⁻²) followed by September 2001 (346.55 g wet wt m⁻²), December 2002 (188.27 g wet wt m⁻²), and March 2002 (64.06 g wet wt m⁻²) (Table 2.5). Near-marsh edge habitat contributed the most to the total biomass (802.10 g wet wt m⁻²), followed by inner-pond SAV (466.92 g wet wt), and nonvegetated bottom (61.06 g wet wt m⁻²). Blue crab *Callinectes sapidus* composed 25.7% of the total nekton biomass, followed by the biomass contributed by the six most abundant nekton taxa. Although blue crab *Callinectes sapidus* was not in the top six numerically dominant nekton species, several large individuals contributed greatly to overall nekton biomass (Table 2.5). The six numerically dominant taxa composed 60.4% of the total nekton biomass collected throughout the study. Rainwater killifish *Lucania parva* contributed 17.9% of the total nekton biomass followed by sheepshead minnow *Cyprinodon variegatus* (15.3%), grass shrimp *Palaemonetes* spp. (9.5%), sailfin molly *Poecilia latipinna* (7.2%), white shrimp *Litopenaeus setiferus* (6.9%), and bayou killifish *Fundulus pulvereus* (3.6%) (Table 2.5).

Numerically Dominant Taxa Abundance and Density

Six taxa accounted for 92.7% of the total nekton abundance. Rainwater killifish *Lucania parva* was the most abundant nekton species (29.9%), followed by sheepshead minnow *Cyprinodon variegatus* (19.7%), grass shrimp *Palaemonetes* spp. (17.1%), sailfin molly *Poecilia latipinna* (17.0%), white shrimp *Litopenaeus setiferus* (4.7%), and bayou killifish *Fundulus*

pulvereus (4.3%) (Table 2.4). Of the numerically dominant taxa, five were most abundant in July 2002; rainwater killifish *Lucania parva*, sheepshead minnow *Cyprinodon variegatus*, grass shrimp *Palaemonetes* spp., sailfin molly *Poecilia latipinna*, and bayou killifish *Fundulus pulvereus* (Figure 2.6), while white shrimp *Litopenaeus setiferus* was most abundant in September 2001. All of the numerically dominant taxa were more abundant in vegetated habitat types (NME and IP) than nonvegetated bottom habitat types (Figure 2.7).

Among the numerically dominant taxa, resident and transient nekton species followed the same trend of nekton being more dense (individuals m⁻²) in vegetated habitats than over nonvegetated bottom. Densities of rainwater killifish *Lucania parva*, grass shrimp *Palaemonetes* spp., and sailfin molly *Poecilia latipinna* were significantly greater in vegetated habitat types (NME and IP) than in nonvegetated bottom habitat types ($p < 0.05$) (Figure 2.8). Densities of sheepshead minnow *Cyprinodon variegatus* were greater in vegetated habitat types, but IP densities were similar to NME and NB ($p < 0.05$). Bayou killifish *Fundulus pulvereus* densities were lower than other numerically dominant species and were similar among all three habitat types ($p = 0.1789$). Densities for the only transient species, white shrimp *Litopenaeus setiferus*, were greater in vegetated habitat types, but densities in NME were similar to both IP and NB habitat types ($p < 0.05$).

Microhabitat Use and Seasonal Patterns

Mean nekton biomass (log ($x + 1$) transformed), mean nekton density, mean nekton diversity, and mean SAV biomass were significantly different among ponds (Wilks' lambda = 0.44, $F_{28,268} = 2.45$; $p < 0.0001$), months (Wilks' lambda = 0.26, $F_{12,196} = 10.94$; $p < 0.0001$), habitat types (Wilks' lambda = 0.34, $F_{8,148} = 13.00$; $p < 0.0001$), and the interaction of months and habitats (Wilks' lambda = 0.32, $F_{24,259} = 4.21$; $p < 0.0001$).

Nekton Density

Analysis of variance indicated mean nekton densities differed significantly among the interactions of months and habitats ($p < 0.0001$) (Figure 2.9). In September 2001, mean nekton densities differed among all habitats: NME (27.25 ± 6.58 individuals m^{-2}), IP (81.88 ± 15.90 individuals m^{-2}), NB (1.75 ± 1.33 individuals m^{-2}). In December 2001, mean nekton densities in the vegetated habitats, NME (22.63 ± 8.72 individuals m^{-2}) and IP (13.50 ± 2.86 individuals m^{-2}), were similar but differed from nonvegetated bottom habitats (7.63 ± 2.54 individuals m^{-2}). In March 2002, mean nekton density did not differ among habitat types NME (9.00 ± 3.30 individuals m^{-2}), IP (7.00 ± 3.73 individuals m^{-2}), NB (5.38 ± 1.81 individuals m^{-2}). In July 2002, mean nekton density in the vegetated habitats, NME (249.25 ± 47.49 individuals m^{-2}) and IP (203.75 ± 45.96 individuals m^{-2}), were similar, but differed from nonvegetated habitat (1.13 ± 0.67 individuals m^{-2}).

Analysis of variance testing mean nekton densities among months showed significant seasonal differences ($p < 0.0001$). Mean nekton density was significantly greater in July 2002 (151.38 ± 30.80 individuals m^{-2}) than in September 2001 (36.96 ± 8.88 individuals m^{-2}), December 2001 (14.58 ± 3.29 individuals m^{-2}), and March 2002 (7.13 ± 1.72 individuals m^{-2}) (Figure 2.10).

Analysis of variance testing mean nekton densities among habitats showed significantly greater mean nekton densities in vegetated habitats, NME (77.03 ± 21.33 individuals m^{-2}) and IP (76.53 ± 18.34 individuals m^{-2}), than nonvegetated bottom (3.97 ± 0.95 individuals m^{-2}) ($p < 0.0001$) (Figure 2.11). See Appendix A for nekton densities by habitat and month.

Nekton Biomass

Analysis of variance indicated mean nekton biomass ($\log(x + 1)$ transformed) (g wet wt m^{-2}) showed a significant interaction of months and habitats ($p < 0.0001$) (Figure 2.12). In

March 2002, mean nekton biomass was similar among all habitats (NME = 1.04 ± 0.33 g wet wt m^{-2} , IP = 0.62 ± 0.20 g wet wt m^{-2} , NB = 0.97 ± 0.36 g wet wt m^{-2}). In September 2001, December 2001, and July 2002 mean nekton biomass differed among habitats. Biomass in the two vegetated habitats was similar (NME = 2.02 - 3.76 g wet wt m^{-2} and IP = 1.71 - 3.12 g wet wt m^{-2}) and was significantly greater than in nonvegetated habitat (NB = 0.25 - 0.88 g wet wt m^{-2}).

Analysis of variance testing mean nekton biomass ($\log(x + 1)$ transformed) among months indicated that significant seasonal variation occurred ($p < 0.0001$) (Figure 2.13). Mean nekton biomass was greatest in July 2002 (2.46 ± 0.35 g wet wt m^{-2}) and least in March 2002 (0.88 ± 0.17 g wet wt m^{-2}). Mean nekton biomass in September 2001 (1.88 ± 0.29 g wet wt m^{-2}), was similar to July 2002 and December 2001 (1.54 ± 0.22 g wet wt m^{-2}).

Analysis of variance testing mean nekton biomass ($\log(x + 1)$ transformed) among habitats indicated that vegetated habitats produced similar mean nekton biomass, but differed significantly from nonvegetated bottom habitats ($p < 0.0001$) (Figure 2.14). Vegetated habitats (NME = 2.36 ± 0.25 g wet wt m^{-2} and IP = 2.06 ± 0.23 g wet wt m^{-2}) produced significantly greater mean nekton biomass than nonvegetated bottom habitats (NB = 0.80 ± 0.14 g wet wt m^{-2}). See Appendix B. for nekton biomass (untransformed) data by habitat and month.

Nekton Diversity

Analysis of variance showed a significant interaction of month and habitats for mean nekton diversity (Shannon Wiener H') ($p < 0.0001$) (Figure 2.15). In September 2001, mean diversities in the vegetated habitats, NME (1.19 ± 0.13) and IP (1.26 ± 0.12), were similar but differed from nonvegetated bottom (0.09 ± 0.09). In December 2001, mean nekton diversities in the vegetated habitats, NME (1.00 ± 0.11) and IP (1.10 ± 0.13), were similar but differed from nonvegetated bottom (0.64 ± 0.19). In March 2002, mean nekton diversities did not differ

among habitat types NME (0.66 ± 0.21), IP (0.61 ± 0.21), NB (0.62 ± 0.16). In July 2002, mean nekton diversities in the vegetated habitats, NME (1.13 ± 0.13) and IP (1.04 ± 0.19), were similar, but differed from nonvegetated habitat (0.00).

Analysis of variance testing mean diversity (Shannon Wiener, H') among months showed no seasonal differences in diversity ($p = 0.0603$) (Figure 2.16). However, diversity did differ significantly by habitat ($p < 0.0001$) (Figure 2.17). Vegetated habitats NME ($H' = 0.99 \pm 0.08$) and IP ($H' = 1.00 \pm 0.09$) contained significantly more diverse nekton communities than nonvegetated bottom habitat ($H' = 0.34 \pm 0.08$).

Pond Effect

Significant differences among ponds occurred for nekton biomass ($\log(x + 1)$ transformed) ($p < 0.0001$), nekton abundance ($p = 0.02$), and nekton diversity ($p = 0.0002$), and SAV biomass ($p = 0.0076$). Differences among ponds could not be explained by variation in pond size, number of tidal channels, or amount of emergent marsh edge.

Submerged Aquatic Vegetation

Microhabitat Use and Seasonal Patterns

Biomass

Study ponds supported monospecific stands of widgeon grass *Ruppia maritima*. Analysis of variance indicated mean SAV biomass showed significant interactions among months and habitats ($p < 0.0001$) (Figure 2.18). In March 2002, mean SAV biomass was similar among all habitats (NME = 0.00 g dry wt m^{-2} , IP = 0.01 ± 0.01 g dry wt m^{-2} , NB = 0.00 g dry wt m^{-2}). In September 2001, mean SAV biomass differed among all habitats NME (5.68 ± 3.06 g dry wt m^{-2}), IP (25.84 ± 3.67), and NB (0.00 g dry wt m^{-2}). In December 2001 and July 2002, mean SAV biomass in vegetated habitats (NME = 8.99 - 38.66 g dry wt m^{-2} and IP = 15.73 - 33.94 g

TABLE 2.2. Environmental variables (mean \pm SE) for sampling habitats: near-marsh edge SAV, inner-pond SAV, and nonvegetated bottom. P-values from general linear model ANOVA with Tukey *post hoc* adjustment.

Environmental variables	Near-marsh SAV	Inner-pond SAV	Nonvegetated bottom	p > F
Temperature (°C)	24.00 \pm 1.03	25.25 \pm 0.94	24.11 \pm 1.01	0.8996
Salinity (ppt)	4.12 \pm 0.38	4.07 \pm 0.32	4.16 \pm 0.39	0.6354
DO (mg L ⁻¹)	5.01 \pm 0.46	4.65 \pm 0.50	5.52 \pm 0.34	0.7014
Secchi depth (cm)	15.67 \pm 1.23	21.14 \pm 1.36	20.80 \pm 1.73	0.0357
Water depth (cm)	27.29 \pm 2.12	31.45 \pm 1.68	36.57 \pm 1.49	0.0006
Distance to marsh edge (m)	0.82 \pm 0.04	3.91 \pm 1.10	20.18 \pm 2.33	< 0.0001

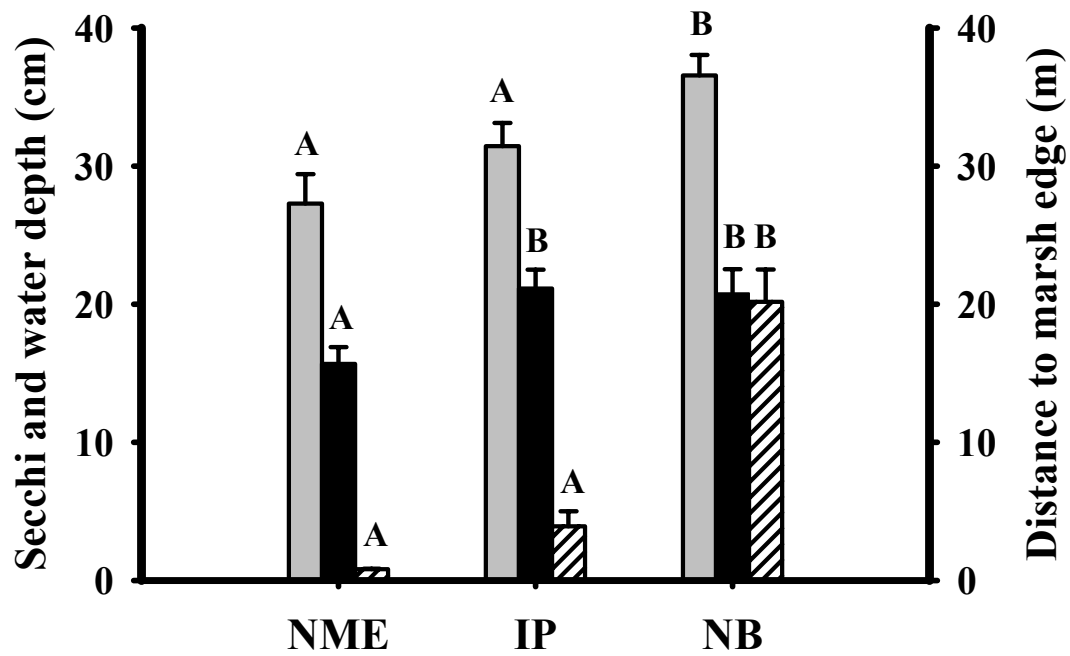


Figure 2.2. Mean water depth (light bars), Secchi depth (dark bars), and distance from marsh edge (hashed bars) by habitat: near-marsh edge (NME), inner-pond SAV (IP), and nonvegetated bottom (NB). Differing letters indicate significant differences ($p < 0.05$). Error bars represent standard errors.

TABLE 2.3. Total number of stems and biomass (g m^{-2}) of emergent vegetation and percent total number of stems and biomass.

Species	No. of stems	Biomass (g m^{-2})	Percent Total	
			Stems	Biomass (g m^{-2})
<i>Aster tenuifolius</i>	439	228.94	1.34	1.66
<i>Cyperus virens</i>	10	5.27	0.03	0.04
<i>Distichlis spicata</i>	3,369	1,703.52	10.28	12.36
<i>Ipomoea sagittata</i>	78	96.26	0.24	0.69
<i>Juncus roemerianus</i>	143	26.18	0.44	0.19
<i>Lythrum lineare</i>	216	203.61	0.66	1.48
<i>Polygonum</i> spp.	4	1.40	0.01	0.01
<i>Sabatia dodecandra</i>	61	13.30	0.18	0.10
<i>Schoenoplectus americanus</i>	6,030	1,165.98	18.40	8.46
<i>Setaria geniculata</i>	95	24.37	0.29	0.18
<i>Spartina patens</i>	21,968	10,161.28	67.03	73.72
<i>Spartina spartinae</i>	6	1.78	0.01	0.01
<i>Vigna luteola</i>	357	151.92	1.09	1.10
Total	32,776	13,783.81	100	100

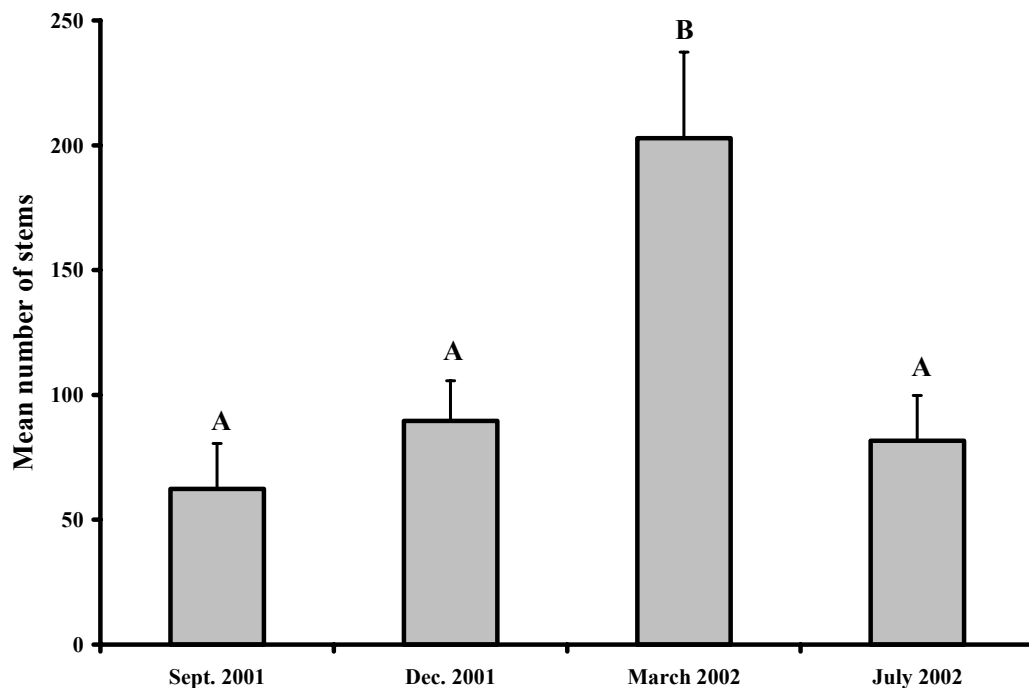


Figure 2.3. Mean number of emergent vegetation stems for study ponds taken from September 2001 to July 2002. Differing letters indicate significant differences ($p = 0.0005$). Error bars represent standard errors.

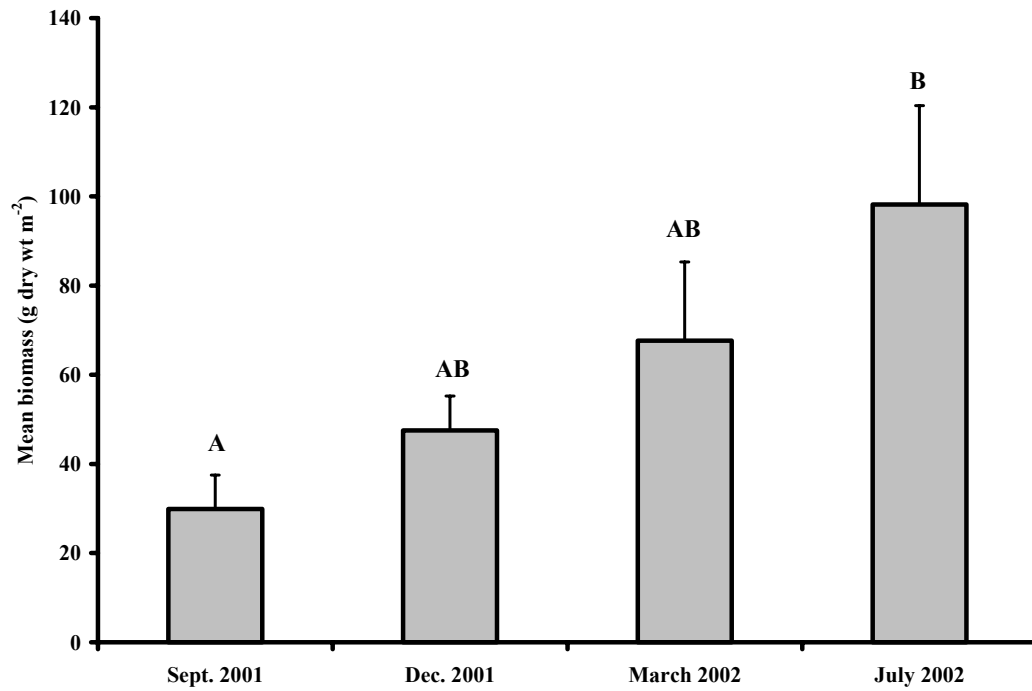


Figure 2.4. Mean emergent vegetation biomass (g dry wt m⁻²) adjacent to study ponds taken from September 2001 to July 2002. Differing letters indicate significant differences ($p = 0.02$). Error bars represent standard errors.

TABLE 2.4. Total nekton abundance (number of individuals) and percent of total catch by habitat type and by month from September 2001 to July 2002.

Species	Total abundance	Density by habitat			Density by month				% of total catch
		Near-marsh SAV	Inner-pond SAV	Non-vegetated bottom	Sept. 2001	Dec. 2001	Mar. 2002	July 2002	
<i>Alosa chrysochloris</i>	16	0	0	16	0	16	0	0	0.31
<i>Anchoa mitchilli</i>	35	2	13	20	12	9	6	8	0.69
<i>Brevoortia patronus</i>	55	11	34	10	0	0	55	0	1.09
<i>Callinectes sapidus</i>	39	20	13	6	10	16	5	8	0.77
Clupeid	2	0	2	0	0	2	0	0	0.03
<i>Cyprinodon variegatus</i>	994	558	432	4	188	16	3	787	19.71
<i>Farfantepenaeus aztecus</i>	3	2	0	1	1	2	0	0	0.05
<i>Fundulus grandis</i>	6	3	3	0	3	0	3	0	0.12
<i>Fundulus pulvereus</i>	217	126	91	0	7	6	0	204	4.30
<i>Gambusia affinis</i>	2	2	0	0	0	0	0	2	0.04
<i>Gobiosoma bosc</i>	22	3	19	0	0	6	1	15	0.43
<i>Leiostomus xanthurus</i>	3	1	0	2	0	0	3	0	0.05
<i>Litopenaeus setiferus</i>	238	88	137	13	217	17	1	3	4.72
<i>Lucania parva</i>	1510	712	797	1	210	78	5	1217	29.90
<i>Membras martinica</i>	2	0	1	1	0	1	0	1	0.03
<i>Menidia beryllina</i>	20	4	16	0	17	2	1	0	0.39
<i>Microgobius gulosus</i>	28	14	12	2	26	1	0	1	0.55
<i>Microphis brachyurus</i>	34	13	21	0	0	0	0	34	0.67
<i>Micropogonias undulatus</i>	26	0	5	21	0	25	1	0	0.71
<i>Mugil cephalus</i>	6	3	1	2	3	1	1	1	0.12
<i>Mugil curema</i>	3	3	0	0	0	0	3	0	0.05
<i>Myrophis punctatus</i>	22	14	6	2	13	1	1	7	0.43
<i>Palaemonetes</i> spp.	861	407	431	23	149	129	77	506	17.07
<i>Poecilia latipinna</i>	858	456	402	0	26	16	0	816	17.02
<i>Sciaenops ocellatus</i>	2	0	0	2	2	0	0	0	0.03
<i>Syngnathus floridae</i>	9	6	2	1	0	4	5	0	0.17
<i>Syngnathus louisianae</i>	2	2	0	0	0	2	0	0	0.03
<i>Syngnathus scovelli</i>	26	15	11	0	3	0	0	23	0.52
Total	5041	2465	2449	127	887	350	171	3633	100

TABLE 2.5. Nekton biomass (g wet wt m⁻²) and percent of total catch from by habitat type and by month from September 2001 through July 2002.

Species	Total biomass (g)	Biomass by Habitat			Biomass by Month				% of total catch
		Near- marsh SAV	Inner- pond SAV	Non- vegetated bottom	Sept. 2001	Dec. 2001	Mar. 2002	July 2002	
<i>Alosa chrysochloris</i>	2.55	0.00	0.00	2.55	0.00	2.55	0.00	0.00	0.19
<i>Anchoa mitchilli</i>	10.69	0.74	3.70	6.25	3.77	2.21	1.10	3.61	0.80
<i>Brevoortia patronus</i>	3.69	0.63	2.38	0.68	0.00	0.00	3.69	0.00	0.28
<i>Callinectes sapidus</i>	341.44	328.26	12.40	0.75	98.61	76.86	1.18	164.79	25.67
Clupeid	0.13	0.00	0.13	0.00	0.00	0.13	0.00	0.00	0.01
<i>Cyprinodon variegatus</i>	203.36	127.04	72.77	3.56	40.89	13.16	3.50	145.81	15.29
<i>Farfantepenaeus aztecus</i>	2.04	0.83	0.00	1.22	0.01	2.03	0.00	0.00	0.15
<i>Fundulus grandis</i>	14.37	11.60	2.77	0.00	1.03	0.00	13.34	0.00	1.08
<i>Fundulus pulvereus</i>	48.48	27.86	20.61	0.00	3.66	4.12	0.00	40.70	3.64
<i>Gambusia affinis</i>	0.37	0.37	0.00	0.00	0.00	0.00	0.00	0.37	0.03
<i>Gobiosoma bosc</i>	5.77	0.82	4.96	0.00	0.00	1.51	0.16	4.10	0.43
<i>Leiostomus xanthurus</i>	0.57	0.03	0.00	0.54	0.00	0.00	0.57	0.00	0.04
<i>Litopenaeus setiferus</i>	91.26	37.57	48.99	4.70	86.86	3.90	0.31	0.19	6.86
<i>Lucania parva</i>	237.82	101.64	136.17	0.01	32.35	18.01	2.25	185.21	17.88
<i>Membras martinica</i>	0.66	0.00	0.41	0.25	0.00	0.25	0.00	0.41	0.06
<i>Menidia beryllina</i>	5.47	1.64	3.82	0.00	4.37	0.61	0.49	0.00	0.41
<i>Microgobius gulosus</i>	7.78	4.27	3.18	0.33	6.16	0.33	0.00	1.29	0.58
<i>Microphis brachyurus</i>	7.44	3.16	4.28	0.00	0.00	0.00	0.00	7.44	0.56
<i>Micropogonias undulatus</i>	6.14	0.00	0.56	5.58	0.00	6.09	0.05	0.00	0.46
<i>Mugil cephalus</i>	92.68	30.80	34.63	27.24	30.80	34.63	20.92	6.33	6.97
<i>Mugil curema</i>	0.69	0.69	0.00	0.00	0.00	0.00	0.69	0.00	0.05
<i>Myrophis punctatus</i>	13.06	7.90	2.62	2.55	8.36	1.54	1.01	2.15	0.98
<i>Palaemonetes</i> spp.	125.82	56.09	65.09	4.64	21.98	13.75	13.99	76.10	9.46
<i>Poecilia latipinna</i>	96.01	54.22	41.79	0.00	7.18	5.56	0.00	83.27	7.22
<i>Sciaenops ocellatus</i>	0.11	0.00	0.00	0.11	0.11	0.00	0.00	0.00	0.01
<i>Syngnathus floridae</i>	1.71	0.92	0.69	0.10	0.00	0.90	0.81	0.00	0.13
<i>Syngnathus louisianae</i>	0.13	0.13	0.00	0.00	0.00	0.13	0.00	0.00	0.02
<i>Syngnathus scovelli</i>	9.86	4.89	4.97	0.00	0.41	0.00	0.00	9.45	0.74
Total	1330.10	802.10	466.92	61.06	346.55	188.27	64.06	731.21	100.00

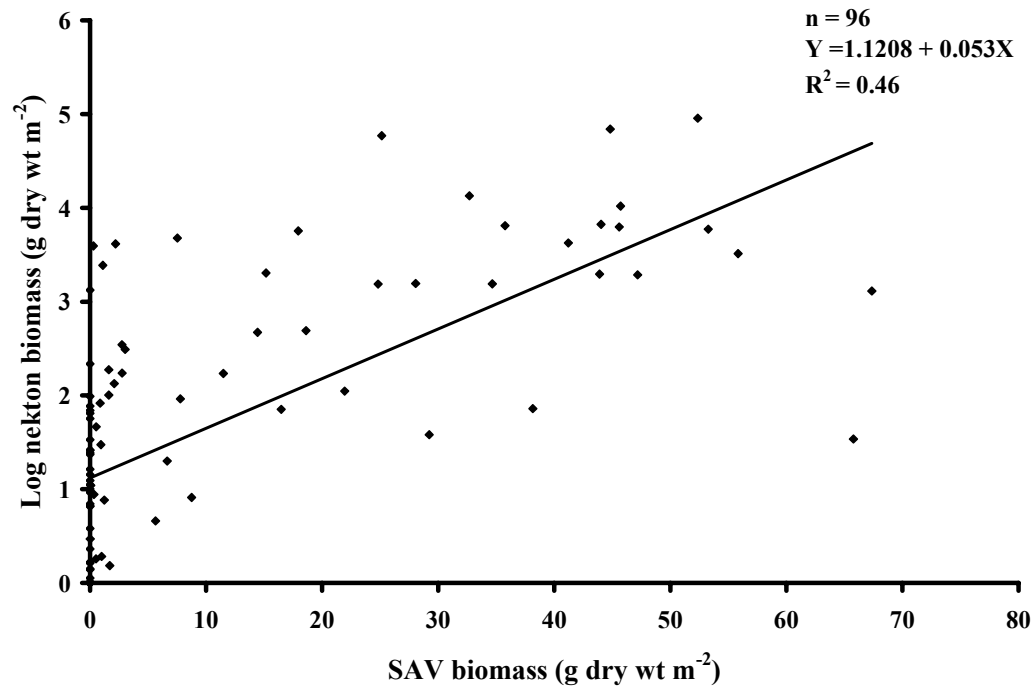


Figure 2.5. Relationship between nektun biomass (g wet wt m⁻²) (log (x +1) transformed) and SAV biomass (g dry wt m⁻²). Regression line indicated by solid line.

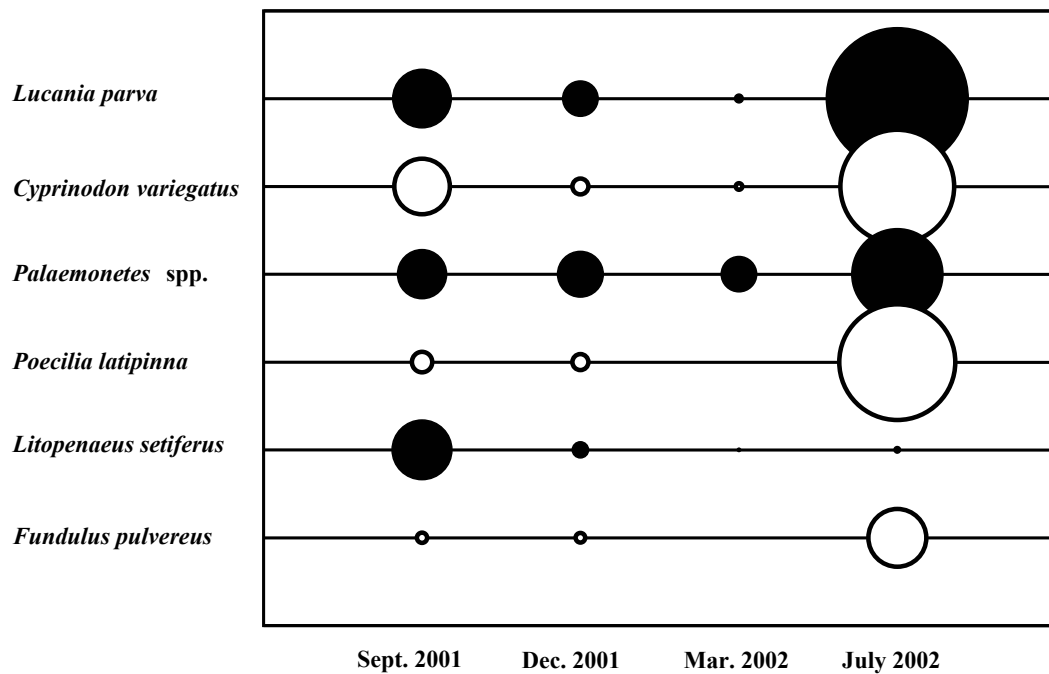


Figure 2.6. Abundance of numerically dominant nektun taxa from September 2001 to July 2002. Circle size is proportional to relative taxa abundance.

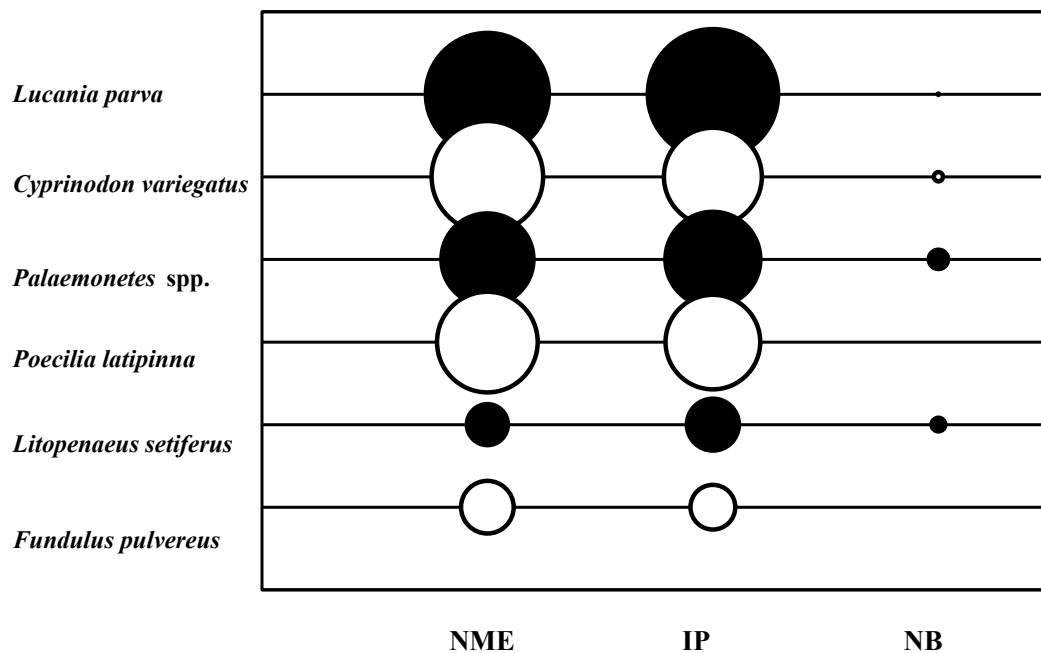


Figure 2.7. Abundance of numerically dominant nekton taxa by habitat: near-marsh edge (NME), inner-pond SAV (IP), and nonvegetated bottom (NB). Circle size is proportional to relative abundance of nekton taxa.

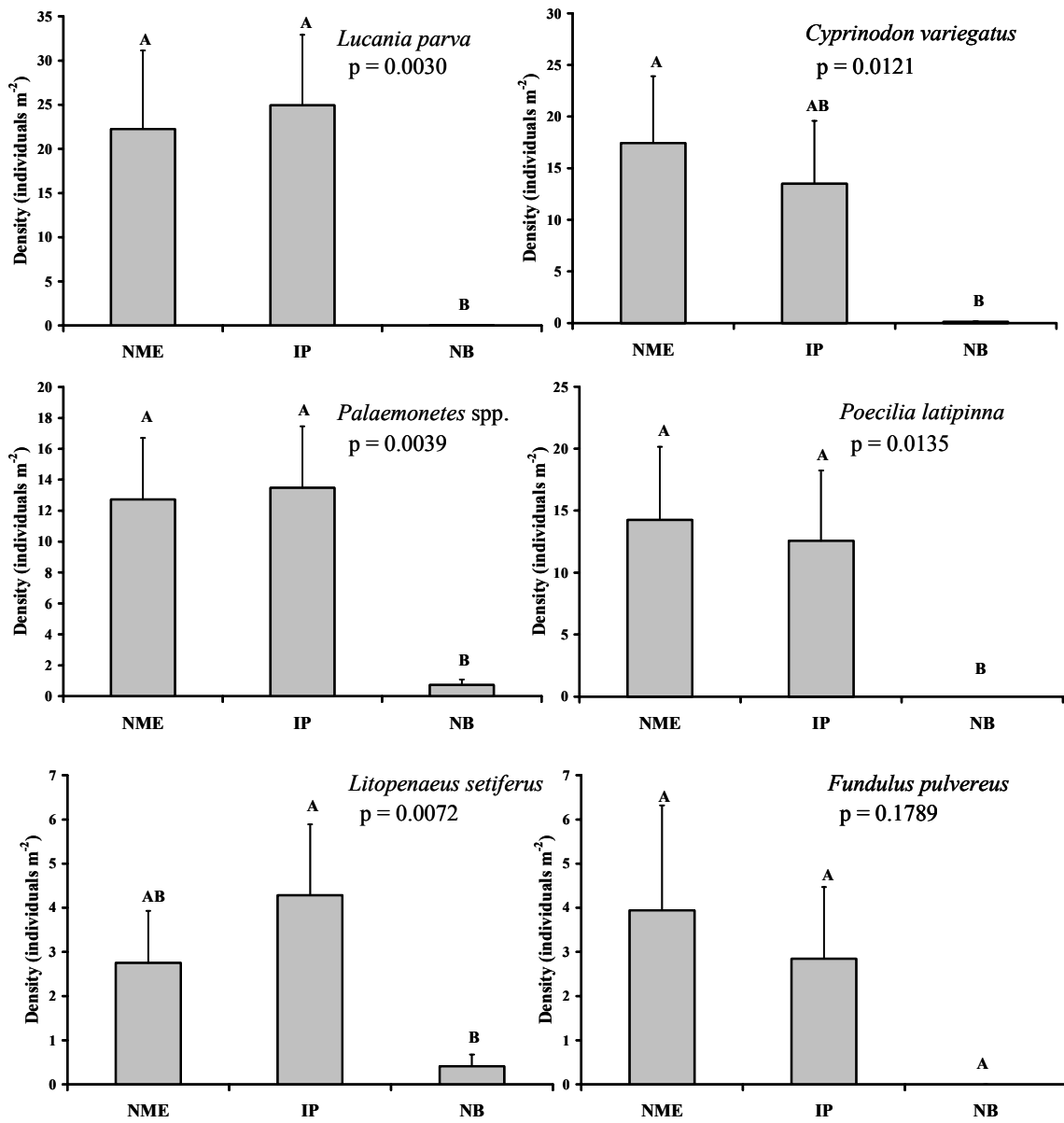


Figure 2.8. Density (individuals m^{-2}) of numerically dominant nekton taxa by habitat: near-marsh edge (NME), inner-pond SAV (IP), and nonvegetated bottom (NB). Differing letters indicate significant differences. Error bars represent standard errors.

dry wt m⁻²) were similar and significantly greater than in nonvegetated habitat (NB= 0.00 - 0.21 g dry wt m⁻²).

Mean SAV biomass differed significantly by month ($p < 0.001$). In July 2002, mean SAV biomass was significantly greater (24.20 ± 4.79 g dry wt m⁻²) than all other sampling dates. However, in September 2001 mean SAV biomass (10.51 ± 2.77 g dry wt m⁻²) was similar to December 2001 (8.31 ± 3.43 g dry wt m⁻²) and greater than March 2002 SAV biomass (0.004 ± 0.003 g dry wt m⁻²) (Figure 2.19).

Mean SAV biomass differed significantly among habitats with nonvegetated bottom having significantly less mean biomass (0.05 ± 0.05 g dry wt m⁻²) than near-marsh edge (13.33 ± 3.52 g dry wt m⁻²) and inner-pond SAV (18.88 ± 3.53 g dry wt m⁻²) habitats ($p < 0.0001$) (Figure 2.20).

Pond Effect

Significant differences among ponds occurred for SAV biomass ($p = 0.0076$). Differences in SAV biomass among ponds can not be explained by variation in pond size, number of tidal channels, or amount of emergent marsh edge.

Discussion

The data presented in this study did not support the null hypotheses 1) nekton community composition was unrelated to habitat type and habitat characteristics, and 2) nekton density and abundance were unrelated to habitat type and habitat characteristics; therefore, they were rejected. Vegetated habitat types supported more diverse nekton communities, with increased nekton biomass, densities, and abundance than nonvegetated bottom habitat.

Regardless of the presence or absence of submerged aquatic vegetation, several environmental variables (e.g., water temperature, dissolved oxygen, and salinity) were similar

among habitat types. Water depth, distance from marsh edge, and Secchi depth were the only environmental variables that differed between vegetated habitat types and nonvegetated bottom. These differences were likely due to SAV bed location within the study ponds. Submerged aquatic vegetation beds typically fringed marsh ponds and rarely extended greater than 5 m from the shoreline therefore, NME (< 1 m from emergent marsh edge) and IP samples (> 1 m from emergent marsh edge) were generally both less than 5m from the marsh edge. Typical of marsh ponds, water depth increased with increasing distance from emergent marsh edge, but water depth did not differ significantly within 5m of the marsh edge. With both vegetated habitats typically less than 5m from the emergent marsh edge, distance from the shoreline apparently did not influence nekton distribution within SAV beds. If SAV beds had extended farther into the marsh ponds, the influence of distance from marsh edge may have been a factor influencing nekton distribution within SAV.

The mere presence of SAV (i.e. SAV biomass as a proxy for cover) and not location of the SAV appeared to be driving nekton distribution within the small study ponds (< 1.5 ha). Submerged aquatic vegetation biomass was similar between vegetated habitats and thus, vegetated habitats supported similar nekton biomass. Vegetated habitats were also similarly located, providing nekton equal access to them and resulting in similar nekton assemblages in both vegetated habitats. Therefore, both vegetated habitats, regardless of their location within the pond, were potentially providing similar nekton refuge.

Typically in estuarine systems, nekton catches are greatest in the spring and fall because increased numbers of transient nekton species (Czapla et al. 1991), but in this study, transient nekton species contributed little to nekton densities and biomass. Resident marsh species dominated all catches throughout the study (e.g., sheepshead minnow *Cyprinodon variegatus*,

rainwater killifish *Lucania parva*, and sailfin molly *Poecilia latipinna*). Even though catches in my study were composed largely of resident species, seasonal influences were evident in nekton density and biomass. The greatest densities and biomass of nekton were collected in July 2002, which coincided with a peak in SAV biomass. Increased SAV biomass in July likely provided the necessary habitat heterogeneity for nekton to use as refuge.

Nekton densities in the vegetated habitat types (~ 78 individuals m^{-2}) in this study were greater than other similar studies. Sogard et al. (1989) reported high fish densities (47.68 individuals m^{-2}) from Florida Bay, and Minello (1999) reported a high mean crustacean density (50.22 individuals m^{-2}) from Texas and Louisiana. In this study, fish densities in the vegetated habitats were approximately 60 individuals m^{-2} and crustaceans were approximately 17 individuals m^{-2} , both greater than similar studies in Louisiana. Duffy and Baltz (1998) reported lower fish densities in vegetated habitats (12.45 individuals m^{-2}) from Lake Pontchartrain and Castellanos and Rozas (2001) reported 30.5 fish m^{-2} and 13.2 crustaceans m^{-2} from the Atchafalaya River Delta.

Greater nekton densities in this study may be explained by differences in marsh system structure. This study was conducted in small marsh ponds where water exchange is restricted to small tidal channels when the emergent marsh is not flooded. The system is composed of a complex matrix of channels and ponds connected to major bayous, which are the main sources of tidal exchange. Resident nekton may be partially restricted (and therefore concentrated) to marsh ponds because of the great distance through the network of channels to the bayou. The resident nekton species associated with these ponds are capable of tolerating a wide range of environmental conditions (e.g., dissolved oxygen) and therefore are not obligated to move out of ponds when environmental conditions are unfavorable. The observed differences in this and

other studies may be due to the fact that previous work was conducted in lacustral and insular environments that would not inhibit fish movement to the associated larger tidal sources. Open environments, compared to ponds, may facilitate fish movement away from unfavorable environmental conditions and would result in lower nekton densities. Sampling in other studies often occurred at high tide when the emergent marsh was flooded and nekton had access to the emergent marsh, potentially decreasing nekton densities. Sampling in this study occurred when the emergent marsh was not flooded and nekton was restricted to the marsh ponds possibly increasing nekton densities.

The SAV biomass collected in this study was similar to other studies (Sheridan 1992, Raposa and Oviatt 2000, Castellanos and Rozas 2001, Hovel 2003). Highest SAV standing crop occurred in July which is consistent with other studies in which standing crop peaked in the summer (Raposa and Oviatt 2000, Castellanos and Rozas 2001).

This study only investigated nekton utilization within *Ruppia maritima* beds, however at least three SAV species exist in the brackish marsh ponds at MIWR: water-milfoil *Myriophyllum spicatum* L., wild celery *Vallisneria americana* Michx., and widgeon grass *R. maritima* L. (Hunter 2000). The structure differs considerably among these species, thereby providing various levels of structural complexity (shoot density and shoot biomass) depending on the SAV species. *Myriophyllum spicatum* is a rooted SAV with leaves whorled around branched stems. Leaves are composed of numerous filiform segments, usually 24, which form dense, submerged colonies (Stutzenbaker 1999). *Vallisneria americana* is a rooted species with long, thin, ribbon-like leaves (13 to 19 mm wide) and tends to form dense stands that seldom contain other SAV species (Stutzenbaker 1999). *Ruppia maritima* is a rooted species with branched stems containing thread-like leaves typically 0.5 to 1 mm wide (Stutzenbaker 1999). Of the three SAV

species found at Marsh Island, *M. spicatum* is the most structurally complex. If nekton are attracted to SAV beds because of the added protection of structurally complex SAV, then nekton abundance and density may differ among SAV beds of different species at Marsh Island. Duffy and Baltz (1998) studied fish assemblages within beds of these SAV species within Lake Pontchartrain, Louisiana. They found the most common fishes to be generally more abundant and dense in *M. spicatum*, as compared to *R. maritima* and *V. americana*. However, Castellanos and Rozas (2001) reported the presence of vegetation to be more important in influencing nekton densities than the species or morphology.

SAV as Essential Fish Habitat

This study provided information on two levels needed to identify EFH; 1) species presence or absence and abundance within SAV and nonvegetated bottom (level 1), 2) habitat associations (level 2). Increased nekton use of SAV habitats, as described in the study, reinforces the need to conserve, protect, and enhance (potentially through restoration) SAV habitats. Submerged aquatic vegetation provided essential fish habitat for both transient and resident marsh species. Essential fish habitat research typically focuses on estuarine dependent nekton species, but it is also important to recognize the role SAV plays in supporting resident species because they are often prey for transient nekton species. Without SAV habitats, resident marsh species abundances would likely decline, thereby negatively impacting transient species growth and survival.

Influences of Wetland Loss on SAV Habitat

Many SAV habitats are being converted into nonvegetated habitats by natural (e.g., subsidence) and anthropogenic (e.g., diminishing water quality and increasing turbidity) processes. With wetland loss rates exceeding $50 \text{ km}^2 \text{ yr}^{-1}$ (Britsch and Dunbar 1993), marsh in

Louisiana is converting into open water faster than at any other location in the United States. Initial stages of this land loss in Louisiana are hypothesized to be beneficial for secondary production by increasing the amount of highly productive emergent marsh edge (Chesney et al. 2000, Minello and Rozas 2002). Intermediate stages of coastal land loss maximize emergent marsh edge and increase shallow open water habitats, which are potential sites for SAV bed establishment. Initial increases in fisheries productions with emergent marsh loss, due to increased marsh edge, may be supplemented by increased secondary production in the newly created SAV habitats. So, intermediate stages of coastal land loss should also be beneficial for secondary production. As with the predictions for eventual decreases in the amount of marsh edge through time (Browder et al. 1985), shallow open water areas will evolve into large, deep embayments which may not continue to support SAV due to light attenuation, increased fetch, and wave energy. In this study, nekton was associated with SAV more than nonvegetated habitat, so continued loss of SAV habitat, by any process, will likely negatively impact nekton production.

Minimum SAV Biomass Threshold

In this study, SAV biomass can be used as a proxy for vegetative structural complexity based on the assumption that increased SAV biomass represents increased percent cover and structure. There may be a minimum threshold of SAV biomass that provides enough structural complexity to support increased nekton biomass. When vegetated habitats meet the minimum threshold requirements, nekton biomass and nekton diversity may be positively influenced. When little or no SAV is present (e.g., March 2002) the minimum threshold of SAV biomass may not be reached causing previously vegetated habitats to function as nonvegetated bottom. In March 2002, minimal SAV biomass in all habitats types was reflected in low nekton biomass in

all habitat types. The presence and abundance of transient nekton species within an estuary fluctuates throughout the year. If the catch was dominated by transient nekton species, seasonal fluctuations in transient species could explain the decreased nekton biomass in March. However, with the majority of nekton biomass contributed by resident nekton species, low nekton biomass in March was attributed to the lack of SAV biomass. The individual influences of season and SAV biomass were difficult to determine in July, however increased densities are likely due to a combination of both spawning and increased SAV biomass.

Research Implications and Future Research Needs

Future research should investigate the influence of SAV bed location and the potential effects of numerous SAV species on nekton use. To investigate the influence SAV bed location on nekton utilization, vegetated habitat sampling should be separated from each other by greater distances. Results from previous studies suggest that all habitat features (e.g., patch area, shoot density, and blade density) should be considered in future studies and in any seagrass restoration project. It appears that similar SAV biomass between vegetated habitats within small ponds, provided equivalent nekton habitat. However, Hovel (2003) investigated the influences of SAV cover, configuration, and structural complexity (shoot density and shoot biomass) on juvenile blue crab survival over a large spatial scale and found increased SAV complexity may not have led to increased crab survival. Crab survival was influenced by landscape structure and geographic locations of SAV habitats because differences in patch area, shoot density, and shoot biomass produced both positive and negative effects on crab survival depending on geographic location.

This study did not account for the influence of vegetative structural complexity among numerous SAV species because only widgeon grass *Ruppia maritima* was studied, but

differences among SAV species is an important concept. Stoner (1983) found that among various SAV species, blade density was a better predictor of fish abundance than SAV biomass. Different SAV species may provide various degrees of vegetative structure thereby contributing to the structural complexity (e.g. numerous small leaves on one stem, single leaves per stem, whorled leaves). Submerged aquatic vegetation species that are morphologically complex may provide better refuge and therefore support higher nekton densities than species that are morphologically simple.

Nekton survival also may be influenced by SAV structural complexity. Hovel (2003) found survival of blue crabs *Callinectes sapidus* to be positively correlated with seagrass shoot density and negatively correlated with seagrass shoot biomass indicating that increasing structural complexity does not necessarily lead to increased nekton survival. Other studies have shown densities of newly settled Sciaenids differed among SAV species, and suggest that SAV canopy structure may influence settlement patterns and postsettlement survival (Rooker et al. 1998). Therefore, SAV species and the structural complexity associated with particular species can influence nekton densities, abundances, and survival. It is important as ecologists develop criteria for marsh management and develop restoration projects that a variety of SAV habitats are managed for structural complexity and species diversity to support increased nekton diversity.

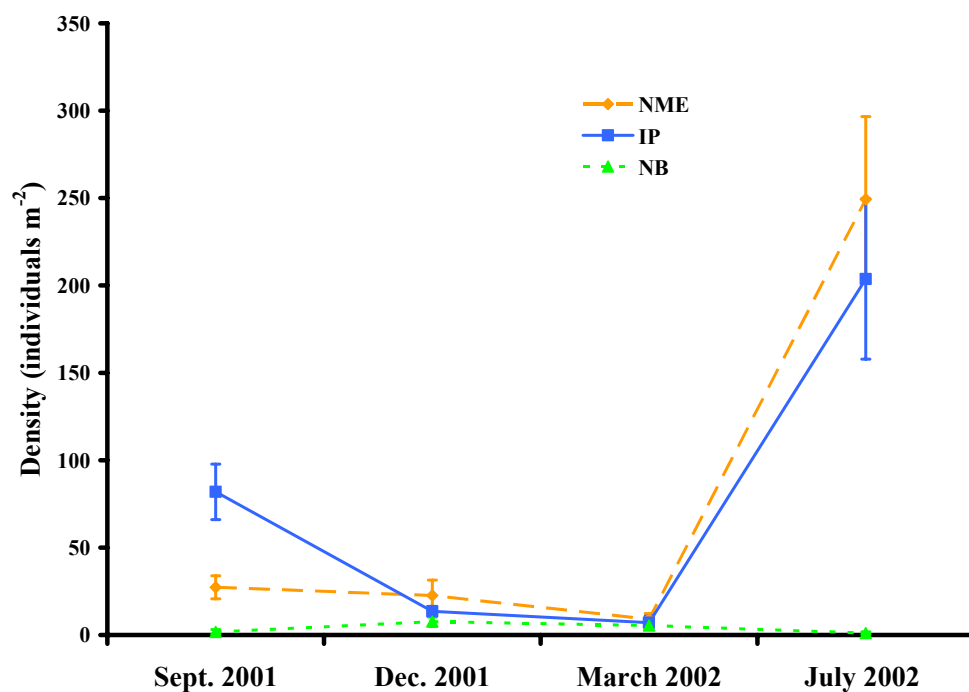


Figure 2.9. Mean nekton density (individuals m⁻²) by habitat: near-marsh edge (NME), inner-pond SAV (IP), and nonvegetated bottom (NB) for September 2001 through July 2002. Error bars represent standard errors.

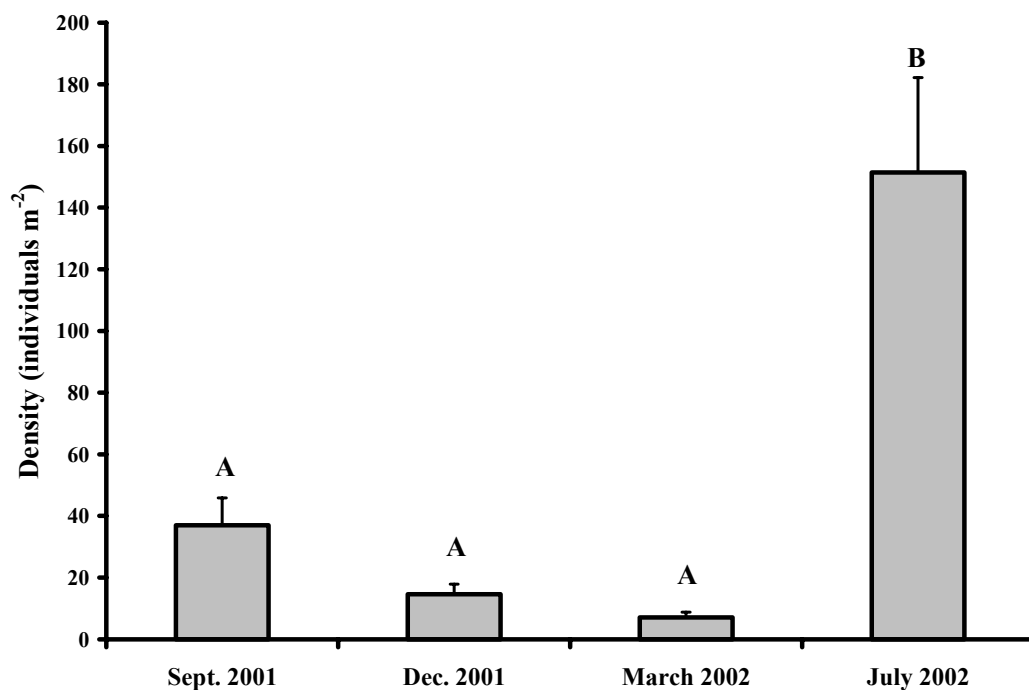


Figure 2.10. Mean nekton density (individuals m⁻²) for all taxa from September 2001 to July 2002. Differing letters indicate significant differences (p < 0.0001). Error bars represent standard errors.

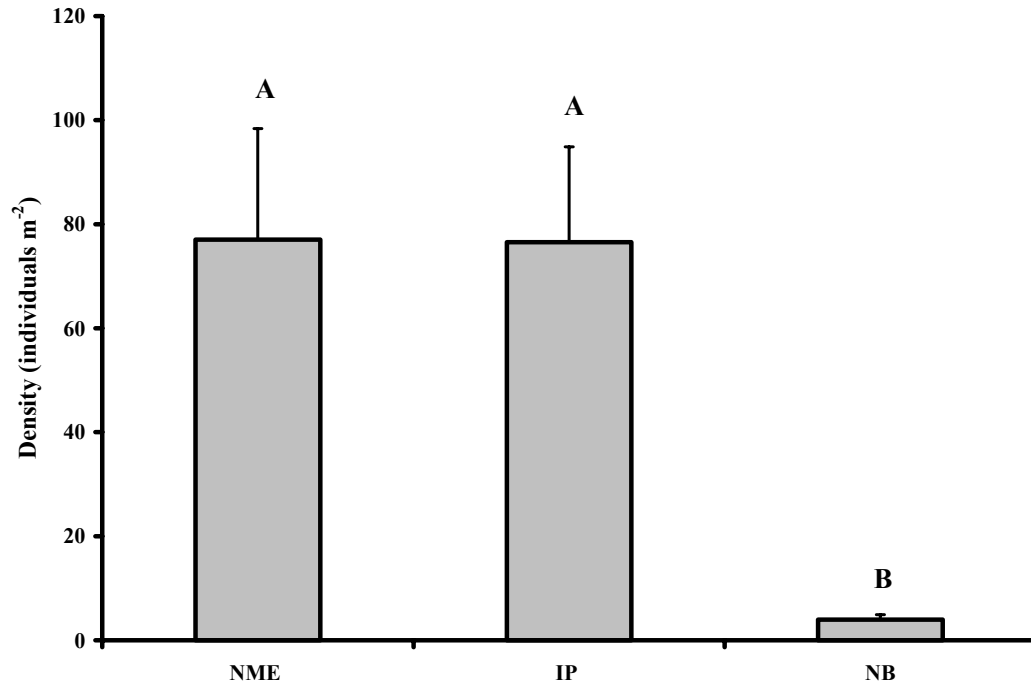


Figure 2.11. Mean nekton density (individuals m⁻²) by habitat: near-marsh edge (NME), inner-pond SAV (IP), and nonvegetated bottom (NB). Differing letters indicate significant differences ($p < 0.0001$). Error bars represent standard errors.

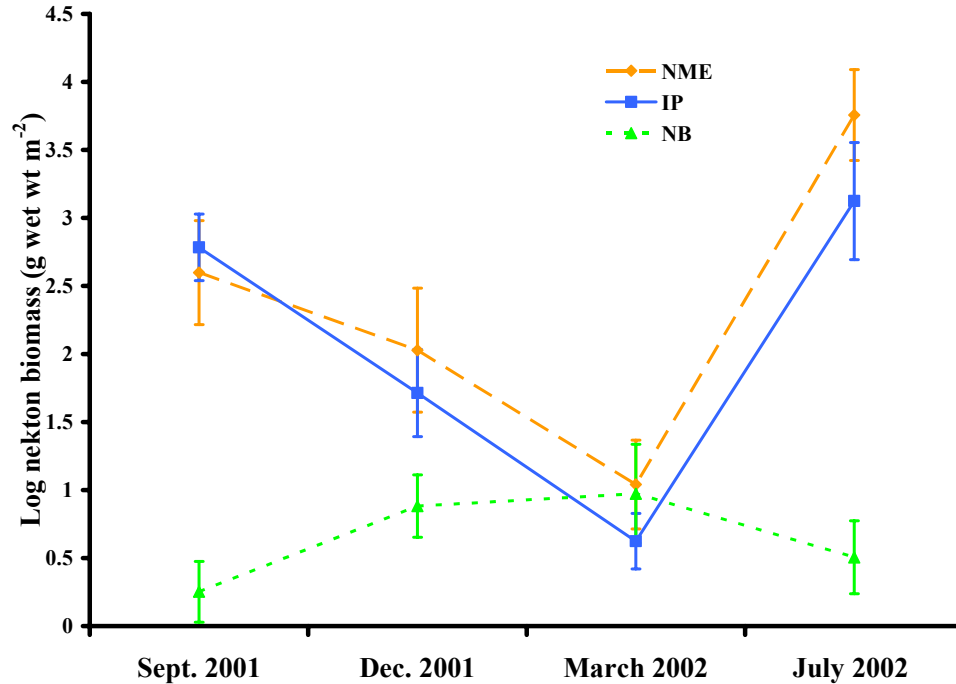


Figure 2.12. Mean nekton biomass (g wet wt m⁻²) log (x + 1) transformed by habitat: near-marsh edge (NME), inner-pond SAV (IP), and nonvegetated bottom (NB) for September 2001 through July 2002. Error bars represent standard errors.

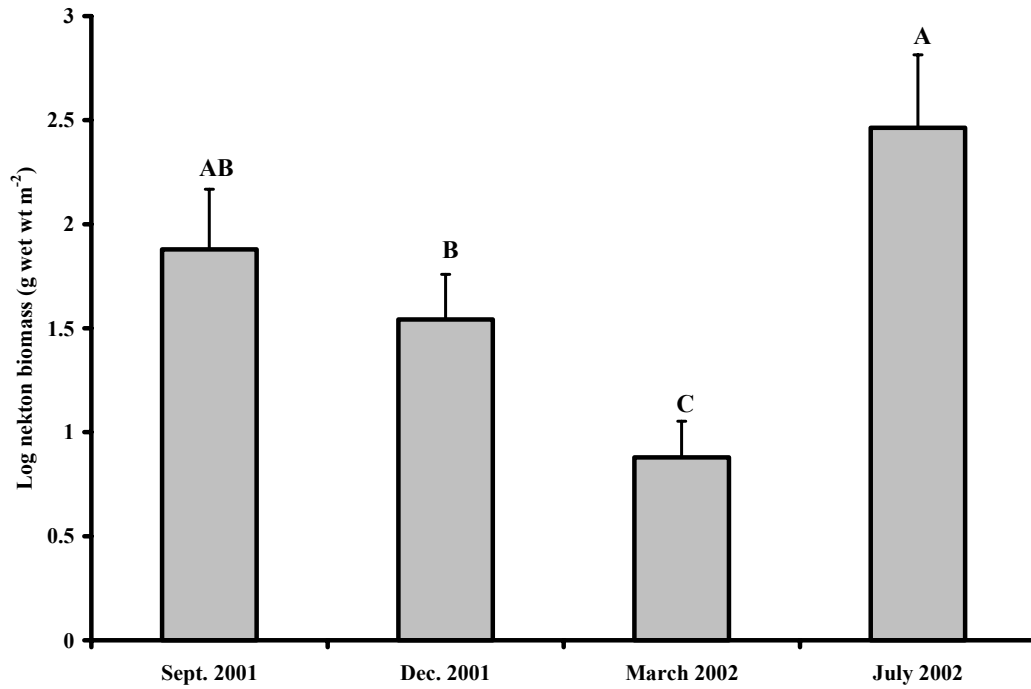


Figure 2.13. Mean nekton biomass (g wet wt m⁻²) log (x + 1) transformed from September 2001 to July 2002. Differing letters indicate significant differences ($p < 0.0001$). Error bars represent standard errors.

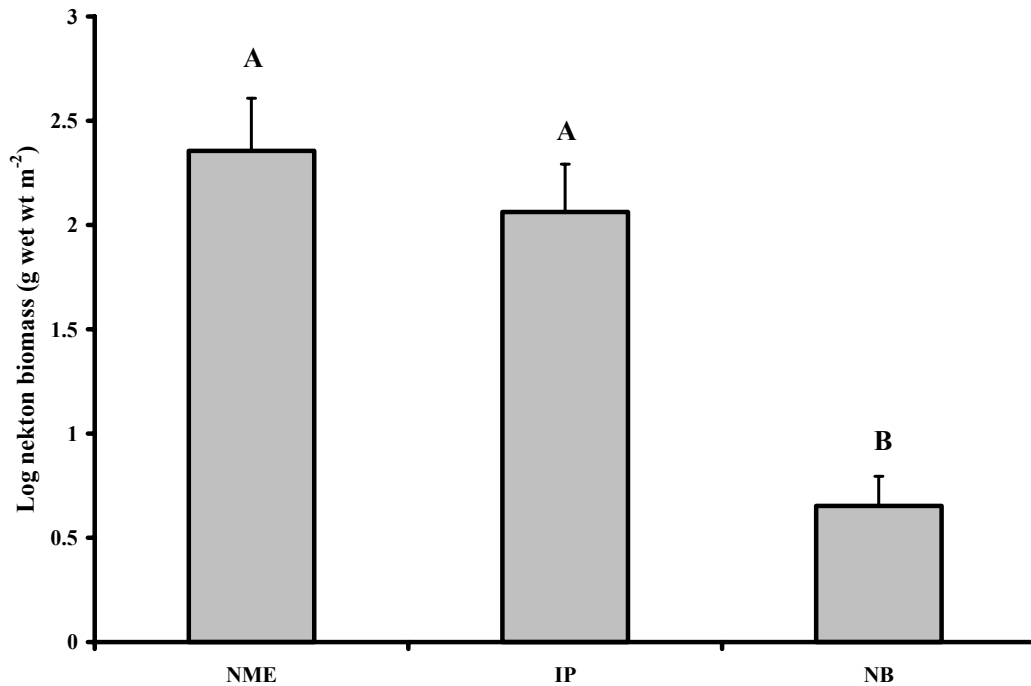


Figure 2.14. Mean nekton biomass (g dry wt m⁻²) log (x + 1) transformed by habitat: near-marsh edge (NME), inner-pond SAV (IP), and nonvegetated bottom (NB). Differing letters indicate significant differences ($p < 0.0001$). Error bars represent standard errors.

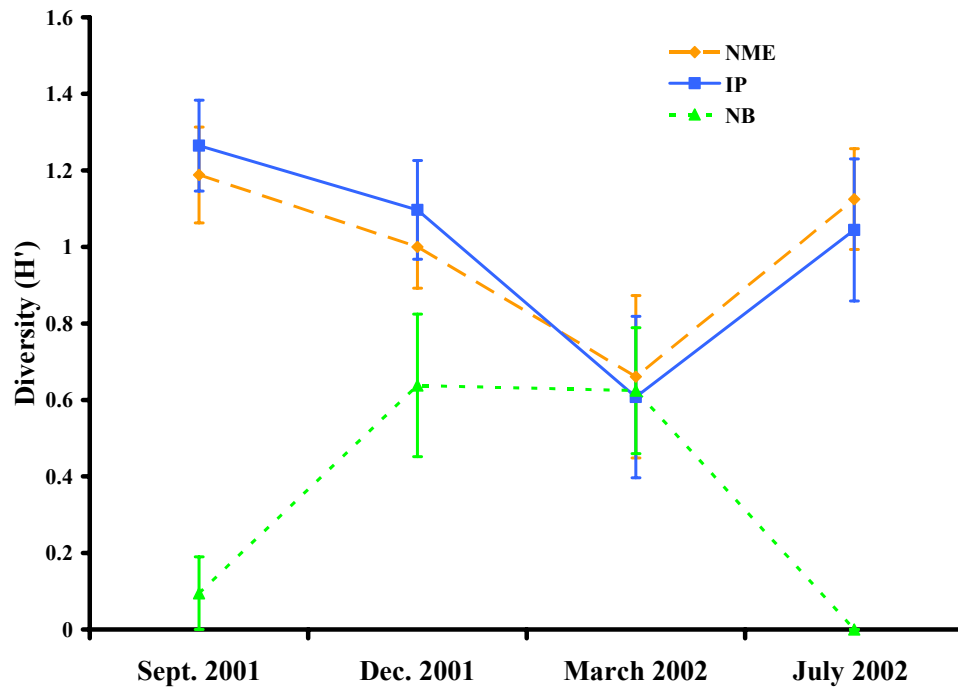


Figure 2.15. Mean Shannon Wiener diversity (H') by habitat: near-marsh edge (NME), inner-pond SAV (IP), and nonvegetated bottom (NB) for September 2001 through July 2002. Error bars represent standard errors.

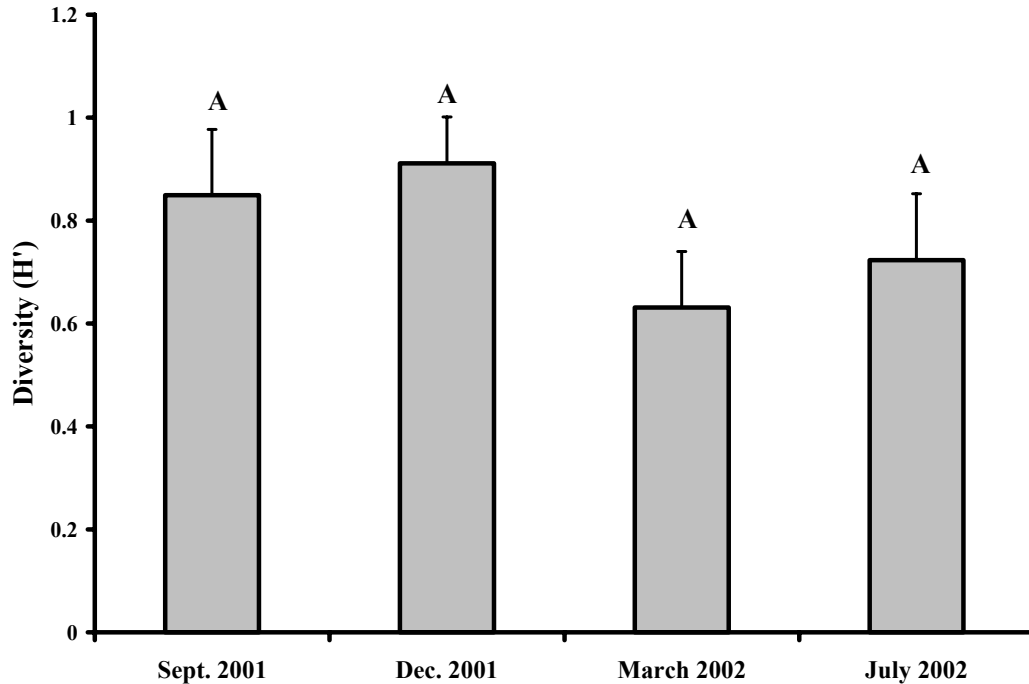


Figure 2.16. Mean Shannon Wiener diversity (H') from September 2001 to July 2002. Similar letters indicate no significant difference ($p = 0.0603$). Error bars represent standard errors.

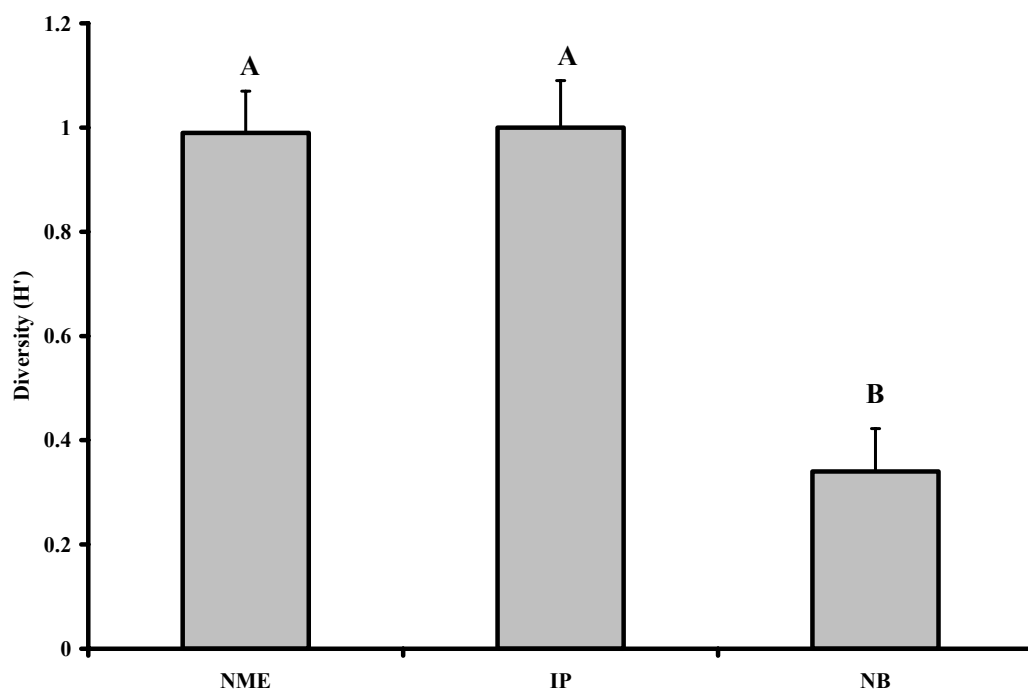


Figure 2.17. Mean Shannon Wiener diversity (H') by habitat: near-marsh edge (NME), inner-pond SAV (IP), and nonvegetated bottom (NB). Differing letters indicate significant differences ($p < 0.0001$). Error bars represent standard errors.

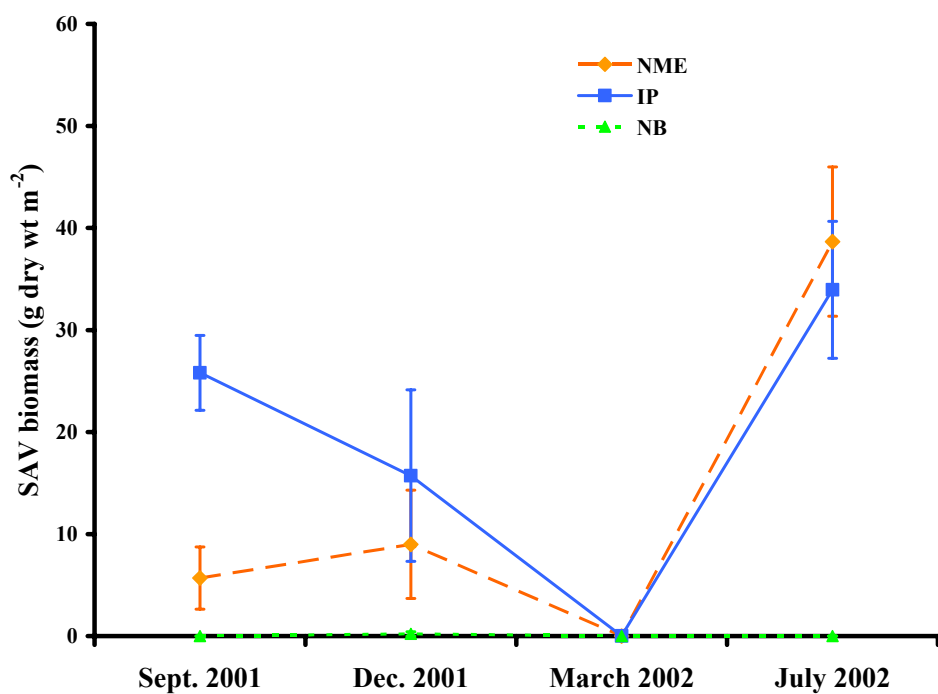


Figure 2.18. Mean submerged aquatic vegetation biomass (g dry wt m^{-2}) by habitat: near-marsh edge (NME), inner-pond SAV (IP), and nonvegetated bottom (NB) for September 2001 through July 2002. Error bars represent standard errors.

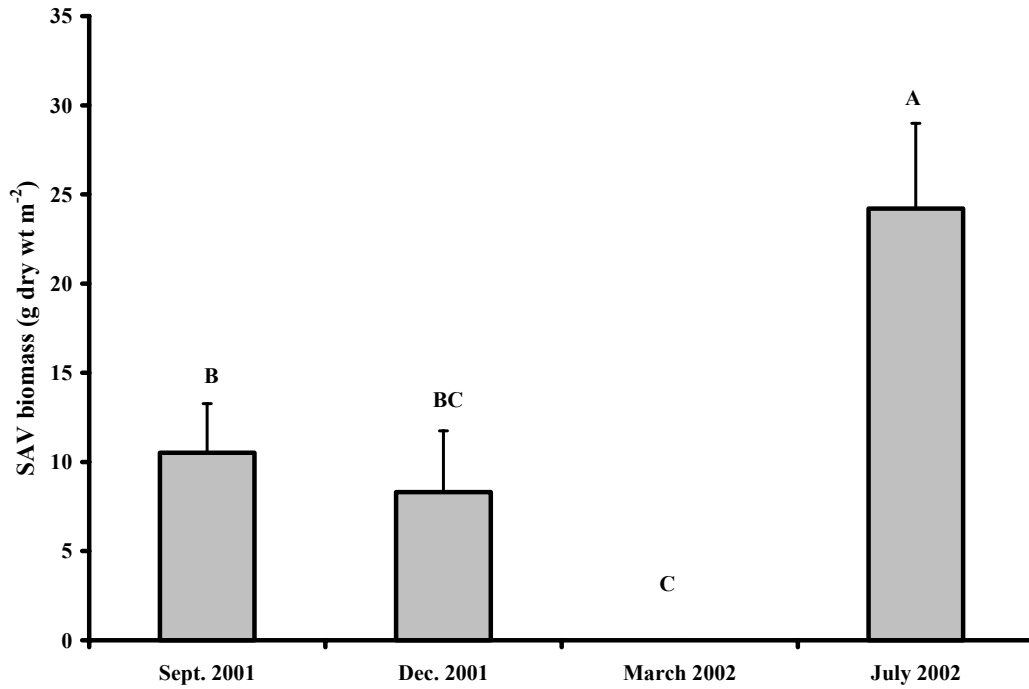


Figure 2.19. Mean submerged aquatic vegetation biomass (g dry wt m⁻²) for study ponds taken from September 2001 to July 2002. Differing letters indicate significant differences ($p < 0.0001$). Error bars represent standard errors.

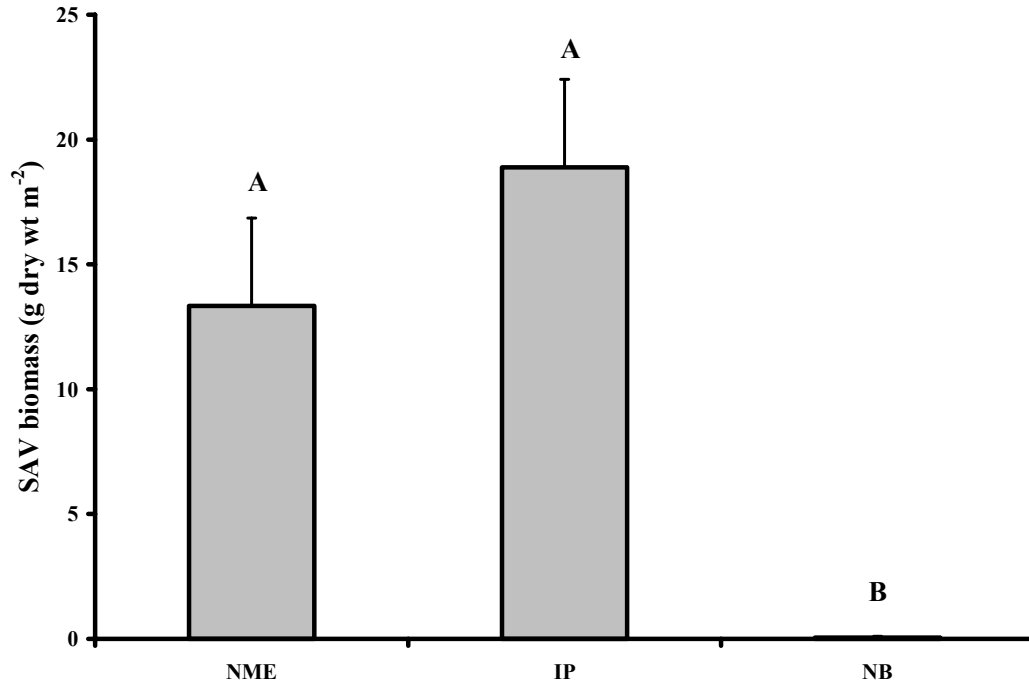


Figure 2.20. Mean submerged aquatic vegetation biomass (g dry wt m⁻²) by habitat: near-marsh edge (NME), inner-pond SAV (IP), and nonvegetated bottom (NB). Differing letters indicate significant differences ($p < 0.0001$). Error bars represent standard errors.

CHAPTER III.

GROWTH OF ATLANTIC CROAKER (*MICROPOGONIAS UNDULATUS*) IN VEGETATED AND NONVEGETATED MICROHABITAT TYPES

Introduction

Nekton densities, abundances, and diversity are often greater in submerged aquatic vegetation (SAV) than over nonvegetated bottom (Arrivillaga and Baltz 1999, Minello et al. 2003), and this pattern is thought to be related to the refuge and high quality growth environment provided by SAV. Because vegetated habitats support greater infaunal communities (benthic and epiphytic algae, detritus, and infaunal organisms) than nonvegetated habitats (Phillips and McRoy 1980) food is more available in vegetated habitats and therefore, hypothesized to be a high quality growth environment.

Increased food supply within SAV beds may provide a growth advantage for nekton (Orth et al. 1984, Irlandi and Crawford 1997). Faster growth is advantageous because it can increase nekton survival by producing larger individuals in a shorter time period. Large individuals are able to move faster (Webb and Corolla 1981) and farther to avoid predators, thus increasing survival. Growth advantages also enable individuals to be larger at the end of a growing season, thus enhancing their chances of surviving because predators may disproportionately affect small nekton (Levin et al. 1997).

Several studies have compared growth rates within different habitat types, but no *in situ* growth experiments have been conducted in Louisiana estuaries. Sogard (1992) found that in a New Jersey estuary fish growth rates were significantly greater in vegetated habitats for only one of the three species in the experiment, tautog *Tautoga onitis*. Stunz et al. (1999, 2002a) reported significantly higher growth rates of young red drum *Sciaenops ocellatus* in seagrass beds and saltmarsh vegetation than in oyster reef and nonvegetated bottom habitats of Galveston Bay. In

laboratory and field experiments conducted in the lower York River, Virginia, Perkins-Visser et al. (1996) showed that blue crabs *Callinectes sapidus* grew faster in SAV than over nonvegetated bottom. However, Phelan et al. (2000) did not show consistent differences in growth of winter flounder *Pseudopleuronectes americanus* and tautog *Tautoga onitis* in New Jersey and Connecticut between vegetated and nonvegetated habitats. Nadeau (1991) found no significant difference in growth rates of red drum *Sciaenops ocellatus* between vegetated and nonvegetated habitats.

If vegetated habitats provide better quality or quantity of food for nekton, then nekton growth rates in vegetated habitats should be greater than in nonvegetated habitats. To address this question, we tested the null hypothesis that nekton growth was similar between vegetated and nonvegetated habitat types.

Methods

Study Area

This study was conducted within Rockefeller Wildlife Refuge (RWR), located in Cameron and Vermilion parishes, Louisiana. The Refuge consists of 30,772 ha of marshlands bordered on the north by the Grand Chenier Ridge complex and to the south by the Gulf of Mexico. Marsh vegetation varies throughout the refuge from bulrush *Schoenoplectus californicus* dominated fresh marsh in the north, to wiregrass *Spartina patens* dominated brackish marsh, to oystergrass *Spartina alterniflora* dominated saline marsh near the Gulf of Mexico. The Louisiana Department of Wildlife and Fisheries (LDWF) actively manages RWR to stabilize isohaline lines, limit saline encroachment, reverse marsh deterioration, and provide productive wildlife habitats (Perry 2000). The study was conducted within management Unit 5 because sufficient quantities of SAV could not be located in the unmanaged areas of the refuge.

Management Unit 5 is a 1,983 ha impounded marsh and is managed by gravity drainage systems to control water and salinity for the propagation of wildlife food plants (Perry 2000). Emergent marsh vegetation in this unit is dominated by wiregrass *Spartina patens* and widgeon grass *Ruppia maritima* is the dominant SAV species.

Study Design

Four ponds, within Unit 5, that supported monotypic stands of widgeon grass *Ruppia maritima* were selected haphazardly to use in the study (Figure 3.1). Study ponds supported monotypic stands of *Ruppia maritima*. Within each pond, four enclosures were created; two vegetated and two nonvegetated (n = 16). Vegetated enclosures were placed randomly within SAV beds. Nonvegetated enclosures were placed randomly over nonvegetated bottom. Five fish were placed into each of the 16 enclosures (n = 80). To record environmental variables, four Hydrolab® Data Sonde continuous recorders were located centrally between enclosures, one per study pond.

Enclosure Design

Enclosures (area = 0.34 m²) were created from 55 gallon plastic drums (Figure 3.3). The bottoms and the tops of the plastic drums were removed, and windows approximately 35 cm by 51 cm and were placed at least 15 cm from the bottom of the drum were created on two sides to allow water exchange. Drums were placed 15 cm into the substrate to prevent burrowing organisms from entering the enclosures. Windows were covered with 3 mm plastic mesh, inside the enclosure, to prevent experimental organisms from escaping. On the exterior of the enclosure, 6.4 mm aluminum mesh was used to prevent predators from creating holes in the plastic netting and entering the enclosures.

Nekton was cleared from the enclosure prior to stocking with experimental organisms by placing a 1 m² aluminum sided throw trap around the enclosure and removing the water with a

sump pump (Figure 3.3). The pump intake hose was placed between the throw trap and the enclosure walls to remove water without disturbing the SAV within vegetated enclosures. After an enclosure was drained, nekton was removed with a 3 mm mesh dip net.

Atlantic croaker *Micropogonias undulatus* were collected with a cast net at water control structures within the refuge. Mean initial total length was 102.98 ± 0.86 mm. Fish were transported to holding pens within Unit 5 in aerated containers for an acclimation period of five days. This acclimation period allowed fish to adjust to any differences in water quality between the collection location and Unit 5.

After the acclimation period, fish were removed from the holding pen, uniquely fin clipped, weighed (nearest 0.0001 g), and measured ($TL \pm 1$ mm). Five fish were placed in each enclosure ($5 \times 16 = 80$ fish total), and enclosure tops were covered with 6.4 mm plastic netting to prevent bird predation on experimental fish. After seven days, fish were removed from enclosures with a 3 mm mesh dip net and immediately weighed, measured, and preserved in 90% ethanol.

Any SAV within the enclosure was removed by hand and rake, refrigerated in the field, and returned to the laboratory for processing. All material was dried to a constant weight at 65°C and then weighed to determine biomass (± 0.0001 g dry weight).

Environmental variables were measured and recorded hourly, in each of the four ponds, throughout the experiment with a Hydrolab® Data Sonde continuous recorder including water temperature (°C), salinity (ppt), dissolved oxygen (mg L^{-1}).

Three 10-cm diameter cores were collected from the top 5 cm of substrate at random locations within vegetated and nonvegetated enclosures. Organic matter content was determined with methods similar to Moy and Levin (1991). Samples were dried to a constant weight at 65°C,

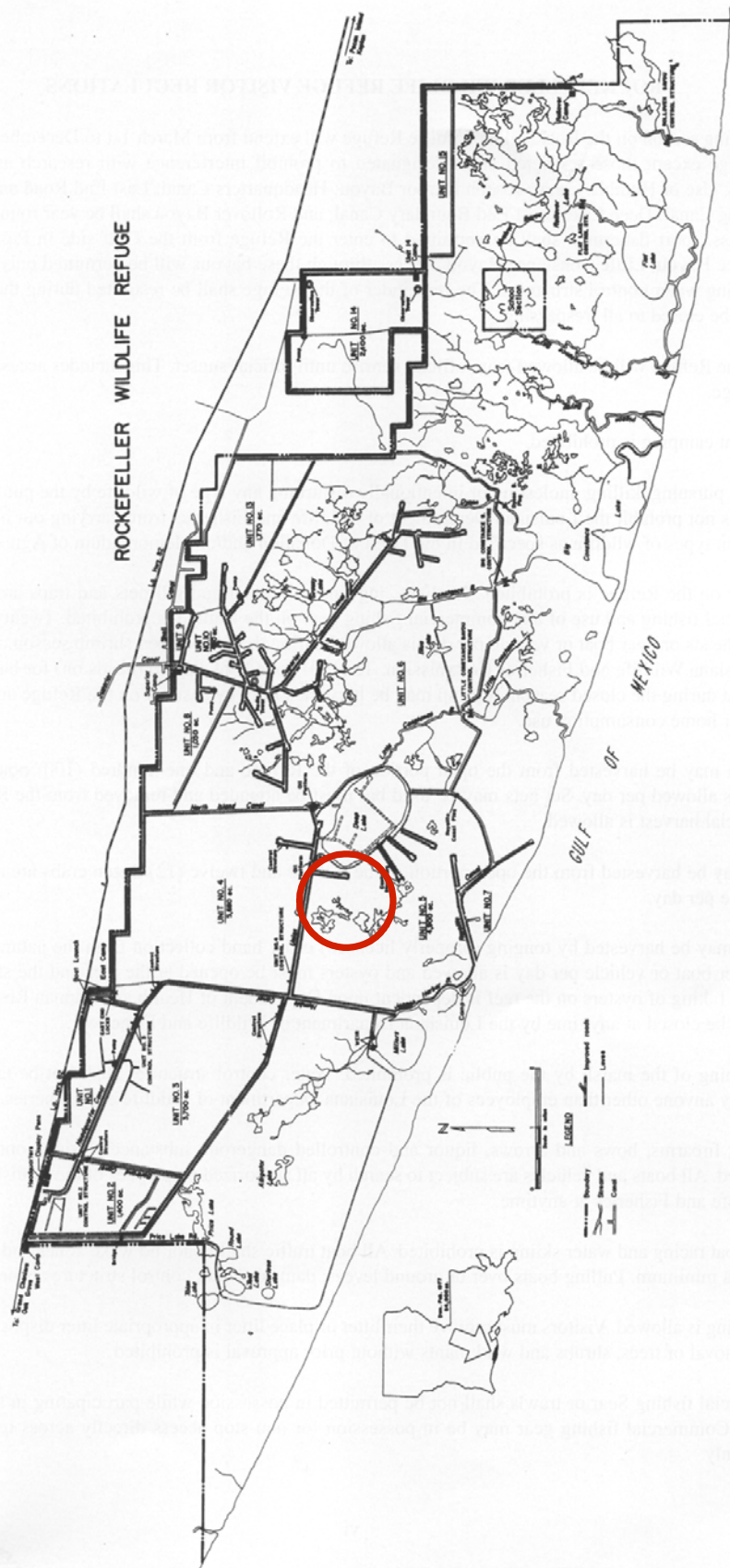


Figure 3.1. Rockefeller Wildlife Refuge. Circle is located in Unit 5 where study took place. Map adapted from Perry 2000.



Figure 3.2. Enlarged view of the circled area in Figure 3.1. Points indicate location within pond of the enclosures. Four enclosures were placed in the immediate vicinity of each point.



Figure 3.3. Enclosure design: window dimensions 35 cm X 51 cm, interior mesh 3 mm and exterior mesh 6.4 mm. Removal of water and nekton from within the enclosure.

weighed (initial dry weight), fired at 500°C in a muffle furnace for 4 hours, and weighed again (final dry weight). Percent organic matter was calculated as: $[(\text{final dry weight})/(\text{initial dry weight}) \times 100]$.

Statistical Analyses

Fish growth data were analyzed with a software package from Plymouth Routines in Multivariate Ecological Research (PRIMER). Procedures conducted with this software require few assumptions of the input data and rely on ‘non-metric’ ordination and permutation tests. The ANOSIM procedure, an analog to ANOVA, was used to test for differences in fish growth between vegetated and nonvegetated enclosures. It performs randomization tests on similarity matrices. The randomization tests calculate a test statistic for the data and repeat permutations of the data (600 times) calculating a test statistic value for each permutation. The p-value from the randomization tests is the proportion of the data permutations in the set that have test statistic values greater than or equal to the experimental results (Manly 2001). An alpha level of 0.05 was used to determine significance for all statistical analyses.

Statistical analysis for organic matter data was completed in SAS (SAS Institute, 1989). Analysis of variance (ANOVA, Proc GLM) was used to test for differences in mean percent organic matter between vegetated and nonvegetated enclosures. Environmental variables were tested with analysis of variance (ANOVA, Proc GLM) to test for differences in means among ponds.

Results

Environmental conditions within the ponds showed normal daily fluctuations of water temperature and dissolved oxygen (Table 3.1). Water temperature ranged from 22.86 to 33.52°C with a mean of $28.17 \pm 0.11^\circ\text{C}$. Salinity remained relatively constant throughout the experiment (4.70 ± 0.02 ppt) because the water control structures reduced tidal exchange in Unit 5. Dissolved oxygen ranged from 0.34 to 12.09 mg L⁻¹ and the lowest dissolved oxygen values occurred at or just before dawn. Analysis of variance indicated similar water temperatures among ponds ($p = 0.09$). Statistically, there were significant differences in salinity among ponds ($p < 0.0001$) however, mean salinities by pond ranged from 4.16 to 5.37 ppt. Mean dissolved oxygen differed significantly by pond ($p < 0.0001$), ranging from 3.73 to 5.05 mg L⁻¹.

Of the 80 experimental fish, 72 individuals (90% of the organisms) were recovered alive at the end of the 7d experiment. Fifty-five of those individuals showed negative growth, by losing weight, length, or weight and length and thus were eliminated from analysis. Data from the 17 individuals showing positive growth, were analyzed with PRIMER software because the data could not be normalized with standard normalization techniques. No significant difference in mean fish growth (weight or length) was detected between vegetated and nonvegetated habitats ($p = 0.125$) (Figure 3.4 and 3.5). Among the vegetated enclosures, SAV biomass ranged from 4.00 to 25.26 g with a mean of 11.12 ± 2.58 g. No significant difference in percent organic matter was detected between vegetated and nonvegetated enclosures ($p = 0.65$) (Figure 3.6).

TABLE 3.1. Environmental variables measured hourly throughout the 7d experiment. Water temperature (°C), dissolved oxygen (mg L⁻¹), salinity (ppt) mean ± SE and maximum and minimum values.

	Water temp. (°C)	DO (mg L ⁻¹)	Salinity (ppt)
Mean (SE)	28.17 (0.11)	4.49 (0.10)	4.70 (0.02)
Max.	33.52	12.09	5.7
Min.	22.86	0.34	3.1

Discussion

In situ nekton growth experiments are more difficult to control than laboratory experiments and often produce inconsistent results. Some studies reported greater nekton growth in vegetated habitats (Sogard 1992, Perkins-Visser et al. 1996, Stunz et al. 2002a) whereas others did not (Sogard 1992, Phelan et al. 2000). Growth rates are often related to many factors (e.g. temperature, salinity, dissolved oxygen, competition, substrate texture). In this experiment, increase of mean fish biomass was greater in vegetated than nonvegetated enclosures, but no statistical differences were found. The small sample size (n = 17) may have contributed to a lack of significant difference between treatments. If the experiment had been repeated with more individuals, statistically significant differences in growth may have been detected. Several other factors may have led to similar nekton growth in the vegetated and nonvegetated habitats in this study: (1) initial size of experimental fish, (2) condition of experimental fish, (3) similarity of substrate between habitats, (4) environmental conditions.

The initial size of the experimental fish was relatively large (mean TL = 102.98 ± 0.86 mm) compared to another study investigating growth in Sciaenids (mean SL = 19 ± 0.6 mm) (Stunz et al. 2002a). Atlantic croaker *Micropogonias undulatus* typically range from 110 to 120 mm standard length at one year in Louisiana marshes (Tarbox 1974). So, the experimental fish were young-of-the-year, but were not newly settled. Other experiments, showing differences

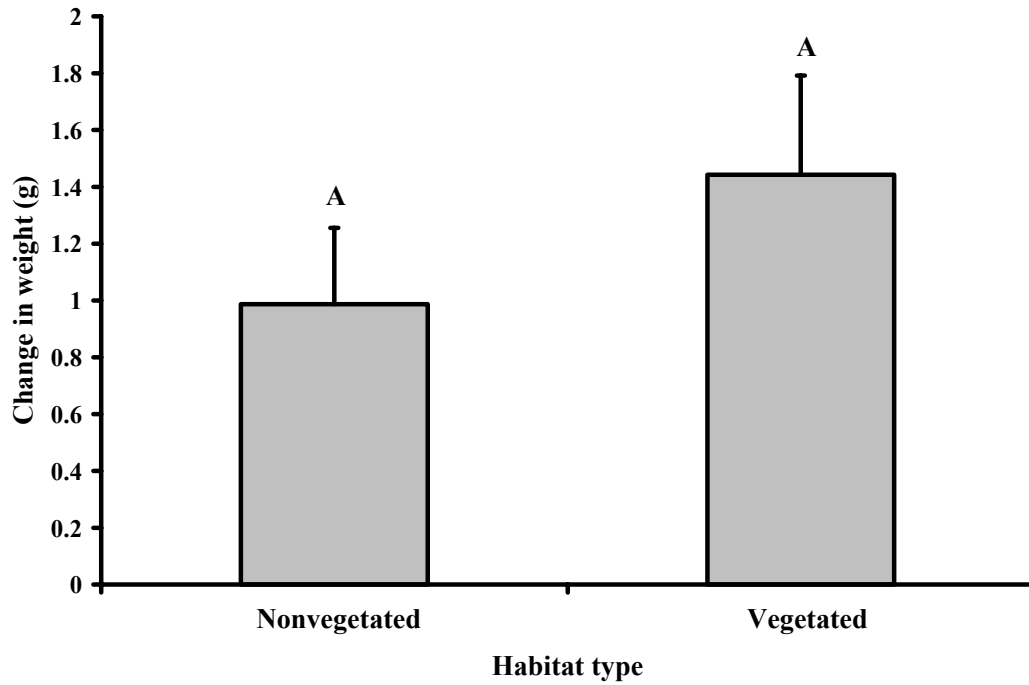


Figure 3.4. Mean change in fish weight (g) for the reduced data set ($n = 17$) between vegetated and nonvegetated enclosures. Similar letters indicate no significant differences ($p = 0.125$). Error bars represent standard error.

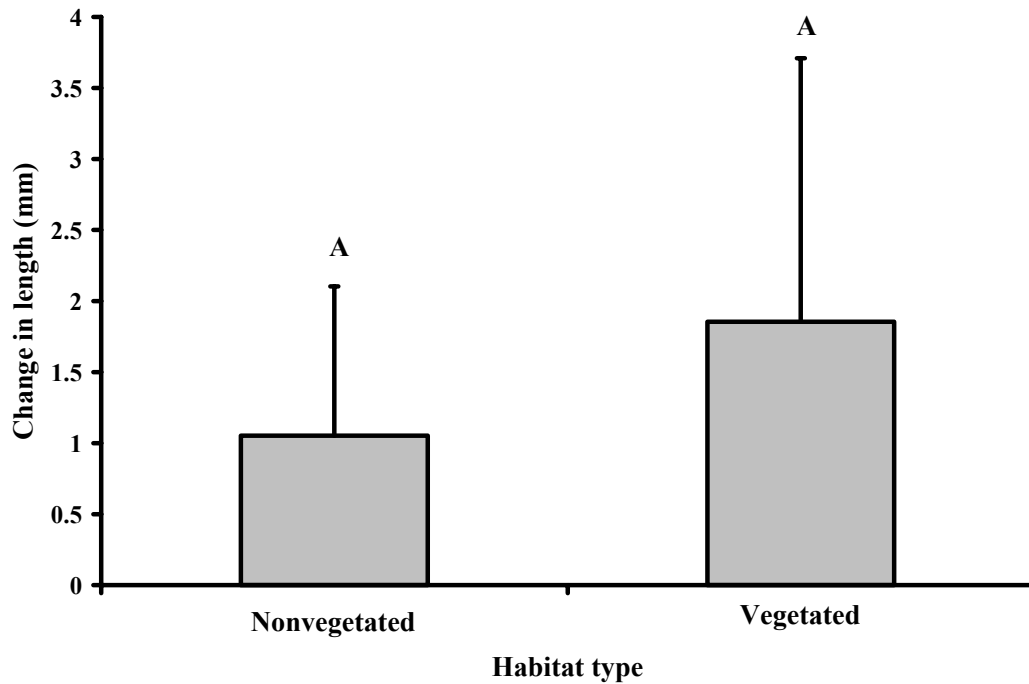


Figure 3.5. Mean change in fish length (g) for the reduced data set ($n = 17$) between vegetated and nonvegetated enclosures. Similar letters indicate no significant differences ($p = 0.125$). Error bars represent standard error.

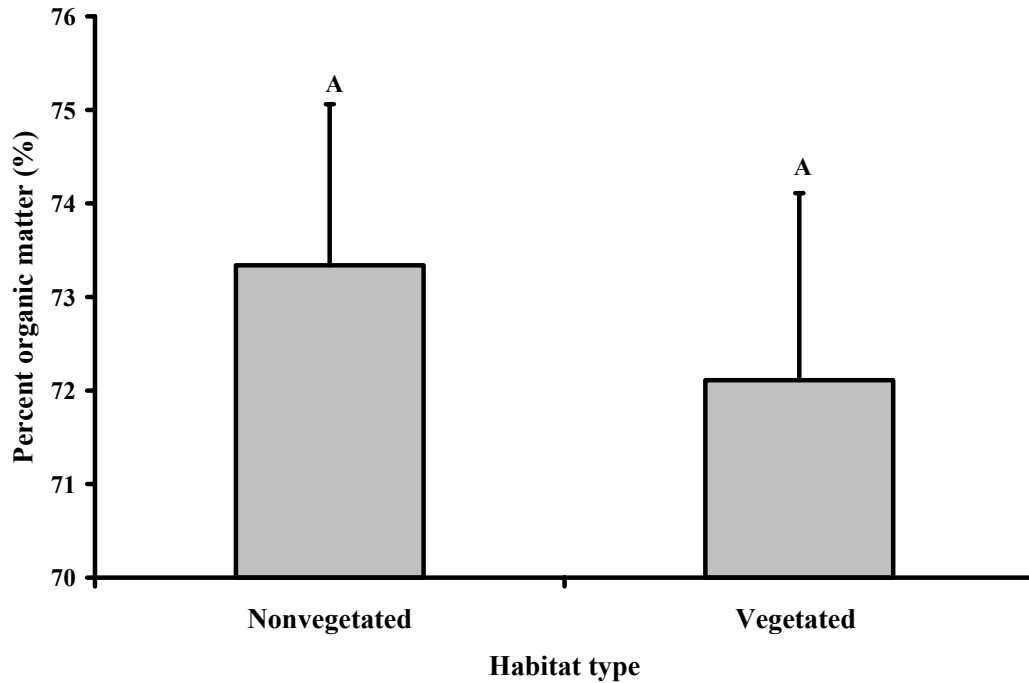


Figure 3.6. Percent organic matter by habitat type: vegetated and nonvegetated. Similar letters indicate no significant difference ($p = 0.65$). Error bars represent standard error.

in growth between vegetated and nonvegetated habitats, were conducted on newly settled fish. Rooker et al. (1998) found that postsettlors (≤ 40 mm SL) enter seagrass as temporary residents and then migrate to alternative habitats shortly after arrival. These experimental fish were not collected in a vegetated habitat, so it is likely that they had already completed their SAV associated stage, were large enough not to depend on vegetated habitat for refuge, and had migrated to alternative habitats. Even though Atlantic croaker *Micropogonias undulatus* are benthic feeders consuming infauna and epifauna throughout their lives (Currin et al. 1984), smaller individuals are associated with SAV habitats more than larger individuals (70 - 110 mm SL) (Rooker et al. 1998). Petrik et al. (1999) found newly settled Atlantic croaker *Micropogonias undulatus* (15 - 20 mm SL) to be habitat generalists, showing no difference in habitat use between marsh edge, SAV, and sand. However, small individuals have faster growth rates than larger individuals and show greater changes in somatic growth over shorter periods of

time. Large individuals are > 160 days old, at this stage age specific growth rates are < 0.010 as compared to 0.020 at 60 - 80 days (Nixon and Jones 1997). Therefore, differences in somatic growth would be easier to detect in smaller individuals than were used in this experiment.

The experimental fish in this study were showing signs of stress after the acclimation period (i.e. worn caudal fins). After the 7d experiment, the caudal fins of some fish showed additional damage resulting in 25 individuals losing length during the experiment. Weight loss during an enclosure experiment can be a result of eating cessation due to handling stress. However, loss of length is not a normal response to handling stress. Only one other study, Sogard (1992), reported negative changes in total length at two sites in a growth experiment with winter flounder *Pseudopleuronectes americanus*, but the author did not address potential causes of this result. I speculate that the fish lost length because their caudal fins continued to deteriorate from the initial injuries acquired during the acclimation period. The fish may not have truly lost body length during the experiment, but any change in caudal fin shape is reflected in total length measurements. This problem could be avoided by measuring standard length rather than total length at the beginning and the end of an experiment thereby alleviating any changes in length resulting from caudal fin injuries.

It is common for vegetated habitats to contain more organic matter (i.e. detritus) than nonvegetated habitats because SAV derived macroscopic debris and detrital organic matter litters the substrates associated with vegetated habitats (Miyajima et al. 1998). However, in this study, percent organic matter of the substrates was similar between vegetated and nonvegetated habitats. Therefore, fish growth could not have been supplemented by additional organic matter in vegetated habitats compared to nonvegetated habitats.

Fish growth within vegetated habitats may have been negatively and disproportionately influenced by hypoxic conditions compared to the nonvegetated habitats. Some studies have

found salinity, temperature, dissolved oxygen, and sediment structure influenced fish growth (Sogard 1992, Peterson et al. 1999) more than external factors (i.e. number of grass stems, nekton density) (Baltz et al. 1998). In this study, mean salinity and mean water temperature were within the tolerance range of Atlantic croaker and should not have negatively influenced growth (Wannamaker and Rice 2000). Dissolved oxygen was the only environmental condition that fluctuated greatly during the experiment. Although dissolved oxygen was not measured hourly within each enclosure, conditions did become hypoxic (≤ 2 ppt) near dawn. Hypoxic conditions induce respiratory stress in Atlantic croaker causing an increase in ventilation rates (Wannamaker and Rice 2000). Atlantic croaker *Micropogonias undulatus* are sensitive to hypoxic conditions and would normally avoid areas with dissolved oxygen concentrations ≤ 2 ppt (Wannamaker and Rice 2000). Low dissolved oxygen conditions within the vegetated habitats were likely more severe than in the nonvegetated habitats due to the effects of plant respiration. All of the experimental fish that died were in vegetated habitats and several corpses were recovered showing signs of low dissolved oxygen stress (gaping mouths).

Suggestions for Future Experiments

Future experiments should be conducted with newly settled Atlantic croaker (20 - 40 mm SL), so that differences in growth can be easily detected. To alleviate any variation in length that resulted from caudal fin injuries acquired during the experiment, standard length should be used as opposed to total length. Identifying and quantifying the infaunal communities associated with vegetated and nonvegetated habitats would be useful to determine if food availability was similar between habitats or was limiting within one habitat. Gut content analysis would also provide information regarding the diets of fish within both habitats. Atlantic croaker *Micropogonias undulatus* are known to feed largely on benthic meiofauna (e.g., harpacticoid copepods) (Soto et

al. 1998). Quantifying the abundance of benthic meiofauna within the two habitats and in fish guts could help identify why differences in fish growth may occur between habitats.

Because the sample size was small, no definite determination was made from this study regarding nekton growth in vegetated and nonvegetated habitats. However, the functional role of SAV in supporting nekton communities is an important question and should be further investigated. Understanding the functional role of SAV is crucial in the identification of essential fish habitat for estuarine dependent nekton species, especially recreationally and commercially important species such as Atlantic croaker *Micropogonias undulatus*.

CHAPTER IV.

GENERAL CONCLUSIONS

The primary goal of this study was to examine the relative value of SAV for supporting nekton. The data indicated that SAV habitats supported greater nekton densities, biomass, and diversity than nonvegetated bottom habitats. Nekton used the two vegetated habitats (near-marsh edge SAV and inner-pond SAV) similarly. Nekton use in the nonvegetated bottom differed significantly from the vegetated habitats. In this study, SAV biomass, not location, appeared to be driving differences in nekton use between vegetated and nonvegetated habitat types. The vegetated habitats were located in close proximity to each other, potentially masking any differences due to location that might have occurred if the vegetated habitats had occurred at greater distances from one another.

The fish growth experiment attempted to identify the specific functional role of SAV in supporting nekton. The results of this experiment were inconclusive because of a small sample size. No statistically significant difference in growth was detected between vegetated and nonvegetated habitats, but a slightly higher mean size of fish in vegetated enclosures may indicate that SAV provided a better growth environment than nonvegetated bottom. Percent organic matter was similar between the two habitat types indicating that a portion of the food resources for juvenile nekton was similar between habitats. In future studies, infaunal communities should be examined to determine if prey densities are similar between habitat types. Further investigation is necessary to definitively determine if vegetated habitats do support nekton growth more than nonvegetated habitats.

Vegetated habitats supported nekton more than nonvegetated bottom. Submerged aquatic vegetation appeared to be playing a crucial role in supporting estuarine dependent nekton.

Secondary production within vegetated habitats was greater than in nonvegetated bottom.

Changes in the abundance and distribution of SAV, due to natural and/or anthropogenic causes, will thus likely influence nekton communities, thereby affecting secondary production.

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APPENDIX A. NEKTON DENSITY

Mean (SE) density (individuals m⁻²) for nekton by habitat and month. Asterisks indicate where all organisms (N) were collected in one sample, therefore no mean or SE.

Species	Mean (SE) Density by Habitat					Mean (SE) Density by Month									
	Near-marsh		Inner-pond		Non-vegetated bottom	Mean (SE) Density by Month					Mean (SE) Density by Month				
	SAV	N	SAV	N		Sept. 2001	N	Dec. 2001	N	Mar. 2002	N	July 2002	N		N
<i>Alosa chrysochloris</i>	0.00	0	0.00	0	16.00*	0.00	0	16.00*	16	0.00	0	0.00	0	0.00	0
<i>Anchoa mitchilli</i>	1.00 (0.00)	2	6.50 (5.50)	13	5.00 (1.29)	6.00 (5.00)	12	3.00 (1.00)	9	1.50 (0.29)	6	4.00 (1.00)	6	4.00 (1.00)	8
<i>Brevoortia patronus</i>	5.50 (4.50)	11	8.50 (6.20)	34	3.33 (1.86)	0.00	0	0.00	0	9.17 (3.79)	55	0.00	0	0.00	0
<i>Callinectes sapidus</i>	2.86 (0.40)	20	2.17 (0.60)	13	1.50 (0.50)	2.00 (0.32)	10	2.29 (0.52)	16	1.25 (0.25)	5	2.00 (0.41)	8	2.00 (0.41)	8
<i>Cupeid</i>	0.00	0	2.00*	2	0.00	0.00	0	2.00*	2	0.00	0	0.00	0	0.00	0
<i>Cyprinodon variegatus</i>	69.75 (18.14)	558	54.00 (26.48)	432	1.00 (0.00)	23.5 (7.26)	188	3.20 (0.97)	16	1.00 (0.00)	3	98.38 (32.87)	787	98.38 (32.87)	787
<i>Farfantepenaeus aztecus</i>	2.00*	2	0.00	0	1.00*	1.00*	1	1.00 (0.00)	2	0.00	0	0.00	0	0.00	0
<i>Fundulus grandis</i>	1.00 (0.00)	3	1.50 (0.50)	3	0.00	1.00 (0.00)	3	0.00	0	1.00 (0.00)	3	0.00	0	0.00	0
<i>Fundulus pulvereus</i>	18.00 (9.77)	126	15.17 (7.07)	91	0.00	3.50 (2.50)	7	3.00 (2.00)	6	0.00	0	34.00 (15.18)	204	34.00 (15.18)	204
<i>Gambusia affinis</i>	2.00*	2	0.00	0	0.00	0.00	0	0.00	0	0.00	0	2.00*	2	2.00*	2
<i>Gobiosoma bosc</i>	1.00 (0.00)	3	3.80 (1.24)	19	0.00	0.00	0	2.00 (1.00)	6	1.00*	1	5.00 (2.00)	15	5.00 (2.00)	15
<i>Leiostomus xanthurus</i>	1.00 *	1	0.00	0	1.00 (0.00)	0.00	0	0.00	0	1.50 (0.50)	3	0.00	0	0.00	0
<i>Litopenaeus setiferus</i>	14.67 (3.95)	88	17.13 (4.60)	137	4.33 (1.86)	27.13 (6.92)	217	3.40 (1.21)	17	1.00*	1	3.00*	3	3.00*	3
<i>Lucania parva</i>	89.00 (27.60)	712	99.63 (29.70)	797	1.00*	26.25 (10.93)	210	13.00 (3.69)	78	1.25 (0.25)	5	173.86 (50.76)	1217	173.86 (50.76)	1217
<i>Menbrus martinica</i>	0.00	0	1.00*	1	1.00*	0.00	0	1.00*	1	0.00	0	1.00*	1	1.00*	1
<i>Menidia beryllina</i>	4.00*	4	3.20 (1.16)	16	0.00	3.40 (1.03)	17	2.00*	2	1.00*	1	0.00	0	0.00	0
<i>Microgobius gulosus</i>	2.80 (1.11)	14	3.00 (1.41)	12	1.00 (0.00)	4.33 (2.06)	26	1.00*	1	0.00	0	1.00*	1	1.00*	1
<i>Microphis brachyurus</i>	13.00*	13	7.00 (1.73)	21	0.00	0.00	0	0.00	0	0.00	0	11.33 (5.90)	34	11.33 (5.90)	34
<i>Microponogonias undulatus</i>	0.00	0	2.50 (1.50)	5	5.25 (3.33)	0.00	0	6.250 (2.95)	25	1.00*	1	0.00	0	0.00	0
<i>Mugil cephalus</i>	3.00*	3	1.00*	1	1.00 (0.00)	3.00*	3	1.00*	1	1.00*	1	1.00*	1	1.00*	1
<i>Mugil curema</i>	3.00*	3	0.00	0	0.00	0.00	0	0.00	0	3.00*	3	0.00	0	0.00	0
<i>Myrophis punctatus</i>	2.80 (1.11)	14	2.00 (0.58)	6	1.00 (0.00)	2.60 (0.93)	13	1.00*	1	1.00*	1	2.33 (0.88)	7	2.33 (0.88)	7
<i>Palaemonetes</i> spp.	50.86 (16.39)	407	53.88 (13.19)	431	3.29 (1.29)	21.29 (8.88)	149	21.50 (8.98)	129	12.83 (2.98)	77	72.29 (23.39)	506	72.29 (23.39)	506
<i>Poecilia latipinna</i>	57.00 (17.28)	456	57.43 (19.11)	402	0.00	5.20 (1.24)	26	2.67 (1.12)	16	0.00	0	116.57 (34.11)	816	116.57 (34.11)	816
<i>Sciaenops ocellatus</i>	0.00	0	0.00	0	2.00*	2.00*	2	0.00	0	0.00	0	0.00	0	0.00	0
<i>Syngnathus floridae</i>	3.00 (1.00)	6	2.00*	2	1.00*	0.00	0	2.00 (0.00)	4	5.00*	5	0.00	0	0.00	0
<i>Syngnathus louisianae</i>	1.00 (0.00)	2	0.00	0	0.00	0.00	0	1.00 (0.00)	2	0.00	0	0.00	0	0.00	0
<i>Syngnathus scovelli</i>	5.00 (2.08)	15	2.75 (1.44)	11	0.00	1.50 (0.50)	3	0.00	0	0.00	0	5.75 (1.65)	23	5.75 (1.65)	23

APPENDIX B. NEKTON BIOMASS

Untransformed mean (SE) biomass (g wet wt m⁻²) for nekton by habitat and month. Asterisks indicate samples with N=1, therefore no mean or SE.

Species	Mean (SE) Biomass by Habitat				Mean (SE) Biomass by Month											
	Near-marsh SAV		Inner-pond SAV		Non-vegetated bottom		Sept. 2001		Dec. 2001		Mar. 2002		July 2002		N	
	SAV	N	SAV	N	bottom	N	SAV	N	SAV	N	SAV	N	SAV	N	SAV	N
<i>Alosa chrysochloris</i>	0.00	0	0.00	0	0.16 (0.01)	16	0.00	0	0.16 (0.01)	16	0.00	0	0.00	0	0.00	0
<i>Anchoa mitchilli</i>	0.37 (0.00)	2	0.28 (0.03)	13	0.31 (0.07)	20	0.31 (0.03)	12	0.25 (0.11)	9	0.18 (0.06)	6	0.45 (0.09)	8	0.00	0
<i>Brevoortia patronus</i>	0.06 (0.01)	11	0.07 (0.00)	34	0.07 (0.01)	10	0.00	0	0.00	0	0.07 (0.00)	55	0.00	0	0.00	0
<i>Callinectes sapidus</i>	16.41 (6.43)	20	0.95 (0.22)	13	0.13 (0.02)	6	9.86 (9.31)	10	4.80 (2.46)	16	0.24 (0.06)	5	20.60 (11.29)	8	0.00	0
<i>Clupeid</i>	0.00	0	0.06 (0.02)	2	0.00	0	0.00	0	0.06 (0.02)	2	0.00	0	0.00	0	0.00	0
<i>Cyprinodon variegatus</i>	0.23 (0.01)	558	0.17 (0.01)	432	0.89 (0.40)	4	0.22 (0.01)	188	0.82 (0.13)	16	1.17 (0.42)	3	0.19 (0.01)	787	0.00	0
<i>Farfantepenaeus aztecus</i>	0.41 (0.40)	2	0.00	0	1.22 *	1	0.01 *	1	1.01 (0.20)	2	0.00	0	0.00	0	0.00	0
<i>Fundulus grandis</i>	3.87 (1.99)	3	0.92 (0.76)	3	0.00	0	0.34 (0.18)	3	0.00	0	4.45 (1.57)	3	0.00	0	0.00	0
<i>Fundulus pulvereus</i>	0.22 (0.02)	126	0.23 (0.02)	91	0.00	0	0.52 (0.14)	7	0.69 (0.18)	6	0.00	0	0.20 (0.01)	204	0.00	0
<i>Gambusia affinis</i>	0.19 (0.09)	2	0.00	0	0.00	0	0.00	0	0.00	0	0.00	0	0.19 (0.09)	2	0.00	0
<i>Gobiosoma bosc</i>	0.27 (0.15)	3	0.26 (0.04)	19	0.00	0	0.00	0	0.25 (0.05)	6	0.16 *	1	0.27 (0.06)	5	0.00	0
<i>Leiostomus xanthurus</i>	0.03 *	1	0.00	0	0.27 (0.08)	2	0.00	0	0.00	0	0.19 (0.09)	3	0.00	0	0.00	0
<i>Litopenaeus setiferus</i>	0.43 (0.05)	88	0.36 (0.02)	137	0.36 (0.09)	13	0.40 (0.03)	217	0.23 (0.03)	17	0.31 *	1	0.06 (0.02)	3	0.00	0
<i>Lucania parva</i>	0.14 (0.00)	712	0.17 (0.00)	797	0.01 *	1	0.15 (0.01)	210	0.23 (0.02)	78	0.45 (0.14)	5	0.15 (0.00)	1217	0.00	0
<i>Membras martinica</i>	0.00	0	0.41 *	1	0.25 *	1	0.00	0	0.25 *	1	0.00	0	0.41 *	1	0.00	0
<i>Menidia beryllina</i>	0.41 (0.11)	4	0.24 (0.04)	16	0.00	0	0.26 (0.05)	17	0.30 (0.09)	2	0.49 *	1	0.00	0	0.00	0
<i>Microgobius gulosus</i>	0.34 (0.08)	14	0.27 (0.10)	12	0.24 (0.09)	2	0.26 (0.05)	26	0.33 *	1	0.00	0	1.29 *	1	0.00	0
<i>Microphis brachyurus</i>	0.24 (0.04)	13	0.20 (0.02)	21	0.00	0	0.00	0	0.00	0	0.00	0	0.22 (0.02)	34	0.00	0
<i>Micropogonias undulatus</i>	0.00	0	0.11 (0.03)	5	0.27 (0.03)	21	0.00	0	0.24 (0.03)	25	0.05 *	1	0.00	0	0.00	0
<i>Mugil cephalus</i>	10.27 (1.06)	3	34.63 *	1	13.62 (7.30)	2	10.27 (1.06)	3	34.63 *	1	20.92 *	1	6.33 *	1	0.00	0
<i>Mugil curema</i>	0.23 (0.04)	3	0.00	0	0.00	0	0.00	0	0.00	0	0.23 (0.04)	3	0.00	0	0.00	0
<i>Myrophis punctatus</i>	0.56 (0.08)	14	0.44 (0.14)	6	1.27 (0.27)	2	0.64 (0.08)	13	1.54 *	1	1.01 *	1	0.31 (0.07)	7	0.00	0
<i>Palaemonetes</i> spp.	0.14 (0.01)	407	0.15 (0.00)	431	0.20 (0.02)	23	0.15 (0.01)	149	0.11 (0.01)	129	0.18 (0.01)	77	0.15 (0.01)	506	0.00	0
<i>Poecilia latipinna</i>	0.12 (0.01)	456	0.10 (0.01)	402	0.00	0	0.28 (0.04)	26	0.35 (0.12)	16	0.00	0	0.10 (0.00)	816	0.00	0
<i>Sciaenops ocellatus</i>	0.00	0	0.00	0	0.06 (0.02)	2	0.06 (0.02)	2	0.00	0	0.00	0	0.00	0	0.00	0
<i>Syngnathus floridae</i>	0.15 (0.03)	6	0.03 (0.00)	2	0.10 *	1	0.00	0	0.12 (0.05)	4	0.12 (0.04)	5	0.00	0	0.00	0
<i>Syngnathus louisianae</i>	0.05 (0.01)	2	0.00	0	0.00	0	0.00	0	0.05 (0.01)	2	0.00	0	0.00	0	0.00	0
<i>Syngnathus scovelli</i>	0.33 (0.06)	15	0.45 (0.09)	11	0.00	0	0.14 (0.07)	3	0.00	0	0.00	0	0.41 (0.05)	23	0.00	0

APPENDIX C. GROWTH EXPERIMENT: LESSONS LEARNED

The growth experiment was attempted two times previous to the one described in Chapter III. This serves as a description of the details leading up to the final experiment.

The first attempt was made in the spring of 2002 at Marsh Island Wildlife Refuge. Enclosures were made with 1m² wood frames and covered in 3 mm plastic mesh (Figure A). Wooden legs extended from the base of the enclosures and were placed into the substrate to prevent wind from moving the enclosures. Enclosures were covered with hinged lids. The lids were created with wood frames and wire mesh to prevent avian predation. Wood braces on the bottom of the enclosure walls, connecting the plastic mesh to the enclosure legs, were placed into the substrate to prevent burrowing organisms from entering the enclosures.



Figure C.1. 1 m² wood enclosure. Tops of the enclosure covered in wire mesh.

Numerous problems were encountered using this enclosure design. The enclosures were generally not solid enough to take the forces needed to drive the legs into the substrate. The wood frames torqued, creating enclosures that were no longer square. The plastic mesh tore

away from the frame when the enclosures torqued, creating holes in the mesh that had to be repaired before the experiment could begin. Recovery of experimental organisms (Atlantic croaker *Micropogonias undulatus*) was low at the end of the 10 d experiment, possibly for several reasons: 1) blue crabs *Callinectes sapidus* created holes in the plastic mesh allowing organisms to enter the enclosures and experimental organisms to escape, 2) enclosures could not be drained of water, so it was difficult to recover experimental organisms with a dip net, 3) there were areas for the experimental organisms to hide and prevent being recaptured because the plastic mesh was stapled to the outside of the enclosures creating hiding spaces next to the wood frame (but within the enclosure). Overall, the enclosures were large, difficult to clear, labor intensive, expensive to build, and not sturdy enough for a field experiment.

The second attempt was at Rockefeller Wildlife Refuge in late spring of 2003. The enclosure design and locations were identical to that described in Chapter III. Atlantic croaker *Micropogonias undulatus* were collected with a seine, outside of the impoundment, and were not acclimated to the conditions within the impoundment prior to the experiment. The fish were newly settled, therefore much smaller than those used in Chapter III. Recovery was low at the end of the 7 d experiment. Very few fish were recovered alive and carcasses were difficult to locate, as well. I believe that the DO fell to levels intolerable for newly settled Atlantic croaker *Micropogonias undulatus* and the fish died within the first few days of the experiment. The carcasses likely decomposed and therefore were not recovered.

Although it may be beneficial to conduct the experiment with newly settled fish, the experiment should be completed in the same location in which the fish are collected and early in the spring when DO levels are more favorable.

The enclosure design described in Chapter III worked well. To stabilize the enclosures in high wind situations, I would suggest placing several rebar pieces into the substrate around the enclosures. The key to recovering the fish seemed to be the sump pump to drain water from the enclosures (as described in Chapter III). This allowed fish to be seen and made dip netting successful.

VITA

Sarai Christine Kanouse graduated from Greenville High School in Greenville, Texas, in 1994. Sarai attended Texas A&M University in Galveston, Texas, where she earned a Bachelor of Science in marine fisheries and a Bachelor of Science in marine biology. While at Texas A&M University at Galveston, she became interested in estuarine habitats and their associated fisheries. Following graduation, Sarai served as a Peace Corps Volunteer in Gabon, Africa, where she was involved in farming and tilapia aquaculture. She then worked as a fisheries technician for US Fish and Wildlife Service in Washington and for the USGS North Carolina Cooperative Fish and Wildlife Research Unit. In August 2001, Sarai began her graduate studies in the School of Renewable Natural Resources at Louisiana State University under the guidance of Dr. Megan La Peyre. The degree of Master of Science will be awarded in December 2003.