RESPONSE OF ESTUARINE-DEPENDENT NEKTON TO SEAGRASS FRAGMENTATION

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ABSTRACT

Seagrass beds play a crucial role in the life history for many estuarine-dependent species by serving as nursery habitat. Seagrass habitats are declining worldwide with increasing anthropogenic activity. Typically, habitat loss is preceded by fragmentation, a process whereby large continuous habitats are broken into smaller more isolated patches. Seagrass beds, like many other habitats, are increasingly becoming fragmented, but little empirical data exists regarding how this degradation process may affect associated biota. To study this, fragmented seagrass habitats (primarily *Halodule wrightii*) within Corpus Christi and Aransas Bays were delineated, quantified, and mapped using a Trimble unit during spring and fall 2009. Within each bay, nine plots representing three levels of fragmentation were established. Samples were taken using an epibenthic sled and nekton densities were compared among fragmentation levels. Results were varied between seasons, whereby higher densities of nekton were found in more fragmented habitats during the spring while higher densities of nekton were found within continuous habitats during the fall. Red drum (*Sciaenops ocellatus*) was used as a model species to test the impact of fragmentation on a typical estuarine-dependent species. I investigated red drum density, growth, and movement in response to varying levels of fragmentation (High, Medium, and Low) in seagrass beds in Corpus Christi Bay and Aransas Bay, TX. There was no significant difference in densities of red drum among fragmentation levels. However, I did observe a significant size effect with larger fish more common in non-fragmented areas. Growth rates were also compared among fragmented habitats using RNA:DNA ratios and otolith microstructure. No significant effect of growth among

fragmentation levels was found, and these results suggested that other parameters such as habitat selection or increased predator avoidance may drive red drum densities. To examine movement at the landscape level within and among fragmented seagrass meadows, two hundred newly settled juvenile red drum were captured, tagged, and released into a single patch within three highly fragmented networks. Within twenty four hours, only one fish was recaptured within the original fragmented network, suggesting either movement out of the network or predation. The majority of recaptured fish (85%) were found in a non-fragmented seagrass bed 50 m from their release point. Movement results suggest a temporal transition in size and density of juvenile red drum from fragmented sites to more continuous seagrass beds. Overall, the effect of fragmentation on organisms seems to be species specific, based upon individual needs and life history. In the case of red drum, there seems to be a fragmentation "threshold", whereby a habitat may become too fragmented, effectively isolating newly settled red drum, leaving them vulnerable to predation.

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INTRODUCTION

Habitat fragmentation within the world's ecosystems is a widespread and ubiquitous problem. Many habitats are in decline and/or degradation worldwide, and some are at risk of being lost permanently **(**Vitousek et al. 1997, Rapport et al. 1998, Pandolfi et al. 2003, Waycott et al. 2009). As habitats experience degradation or loss, they typically go through phases of fragmentation (Jaeger 2000). Habitat fragmentation occurs when large contiguous habitats are broken into small discrete habitats with increasing isolation among patches (Bender et al. 1998), and this phenomenon is seen in both terrestrial and marine environments. Often, these irregular, widely-separated patches shrink in size and eventually disappear (Forman 1995). Typically, attention has focused on more obvious and well known environments such as tropical rain forests, coral reefs, and mangrove forest. However, recently more attention has been given to seagrass habitat as a threatened ecosystem with overall loss rates comparable to those more charismatic ecosystems (Waycott et al. 2009).

Across the globe, seagrasses are experiencing stressors such as overexploitation, physical modification, nutrient and sediment pollution, introduction of nonnative species, and global climate change (Waycott et al. 2009), all of which can lead to habitat loss. Human presence can cause direct loss of seagrass habitat through many processes such as coastal engineering and propeller scarring, while indirect loss occurs through degradation of water quality through sedimentation and water-shed eutrophication. For example, from 1950 to 1989, an estimated 2,200 acres of seagrass (*Halodule wrightii*) has been lost from West Galveston Bay (Pulich and White 1991) leaving only 700 acres of submerged vegetation. This loss was attributed to a combination of removing groundwater, dredging,

and sediment destabilization. With increasing human population and increased activity along the south Texas coast similar habitat loss and fragmentation is likely. Since seagrass meadows are essential coastal habitats that provide a variety of services, their decline or loss could have far reaching effects.

Seagrass beds are considered one of the more productive habitats on earth producing more organic dry weight per day ($2.7g/m^2/d^{-1}$) than coral reefs ($0.8\ 7g/m^2/d^{-1}$) and cultivated land ($1.8\ 7g/m^2/d^{-1}$) **(**Duarte and Chiscano 1999). Seagrasses provide essential ecological services to coastal environments such as nutrient recycling, sediment stabilization, improved water quality, and habitat. Large numbers of organisms are associated with seagrass because of the high abundance of food and predation refuge (Zieman and Zieman 1989). Moreover, many ecologically and economically important marine species use seagrass meadows as "nurseries" (Boesch and Turner 1984, Minello 1999, Beck et al 2001). Given their importance, degradation of these habitats through fragmentation could have far-reaching consequences such as disruption of ecosystem processes or declines in fish abundance.

The dramatic decline of the world's fisheries is well known and understandably, a source of great concern (Pauly et al. 2002). In addition to overfishing (Jackson et al. 2001), other human activities have also been important in fisheries decline (Hilborn et al. 2003). Predominant among these anthropogenic impacts is destruction and fragmentation of critical habitat, particularly of nursery areas (Vitousek et al. 1997). The availability of nursery and juvenile habitats play a crucial role in determining the dynamics and structure of marine fish populations (Connell and Jones 1991). Examining the relationship between fish recruitment and nursery habitat at the landscape-scale will

provide a better understanding of spatial and temporal habitat requirements for population persistence, and the impacts of habitat fragmentation on these processes. There is an important link between fish recruitment and nursery habitat quality that sustains estuarine-dependent fish populations at various life stages, and these processes may be compromised by habitat fragmentation (Levin and Stunz 2005). Nursery habitats must provide rapid growth for juvenile fish and refuge from potential predators. Rapid growth into adult stages often confers lower prey vulnerability and ultimately contribution to adult populations. Therefore, measuring growth is a useful proxy for accessing the health of nursery habitats and survival into adulthood (Houde 1987).

Recently, RNA:DNA ratios and otolith microstructure have become well-accepted methods for determining both age and growth rates in juvenile fish (Hovenkamp 1991, Rooker & Holt 1996, Caldarone et al. 2006). DNA is species-specific and relatively constant throughout an organism's life while RNA levels increase with somatic growth. RNA codes for protein, which in turn contributes to the development of new biomass. Higher RNA:DNA ratios represent faster growth while lower RNA:DNA ratios reflect slower growth. Similarly, patterns recorded in otolith microstructure are useful for measuring fish growth rates at various life history stages (Campana 1985, Secor et al. 1991) and can be also be used as a proxy for recent fish growth (Stunz et al. 2002).

Red drum are an estuarine-dependent species and represent an excellent model to assess growth rates from different levels of fragmented seagrass habitat. Moreover, well-established models for both otolith microstructure and RNA/DNA ratios in red drum have been developed (Rooker & Holt 1996, Stunz et al. 2002). Red drum (*Sciaenops ocellatus*) are common to the Gulf of Mexico and support an economically important fishery

(Patillo et al. 1997). According to the 2006 NOAA Fisheries U.S. Fisheries Report, red drum generate 5.5 billion dollars of income for Texas. Persistence of the red drum population along the Texas coast is dependent upon seagrass habitat (Holt et al 1983, Rooker and Holt 1997). Transition from juvenile into adulthood is critical to sustain these populations, and relative changes in daily growth can have important consequences for recruitment (Houde 1987) and in regulating the year-class strength of red drum (Scharf 2000). Juvenile drum can enhance their survival by selecting environments that maximize energy intake and minimize predator interactions (Sogard 1997).

Understanding animal movement patterns within habitats is fundamental to the study of animal ecology and to resource management strategies (Pittman and McAlpine 2003). As a habitat becomes more fragmented, the distance between individual patches increases and density dependent processes like competition and predation tend to be greater in small habitats. As a patch size shrinks in size, these effects may become magnified unless migration to more suitable habitat occurs. The well known "Settle-and-Stay Hypothesis" (Bell et al. 1987) predicts that fish remain in the seagrass beds at which they first arrive, because predation risks are too great and outweigh benefits of moving to new areas. This hypothesis has rarely been observed in an estuarine-dependent species within a fragmented habitat. Red drum movement among seagrass types has been suggested (Rooker et al. 1998), and fragmentation in seagrass meadows presents an opportunity to test this hypothesis under a variety of interactive conditions such as patch size dynamics (i.e., settle and stay in response to patch size), distance to nearest suitable habitat, and density of conspecifics.

Mark-and-recapture experiments are effective ways to monitor animal movements and have been used successfully in nearly all forms of animal including birds, bears, beetles, butterflies, and bonito (White & Burnham 1999, Mowat & Strobeck 2000, Turchin et al. 1993, Baguette & 1994, Etnier 1972). A recent study tagged red drum (<50 mm TL) using visible implant elastomere (VIE; Northwest Marine Technology, Inc.) in order to track movement patterns within large continuous seagrass meadows (Bushon et al. 2007). They successfully recaptured a marked red drum three days after release, 200 meters from release point. This result suggests that juvenile red drum are capable of large-scale movement within a continuous seagrass meadow. I have expanded on this work by examining inter-patch movement of juvenile red drum within a fragmented network of seagrass. Results of this research will provide valuable information in determining which attributes of the seagrass habitat are essential to red drum growth and survival.

Testing these complex hypotheses involves both an accurate description of the spatial arrangement of individual patches within a network and a reliable method to track movement. A major part of this habitat fragmentation study is mapping and remote sensing coupled with ground-truthing. Typically, fragmented habitats are assessed using some form of remote sensing like satellite imagery or aerial photography (Skole and Tucker 1993, Ihse 1995). Since seagrass beds show dynamic growth between seasons and remote sensing is often expensive, other approaches may be more practical or relevant. New technology such as the Trimble® GeoXT™ handheld from the GeoExplorer® 2008 series can provide detailed geo-referenced habitat coverage data. This is a highly accurate sub-meter GPS device which allows the user to record a position and creates shapefiles

(polygons) that can later be accessed in any GIS program. This technology allows accurate description of seagrasss systems in terms of percent cover, number of patches, total areas, perimeters, and nearest neighboring patch. These data can them be used to calculate an area-weighted mean perimeter to area ratio and the patch dispersion indices to quantify bed fragmentation as described by Sleeman et al. (2005). Together these can provide highly detailed patch network of seagrass meadows from continuous to highly fragmented areas. Data generated from these methods have the potential to provide quantifiable reference meadows to assess habitat fragmentation effect on estuarine nekton.

Because seagrass habitats support abundant and diverse communities of plant and animal life, and house 99% of our commercially harvested fishery species during at least one critical phase of their life cycle, understanding fragmentation in these systems is imperative. The ***overall goal*** of this project was to determine the response of nekton to variability in fragmentation of seagrass meadows within an estuarine complex. The ***rationale*** for the present research is that information on how fragmentation affects fisheries populations will allow resource managers to make more informed decisions on conserving and protecting this habitat type. Understanding the process of fragmentation and its effects on dynamic processes occurring in seagrasses will allow for a more comprehensive understanding of their ecological role. I address the impact of fragmented seagrass meadow on nekton using field sampling, and *in situ* mapping. In addition, I focus on red drum as a model estuarine dependent species by investigating densities, size distribution, growth, and movement. Specifically, the ***goals*** of this study were to: (1) identify fragmented seagrass beds within Corpus Christi and Aransas Bay; (2) describe

the effect of fragmentation on nekton diversity and abundance as a function of extent of fragmentation; (3) examine if growth rates of a seagrass-dependent fish, red drum, are influenced by varying fragmentation; (4) and observe fine scale movement of juvenile red drum within a fragmented seagrass system.

MATERIALS AND METHODS

Study Site

This study was designed to cover two bay systems long the northwestern Gulf of Mexico, Corpus Christi Bay and Aransas Bay. Both of these bays are good models systems for seagrasses on the entire Gulf Coast and were selected for that reason. Collectively, these areas compromise 497 km^2. Fresh water inflow is provided by Nueces River and Aransas rivers. The average salinity is 15 - 22 ppt and the average depth is 3.0 m (USEPA 1999). The system is separated from the Gulf of Mexico by both Mustang and Padre barrier islands. Major exchange with Gulf water occurs at the Aransas ship channel on the North end and Packery channel to the south (Fig 1).

The U.S. Environmental Protection Agency has recognized the Corpus Christi Bay System as a habitat of significant importance housing over 490 species of birds and 234 species of fish. Several species of seagrass occur within the bay including *Halodule Wrightii, Thalassia testudinum, and Syringodium filiforme*. The study sites include nine plots spanning approximately 4.5 km along the Eastern edge of Corpus Christi Bay located on the back side of Mustang Island. Mud Island (27.56^o N 97.01^o W) is located North of Redfish Bay within the Aransas Bay system, a 539 km^2 expanse composed of Aransas and Copano Bays. Mud Island spans 7 kilometers east to west and is separated from the Gulf of Mexico by St. Joes Island. The Aransas and Mission rivers provide

minimal freshwater inflow to the Aransas Bay system. Major exchange with the Gulf of Mexico occurs via the Aransas ship channel and Cedar Bayou (ephemeral pass). This locale is part of the Mission-Aransas National Estuarine Research Reserve (MANERR), a 751.8 km^2expanse including wetland, terrestrial, and marine environments. Within this system there is an estimated 85 km^2 of submerged aquatic vegetation. Study plots span a length of 3.82 km along the northern side of Mud Island.

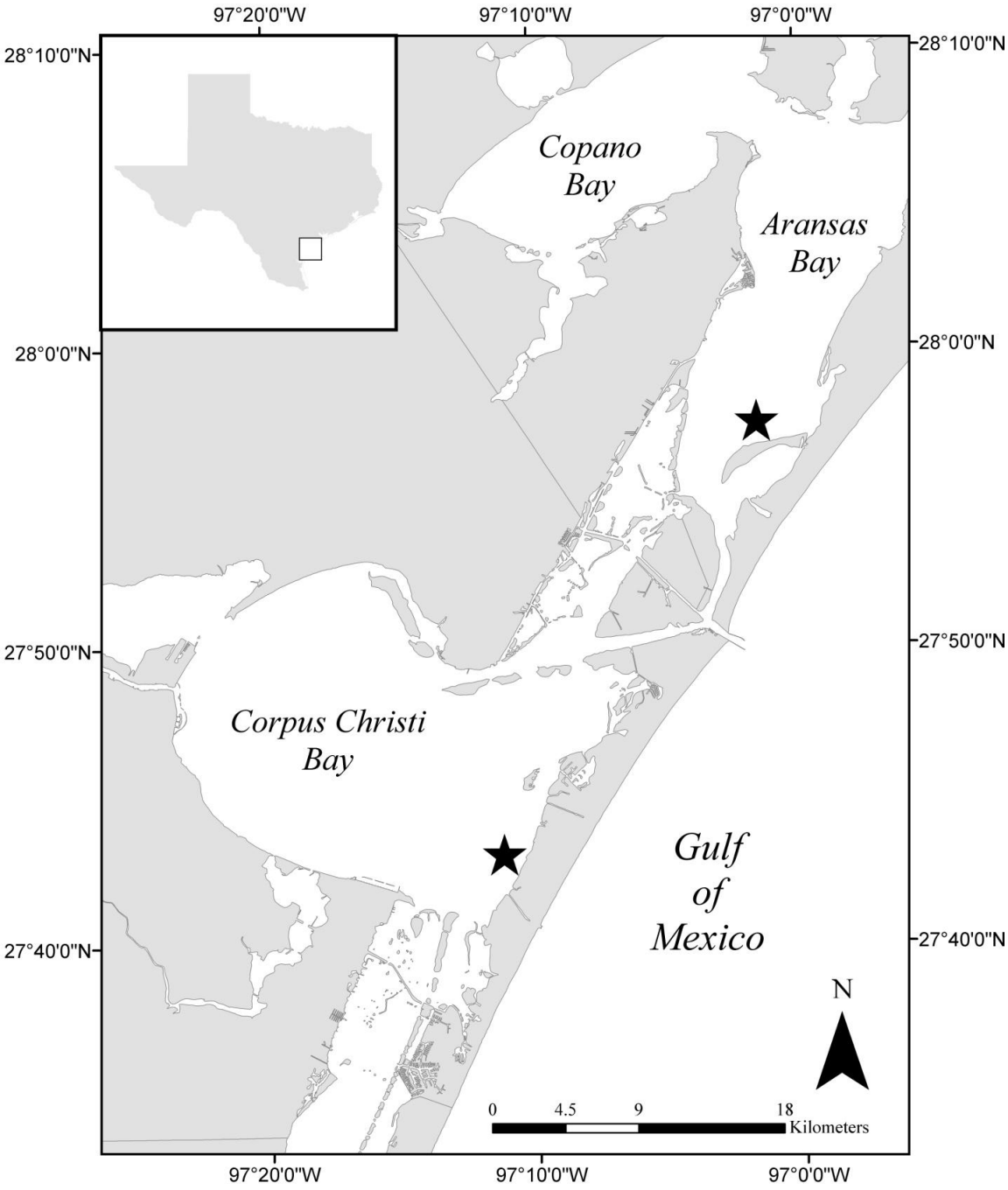
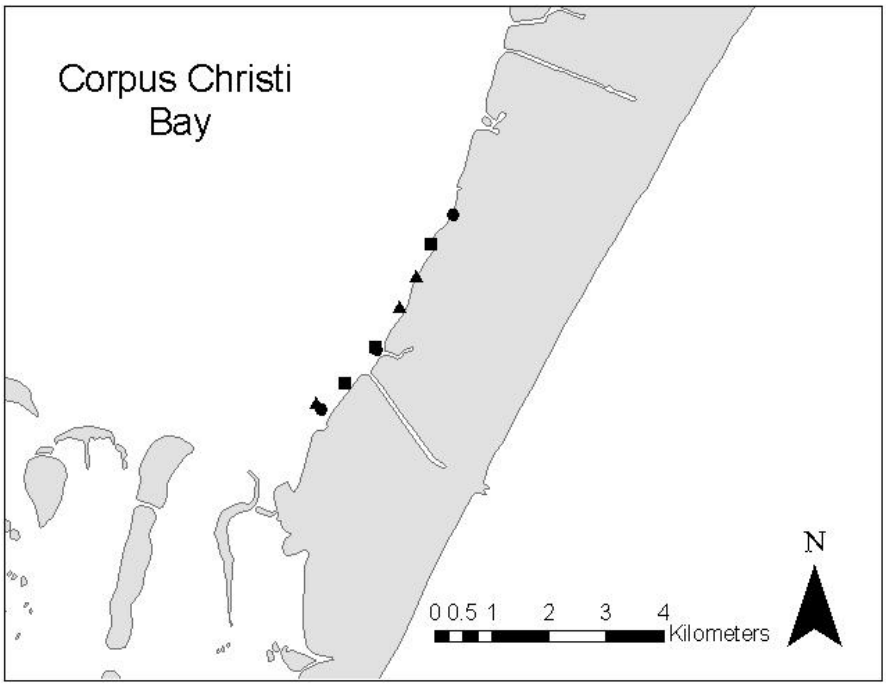


Figure 1. Map of study sites along the Texas coast. Stars represent location of fragmented study sites in Corpus Christi Bay and Aransas Bay.

Mapping

During spring and fall 2009, nine plots (50 x 100 m) of varying degrees of fragmentation were selected from both Corpus Christi Bay and Aransas Bay. Only monotypic beds of *Halodule wrightii* were selected for this study. Initially, preliminary study plots were selected based upon a visual approximation of cover. Care was taken in selecting plots composed of more than one seagrass patch. Study sites have both fragmented plots and continuous meadows within close proximity of each other and selected to control for differences in hydrodynamic conditions (Fig 2). A visual approximation of cover was made taking into account the size, number, and proximity of patches within each plot. A highly fragmented site was characterized by having small seagrass patches far apart ($> 2\ m^2$), while a medium fragmented site consisted of larger patches closer together ($< 2\ m^2$) (Fernandez et al. 2005). Low or no fragmentation was represented by a large continuous seagrass meadow ($\geq 4{,}000\ m^2$). Throughout this paper, Low Cover = High Fragmentation, Medium Cover = Medium Fragmentation, and Continuous Cover = No Fragmentation.

(A)



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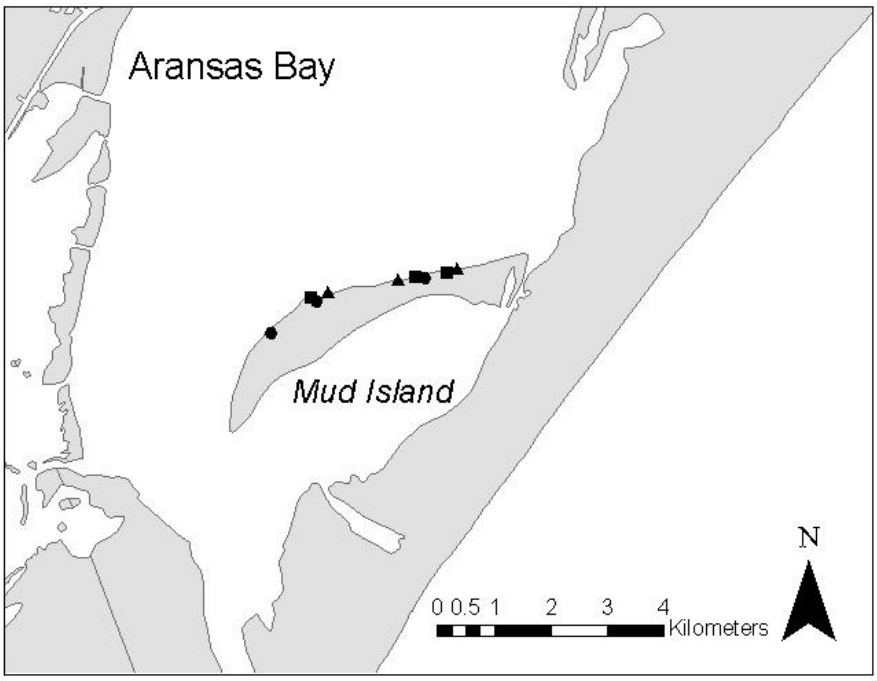


Figure 2. Map of Corpus Christi (A) and Mud Island (B) study plots; where (●) represents continuous seagrass bed (no fragmentation), (▲) low cover (high fragmentation) and, (■) medium cover plots (medium fragmentation).

To further refine the fragmentation level, each individual seagrass patch within every study plot was mapped using a Trimble® GeoXT™ handheld from the GeoExplorer® 2008 series. The Trimble is a highly accurate sub-meter GPS device which allows recording of geographic position and creation of shapefiles (polygons) that can be accessed in ArcMap software (Arc View, ESRI, Redmond, CA, USA). Mapping involved slowly walking the perimeter of each seagrass patch within study plot. Plots were mapped within a maximum of three weeks prior to sampling events to minimize change in cover and spatial arrangement of patches through growth and/or degradation. After shapefiles were imported into ArcMap the total area, area/perimeter ratio, and distance to nearest neighbor for each patch, as well as the total cover and number of patches were calculated for each study site. Because fragmentation involves the *breaking apart* and *loss of habit*, percent seagrass cover was used as an approximation for fragmentation *per se* (Robinson et al. 1995, Fernandez et al., 2005). For example, low cover represents high fragmentation, medium cover equals medium fragmentation, and continuous cover equals low or no fragmentation (Fig 3). Percent cover of seagrass was verified using ArcMap. Substrate was classified as either grass or bare substrate. Mean and standard error (SE) for percent cover was calculated for each fragmented plot within each bay. Within each bay three low cover plots (≤ 20% seagrass), three medium cover plots (25 - 55% seagrass), and three continuous plots (≥ 75% cover) were selected. Two way ANOVAs (SAS 9.2; Alpha = 0.05) were used to verify significant differences among varying covers in response to the main effects bay and season. A priori linear contrasts were used to test for significant differences among varying levels of cover.

(A)

(B)

(C)

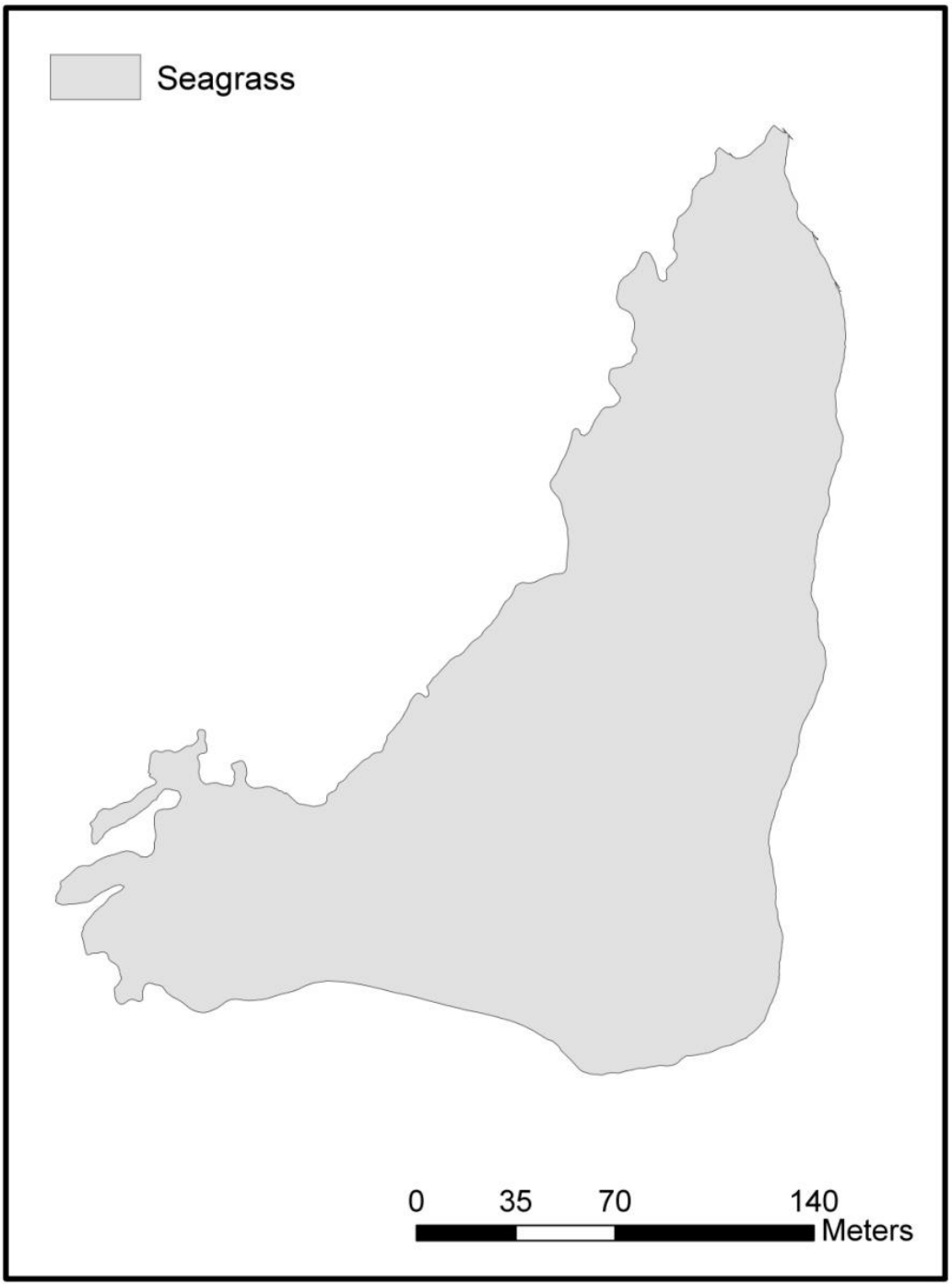


Figure 3. Examples of (A) low cover (high fragmentation), (B) medium cover (medium fragmentation), and (C) continuous cover (no fragmentation) seagrass plots mapped using a GeoXT Trimble coupled with ArcMap software.

Nekton density

Nekton sampling took place during two seasons: late spring and early fall of 2009. During each season, plots were mapped and then sampled twice with one week between sampling events. All samples were taken from a 50x50 meter grid within the larger 50x100 meter study plots. Patches were sampled using an epibenthic sled. This sampling device is very efficient and has been use effectively by numerous investigators (see Stunz et al 2002 for detailed description). Briefly, the sled includes a metal frame opening of 0.6 m (length) x 0.75 m (height) equipped with a 1-mm mesh conical plankton net and a 17 meter tow line. The sled was placed on the ground and a semi-circular path was walked to avoid immediate disturbance. In medium and low cover plots, at least three separate patches were sampled. A minimal fifty-one linear meters (30 m^2) total grass was sampled at each plot. The tows avoided bare substrate. Each tow length and position was mapped using the GEOXT Trimble unit, allowing for patch identification and total nekton density (m^{-2}) (Fig 4). There were two sampling events within each season, yielding a total of 4 events. During each sampling event dissolved oxygen (DO) and temperature was measured using a YSI model DO 200. Salinity was measured using a refractometer. Depth readings were taken three times along each epibenthic tow using a meter stick and tide chart snapshots were saved for each sampling date.

All samples collected were rough sorted in the field and preserved using 10% formalin for later identification in the laboratory. After returning to the lab, all species were removed, identified to species, and measured to the nearest 0.1 mm. I specifically focused on nekton densities; however, other investigators at the University of Texas Marine Science Institute analyzed the nekton response to fragmentation in greater detail using a variety of community analyses (Hensgen 2011).

Spring and fall seasons were analyzed separately using a two-way ANOVA (CI = 95%) to determine if there were differences in density (m^{-2}) in response to cover (low, medium, and continuous) and bay. Data were transformed (log10[x+1]) to ensure homogeneity of variance and normality of residuals. A priori linear contrasts were performed to test for significant differences in nekton densities among different cover plots (Low, Medium, Continuous) within separate bays (Alpha = 0.05). In addition to total nekton, crustacean and fish densities were analyzed separately using the same model as mentioned above.

Figure 4. Photograph of the epibenthic sled being towed through a small seagrass patch within a highly fragmented network. The sled's path is then recorded using a Trimble.

Red drum density and standard length

Nekton data from fall 2009 was used to compare densities and standard length (SL) of newly settled red drum among different levels of cover within Corpus Christi Bay and Aransas Bay. According to Holt (1983), red drum are thought to settle into seagrass habitat between 6 and 8 mm standard length (SL), for this reason fish ≥ 6mm standard length were used for this analyses. A two-way ANOVA used for total nekton density was also used red drum densities where bay and cover were fixed factors. A priori linear contrasts were used to test for any significant differences in red drum density and SL among different levels of cover.

Red drum growth

Growth rates for newly settled red drum were analyzed using two methods, RNA:DNA ratios and otolith microstructure. During peak red drum recruitment in November 2009, collections for red drum were done at each level of cover (low, medium, and continuous) from both Aransas and Corpus Christi Bays using a bag seine (6 m long with 5-mm mesh wings and a 3-mm mesh bag). All patches within all eighteen plots were sampled at least twice. The total number of red drum collected for growth analyses in this study was 100 fish. Sixty-four red drum were collected from Corpus Christi Bay and 36 were collected form Aransas Bay. Only 3 fish from Aransas Bay were collected from fragmented habitats. Because enough replicates were not found within Aransas Bay, growth analysis was only performed on the 64 red drum collected from Corpus Christi Bay.

In the field, juvenile red drum were measured to the nearest 0.1mm SL, and then cut in half. The head was preserved in 70% ethanol and later used for otolith analyses

while the body was preserved on dry ice and later used in RNA:DNA analyses. Care was taken in dividing the fish in such a way that excluded gut contents from the part of the fish used for RNA:DNA analyses as this would disrupt results. Fish heads remained in 70% ethanol until all three sets of otoliths were removed. The body of the fish was kept in a -80 degree freezer until RNA:DNA analysis could be performed.

RNA:DNA

RNA/DNA ratios analyses were performed by the mariculture lab in Port Aransas using ethidium-bromide flourometric techniques (Westerman and Holt 1988, Caldarone et al. 2006). Ethidium-bromide is the most commonly used stain for detecting DNA/RNA. It works by inserting itself between base pairs in a double helix. Ethidium-bromide is UV absorbent and can be read with a spectrophotometer at wavelengths between 300 and 360 nm. Red drum bodies were homogenized in a cold buffer and duplicate portions of supernatant were added to a micoplate containing Tris EDTA buffer, ionic cofactors, flourophore EB, and proteinase K (10% wt:vol). RNA concentrations were calculated by measuring fluorescence of each sample before and after RNase was added to each well. Similarly, DNA concentrations were calculated by measuring the difference in fluorescence after DNase was added to each sample. Calculations are based on comparisons to known calibration curves using calf thymus DNA and yeast RNA standards.

RNA:DNA ratios were compared between habitats integrated over the fish's life. An ANCOVA model was used to test for the assumption of no significant interaction between the treatment (fragmentation) and the effect of the covariant (age) on the dependent variable (RNA:DNA ratio) (Alpha = 0.05). ANCOVA tested for differences in

y intercepts and if no significant interaction was found then the model was re-rerun without the interaction.

Otolith Analysis

All otoliths were removed under a Leica S4E dissecting microscope. Only lapillar otoliths were used for analysis because they are the smallest and logistically the easiest with which to work. Smaller otoliths allow for easier analyses because no cross sectioning or additional preparation such as polishing are needed. Once the lapilli were retrieved from their cavities the head was removed and all bits of skin and discharge were removed from the slide. Probes were used to carefully remove excess skin from the otoliths. A series of water drops from the corner to the center of the slide were used to move the otoliths without breakage or excessive movement. Otoliths were dried, and one drop of Flo-Texx® was added to the top of the otoliths. A probe was used to orientate all otoliths concave up. Prepared slides were left to harden for twenty four hours before analyses.

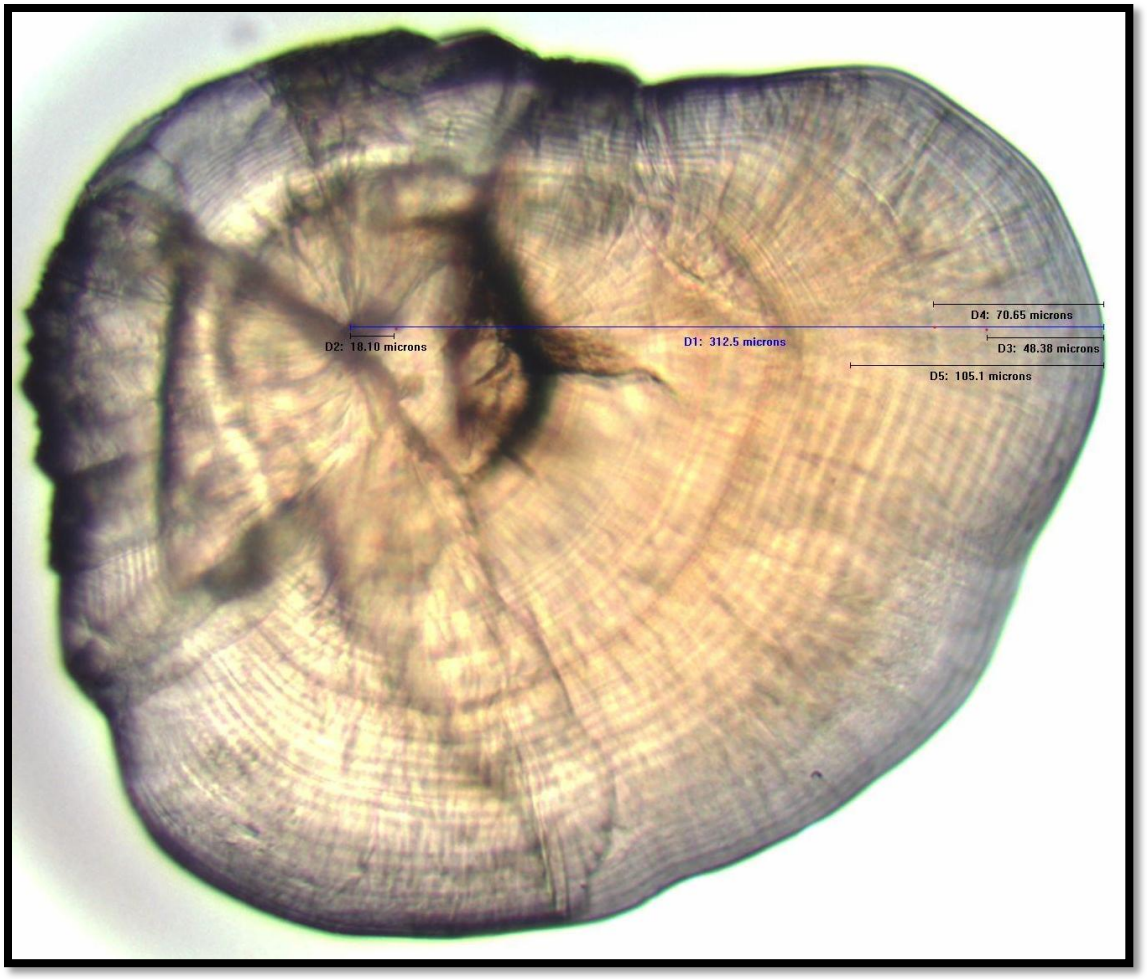
All otoliths were analyzed using an Olympus CX41 compound microscope under 40X magnification. Higher magnifications blur the detail of individual rings. Left and right asterisci and lapilli were removed for age determination based upon ring count and otolith microstructure (David et al. 1994, Rooker 1997, Stunz 2002). Left and right lapilli were photographed using an Olympus QColor-3 camera and the Qcapture© program. Left and right otolith radii were measured to the nearest 0.01 micron. All measurements were made from the inner primordium (origin) to the longest edge of the otolith. A linear relationship was established between fish SL and otolith radii. Ring counts were performed on left and right lapilli using the image program GIMP 2.6.2©. Multiple

photographs of otoliths and counts were saved for future query. Often, the inner core rings were difficult to read, in which case a regression established by Rooker (1998) was used to supplement inner ring counts. A measurement was taken from the primordium to the inner most observable ring and put into the equation (Age in days = 34.46 + 15.94 log (Radius mm), thereby completing an age count. Counting was performed once for each left and right otolith and an average age was taken between the two.

Growth rates for individual fish have been based upon incremental widths of individual rings (Hovenkamp 1991, Stunz et al 2002). I compared growth rates between fragmented and non-fragmented habitats integrated over the life of the fish using analysis of covariance (ANCOVA). A regression model was used to test for the assumption of no interaction between the treatment (Fragmentation) and the effect of the covariate (Age) on the dependent variable (Fish Length). ANCOVA tested for differences in the *y* intercepts.

In addition, outer rings were used to establish recent growth rates (Stunz et al 2002). Measurements of the outer seven, ten, and fourteen rings were used to derive a daily incremental growth rate (μd^{-1}) (Fig 5). This assumes that the fish captured had been utilizing the habitat for the past two weeks. A one way ANOVA was used to test for significant differences in growth rates between fragmented and non-fragmented habitats for the last seven, ten, and fourteen days.

(A)



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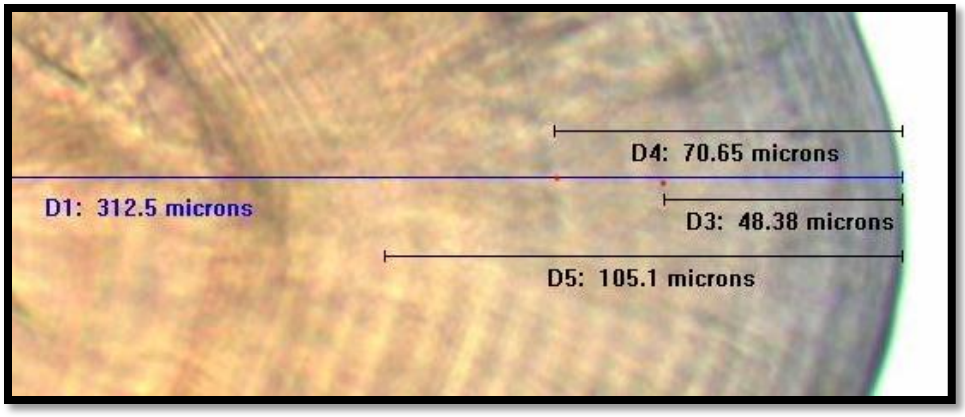


Figure 5. (A) Micrographs of lapillar otolith (40x) with multiple measurements taken from otolith primordium. Measurements include total radius, inner unknown ring radius, and outer 7, 10, and 14 rings. (B) Zoom-in on measurements of radius (blue) and widths of outer 7, 10, and 14 rings.

Red drum movement

In fall 2009, juvenile red drum were collected from Aransas Bay using a 6-m bag seine long with 5-mm mesh. Two hundred twenty-nine red drum were captured and returned to the laboratory and allowed to acclimate for 24 hours. Fish were then injected with a VIE tag along the back midline (see Bushon et al 2007; Fig 6). Once tagged, fish were observed for an additional 24 hours. Of the 229 fish, 200 were selected for the movement experiment.

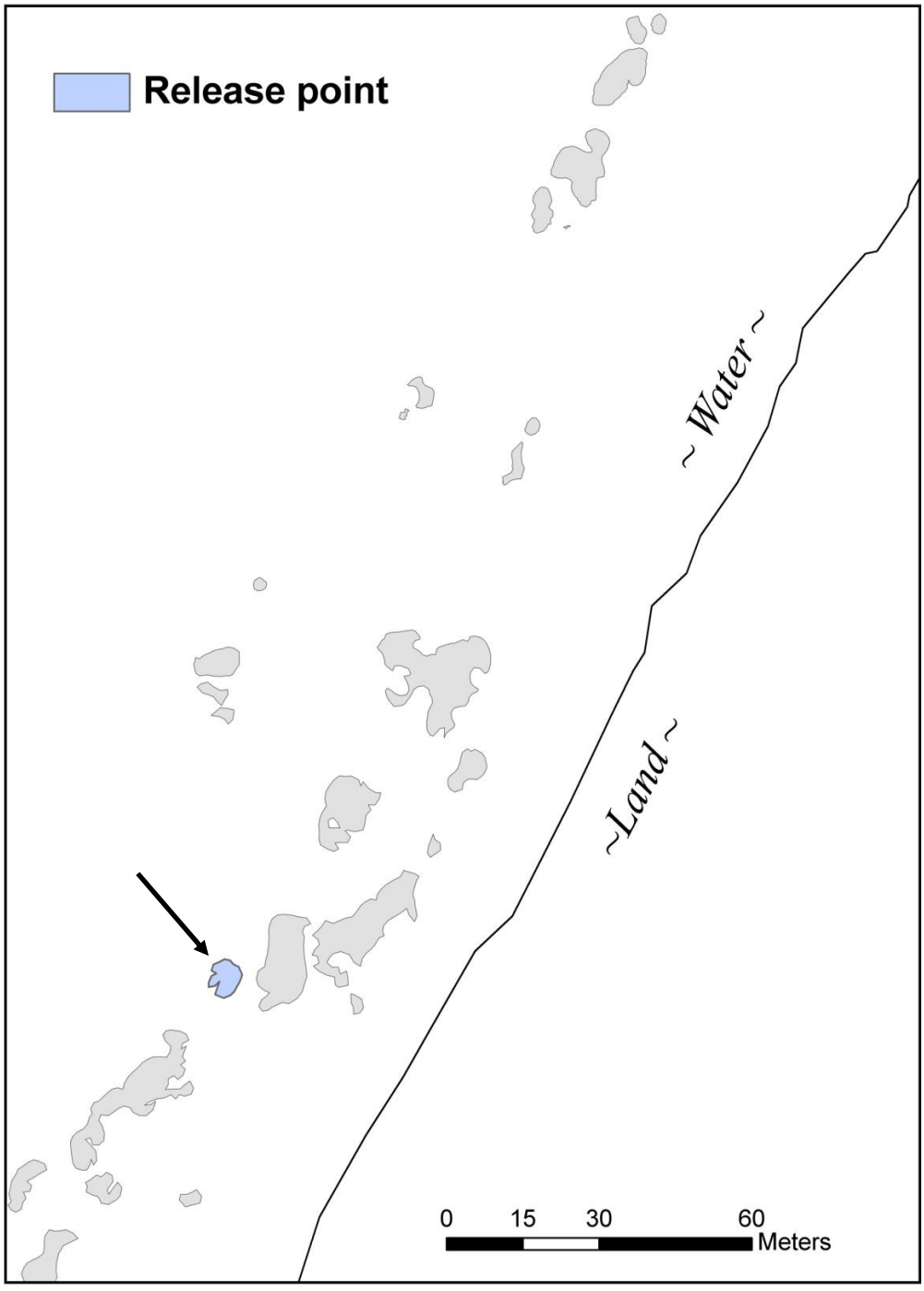
On November 26th, 2009, two hundred marked red drum were released into three previously mapped, highly fragmented patch networks within Corpus Christi Bay at natural densities of (1.5 red drum m^{-2}) (Fig 7). Recapturing sampling occurred 24, 48, and 72 hours after release date. Each patch within each network was sampled using the same bag seine described above. A bag seine (6 m wide, 5-mm mesh wings, and 3-mm mesh bag) was pulled over every patch a minimum of two times or until no additional red drum were captured. All other nekton were quickly and carefully placed back into the sampled patch. If a marked fish was recaptured it was measured, recorded, and placed back into the patch where it was found. Location of recaptured fish was recorded and a total distance travelled from start point was measured using ArcMap. Any wild red drum caught during recapture were recorded and released back into the seagrass.

(A)

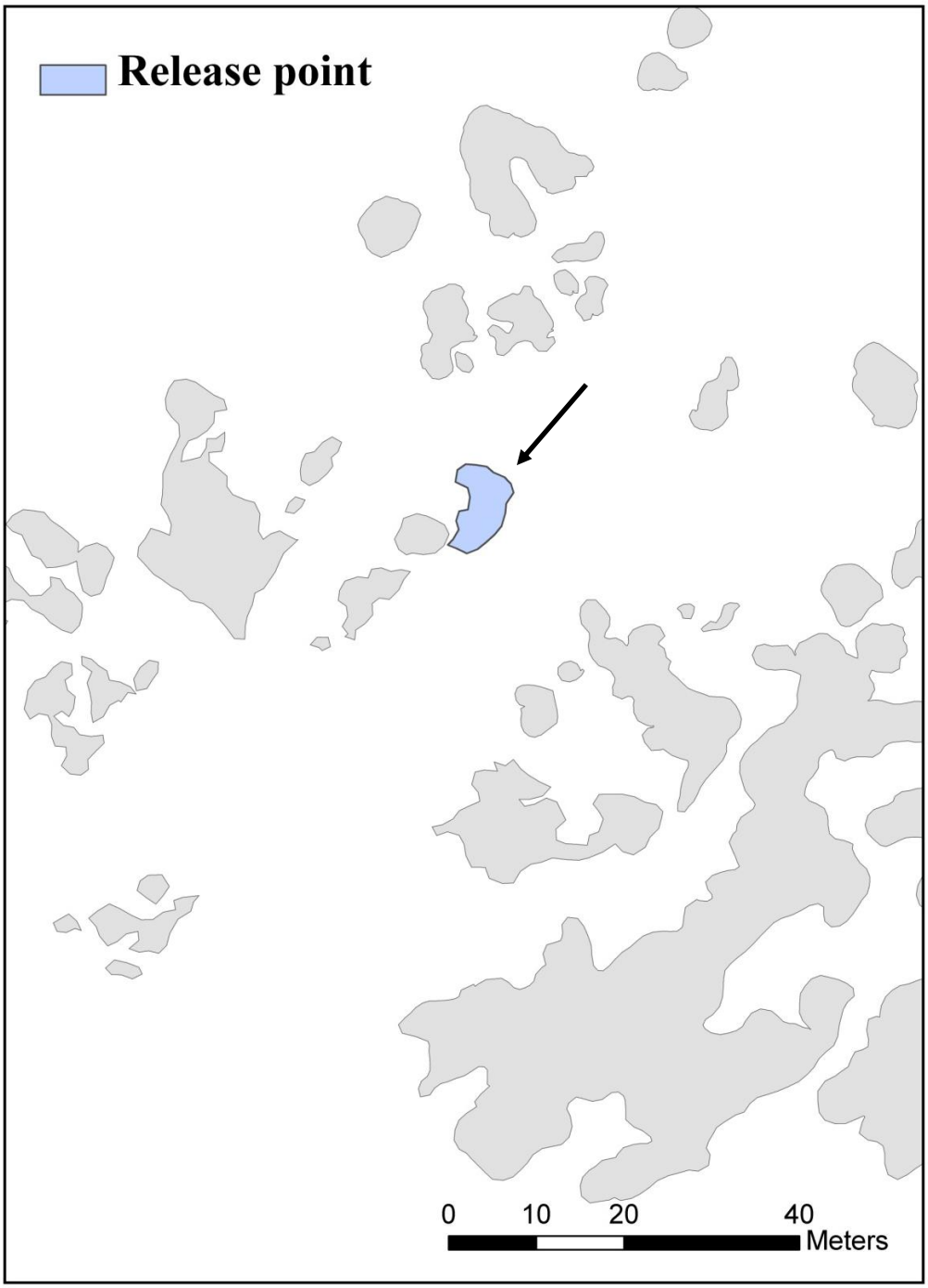
(B)

Figure 6. *Sciaenops ocellatus*. (A) Red drum marked with green Visible Implant Elastomer (VIE) tag along dorsal midline; (B) several marked red drum ready for deployment (B).

(A)



(B)



(C)

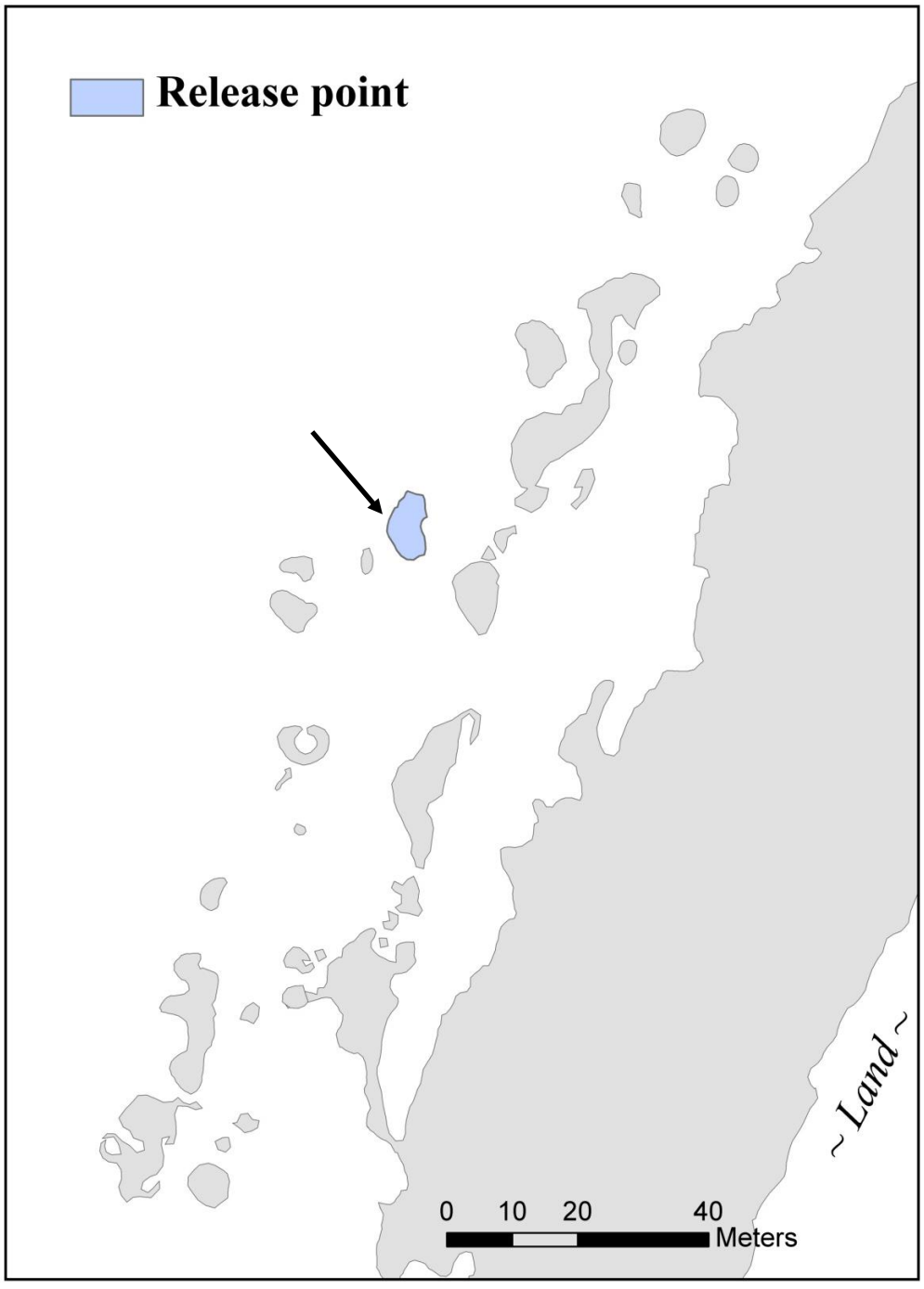


Figure 7. Maps of movement study sites (A) 1, (B) 2, and (C) 3; highly fragmented seagrass networks located in Corpus Christi Bay. Marked red drum were released into a single patch within each network on November 26th, 2009. Arrows indicates release point.

RESULTS

Site Characteristics

During each sampling event, temperature (temp), salinity (sal), and dissolved oxygen (do) were measured. Dissolved oxygen and salinity were comparable during spring and fall. Water temperature was higher during spring (30.9 C^0 ± 0.3 SE) than in fall (22.7 C^0 ± 0.3 SE) (Table 1A). Temperature, dissolved oxygen, and salinity values were comparable among different cover levels within Corpus Christi Bay and Aransas Bay (Table 1B).

Table 1. Mean environmental parameters (standard error, SE) for season (A) and three levels of seagrass cover (Low, Medium, Continuous) (B) in both Corpus Christi Bay and Aransas Bay collected during spring and fall seasons 2009. Mean and SE were calculated from measurements taken at each site during each sampling event (n=4).

(A)

	Spring		Fall	
Parameter	Mean	SE	Mean	SE
Water Temperature (°C)	30.9	(0.3)	22.7	(0.3)
Dissolved Oxygen (mg l^{-1})	7.5	(0.3)	7.1	(0.2)
Salinity (ppt)	39.5	(0.5)	41.4	(0.2)

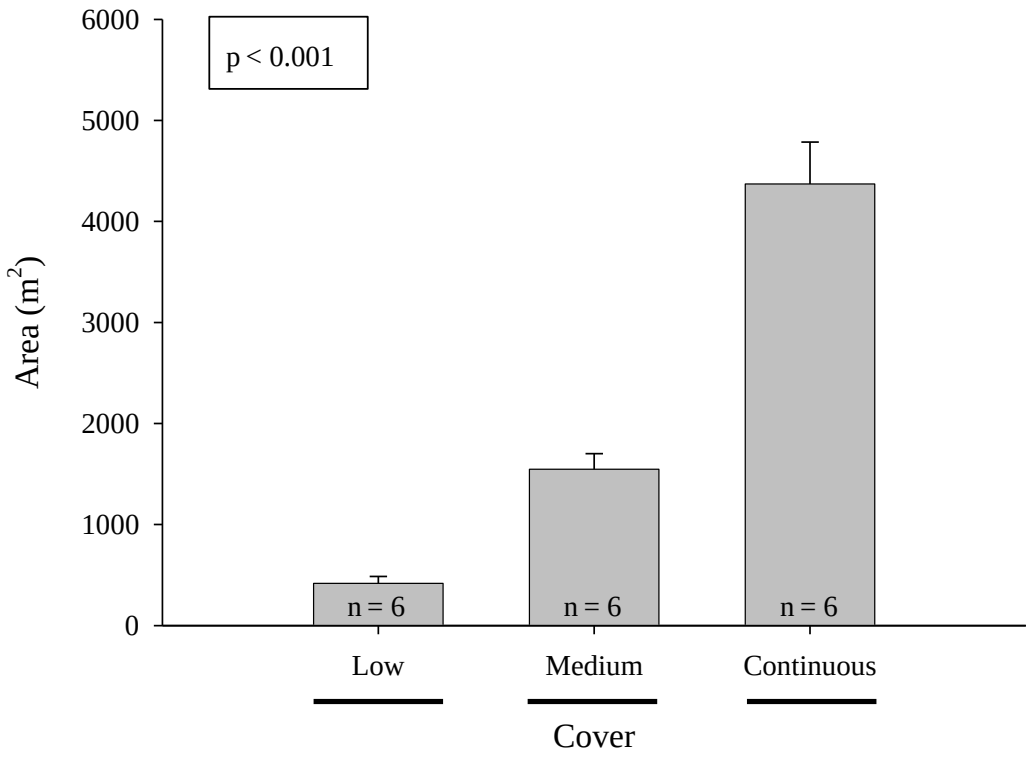
(B)

	Corpus Christi Bay						Aransas Bay					
	Low		Medium		Continuous		Low		Medium		Continuous	
Parameter	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Spring												
Water Temperature (°C)	30.5	(0.8)	29.7	(0.6)	30.2	(1.0)	31.3	(0.5)	31.5	(0.7)	32.4	(0.9)
Dissolved Oxygen (mg l^{-1}	7.2	(0.5)	6.9	(0.5)	7.5	(0.9)	7.4	(0.5)	7.5	(0.6)	8.7	(1.0)
Salinity (ppt)	42.3	(1.1)	41.0	(0.7)	41.8	(0.9)	37.5	(0.8)	37.7	(0.8)	36.8	(0.4)
Fall												
Water Temperature (°C)	22.8	(0.6)	22.5	(0.6)	22.5	(0.6)	22.3	(0.9)	22.6	(1.0)	23.4	(1.2)
Dissolved Oxygen (mg l^{-1}	6.8	(0.2)	6.8	(0.4)	7.0	(0.5)	6.5	(0.4)	7.2	(0.5)	8.5	(0.7)
Salinity (ppt)	42.0	(0.4)	41.7	(0.3)	42.5	(0.6)	40.5	(0.4)	41.0	(0.6)	40.8	(0.3)

Mapping

Using the handheld Trimble GeoXT unit allowed for highly accurate accounts of percent cover within study plots during each season. Total seagrass among three levels of cover were significantly different for spring ($F_{2,12} = 98.64$; $p < 0.001$) and fall ($F_{2,13} = 95.79$; $p < 0.001$). *A priori* linear contrasts indicated that low (417.65 m^{-2} ± 68.97, 945.15 m^{-2} ± 113.65), medium (1547.5 m^{-2} ± 153.63, 2719.91 m^{-2} ± 245.58), and continuous (4371.17 m^{-2} ± 414.25, 4776.17 m^{-2} ± 146.79) cover plots were significantly different from each other during spring and fall, respectively (Fig 8).

(A)



(B)

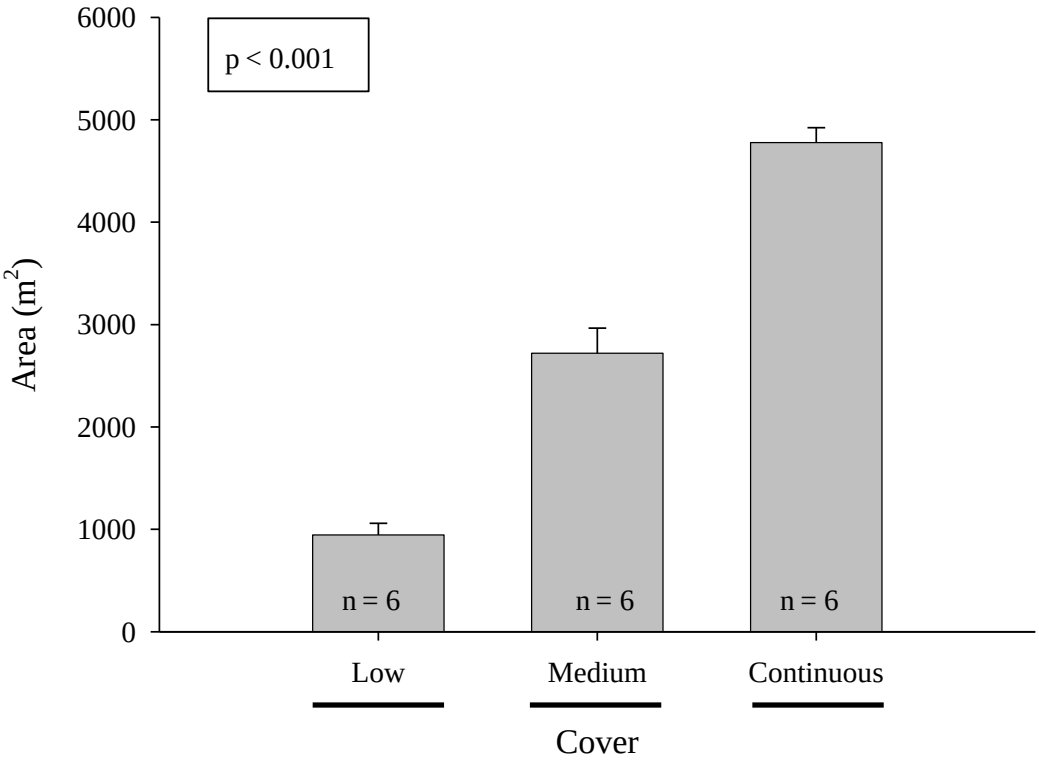


Figure 8. Total seagrass cover (m^2 ± SE) among three levels of cover during (A) spring and (B) fall 2009 for study sites selected in Corpus Christi Bay and Aransas Bay. Horizontal bars represent amount of seagrass within 50x100 meter study plot. All levels of cover are significantly different from each other.

Nekton density

A total of 317,841 organisms (crustaceans = 312,877; fish = 4,964) were collected during spring and 117,863 organisms (crustaceans = 116,533; fish = 1,330) were collected during fall. Crustaceans, such as arrow shrimp, *Hippolytes* sp., and American prawn represented 96.58% and 97.23% relative abundance for spring and fall, respectively. There was a significant difference in total nekton density (m^{-2}) among cover levels during spring (F= 11.88; df= 2, 29; p= 0.0002) and fall (F= 3.96; df= 2, 29; p=

0.0301) (Table 2). *A priori* linear contrasts were used to test for significant differences in nekton density among cover levels. In each season, nekton densities in continuous cover differed significantly from medium and low cover. Specifically, spring nekton was higher in medium (349.82 m^{-2} ± 56.75) and low (377.65 m^{-2} ± 48.75) cover plots compared to continuous (137.20 m^{-2} ± 22.87) plots; while fall medium (112.10 m^{-2} ± 18.68) and low (125.45 m^{-2} ± 16.62) cover plots were significantly lower than continuous (165.52 m^{-2} ± 28.59) cover plots (Fig 9, Table 3).

Table 2. Analysis of variance for nekton density (m^{-2}) among three levels of cover (Low, Medium, Continuous) for Aransas Bay and Corpus Christi Bay during spring (A) and fall (B) seasons. Event was blocked for all models to control for any temporal variability. A priori linear contrast was used to test for significant difference in densities among cover levels (Alpha = 0.05). Level of significance is represented by number of *, where (*) represents significance, (**) represents very significant, and (***) represents highly significant.

A.

SOURCE	*df*	Sum of Squares	Mean Square	F Value	P Value	
SPRING MODEL						
Bay	1	0.074	0.074	0.26	0.6111	
Cover	2	7.034	3.517	12.58	0.0001	***
Bay x Cover	2	1.404	0.702	2.51	0.0987	
Event	1	0.104	0.104	0.37	0.5477	
Residual	29	8.111	0.280			
LINEAR CONTRAST						
Low vs. Continuous	1	6.121	6.121	21.89	<.0001	***
Low vs. Medium	1	0.168	0.168	0.60	0.4400	
Medium vs. Continuous	1	4.262	4.262	15.24	0.0005	***

B.

SOURCE	*df*	Sum of Squares	Mean Square	F Value	P Value	
FALL MODEL						
Bay	1	4.935	4.935	13.05	0.0011	**
Cover	2	2.962	1.481	3.92	0.0312	*
Bay x Cover	2	0.694	0.347	0.92	0.4105	
Event	1	1.757	1.757	4.65	0.0395	*
Residual	29	10.965	0.378			
LINEAR CONTRAST						
Low vs. Continuous	1	2.203	2.203	5.83	0.0223	*
Low vs. Medium	1	0.000	0.000	0.00	0.9840	
Medium vs. Continuous	1	2.240	2.240	5.92	0.0213	*

Crustacean density

I used the nekton data set to look only at crustacean densities in response to varying cover levels, during spring and fall seasons. There was a significant difference in crustacean density among cover levels during both spring ($F_{2,29} = 12.58$; p= 0.0002) and fall ($F_{2,29} = 3.92$; p= 0.0312). *A priori* linear contrasts yielded similar results to total nekton densities for spring and fall. Spring medium (333.92 $m^{-2} \pm 53.22$) and low (360.13 $m^{-2} \pm 38.37$) cover plots were significantly higher than continuous (130.97 $m^{-2} \pm 17.51$) cover plots while fall medium (87.67 $m^{-2} \pm 17.18$) and low (88.09 $m^{-2} \pm 18.53$) cover plots were significantly lower than continuous (146.50 $m^{-2} \pm 22.47$) plots (Fig 10). These results mimic nekton densities and suggest that crustaceans may be driving the observed trend in nekton densities.

Fish density

Similar to my analyses of crustacean densities, I investigated the response of fish densities to fragmentation using the nekton data collected during spring and fall seasons. A simple main effects (SME) analysis was performed on spring fish densities (m^{-2}) due to a significant interaction between bay and cover ($F_{2,29} = 6.32$; p= 0.0053). SME results indicate fish densities differ significantly among three levels of seagrass cover ($F_{5,29} = 5.90$; p = 0.0007). Specifically, continuous (6.04 $m^{-2} \pm 1.14$) cover plots had higher fish densities than medium (3.21 $m^{-2} \pm 0.56$) cover plots. Fish densities in low (3.87 $m^{-2} \pm$ 0.70 SE) cover plots did not differ significantly from medium or continuous cover plots. Fall fish densities were significantly different among fragmentation levels ($F_{2,29} = 5.85$; p= 0.0073). *A priori* linear contrasts for fall fish densities indicate continuous (1.71 $m^{-2} \pm$ 0.22) cover plots had higher densities than both medium (0.94 $m^{-2} \pm 0.18$) and low (1.01

m^{-2} ± 0.26) cover plots (Fig 11). A similar trend of higher densities of fish being found in continuous habitats can be observed for spring and fall seasons.

Table 3. Overall mean densities (m^{-2}) and standard error (SE) of organisms collected from three levels of cover (low, medium, continuous) in Corpus Christi Bay and Aransas Bay during spring and fall 2009. Relative abundance (number of individuals/total number of organisms x 100) is also provided.

				Low Cover		Medium Cover		Continuous Cover	
COMMON NAME	SCIENTIFIC NAME	TOTAL NUMBER	RELATIVE ABUNDANCE (%)	MEAN	SE	MEAN	SE	MEAN	SE
SPRING 2009		317,841							
Total Fishes		4,964	1.56	16.75	(3.05)	10.35	(1.96)	18.13	(3.43)
Darter Goby	*Gobionellus boleosoma*	1,803	0.57	3.90	(0.51)	2.30	(0.40)	9.68	(3.55)
Code goby	*Gobiosoma robustum*	835	0.26	1.98	(0.59)	2.04	(0.73)	3.40	(1.03)
Spotfin mojarra	*Eucinostomus argenteus*	638	0.20	3.29	(2.90)	0.45	(0.20)	2.12	(0.97)
Gulf pipefish	*Syngnathus scovelli*	516	0.16	2.60	(0.64)	1.40	(0.30)	1.12	(0.26)
Dusky pipefish	*Syngnathus floridae*	382	0.12	1.52	(0.35)	1.83	(0.50)	0.20	(0.08)
Silver perch	*Bairdiella chrysoura*	292	0.09	1.18	(0.38)	1.07	(0.69)	0.31	(0.07)
Spotted seatrout	*Cynoscion nebulosus*	139	0.04	0.78	(0.25)	0.45	(0.13)	0.20	(0.08)
Pinfish	*Lagodon rhomboides*	73	0.02	0.14	(0.04)	0.18	(0.04)	0.30	(0.09)
Naked goby	*Gobiosoma bosc*	69	0.02	0.10	(0.05)	0.09	(0.03)	0.42	(0.21)
Dwarf seahorse	*Hippocampus zosterae*	52	0.02	0.19	(0.09)	0.18	(0.09)	0.14	(0.07)
Blackcheek tonguefish	*Symphurus plagiusa*	48	0.02	0.42	(0.13)	0.08	(0.04)	0.05	(0.02)
Chain pipefish	*Syngnathus louisianae*	32	0.01	0.24	(0.08)	0.09	(0.04)	0.03	(0.02)
Pigfish	*Orthopristis chrysoptera*	17	0.01	0.10	(0.03)	0.02	(0.01)	0.02	(0.01)
Bay anchovy	*Anchoa mitcheli*	15	0.00	0.15	(0.10)	0.00	(0.00)	0.00	(0.00)
Striped blenny	*Chasmodes bosquianus*	11	0.00	0.04	(0.03)	0.03	(0.02)	0.03	(0.01)
Feather blenny	*Hypsoblennius hentz*	11	0.00	0.04	(0.02)	0.02	(0.01)	0.02	(0.02)
Gulf toadfish	*Opsanus beta*	5	0.00	0.01	(0.01)	0.01	(0.01)	0.03	(0.01)
Striped Burrfish	*Chilomycterus shoepfi*	4	0.00	0.01	(0.01)	0.02	(0.02)	0.01	(0.01)
Shrimp eel	*Ophichthus gomesii*	4	0.00	0.00	(0.00)	0.01	(0.01)	0.02	(0.02)
Skilletfish	*Gobiesox strumosus*	3	0.00	0.01	(0.01)	0.02	(0.01)	0.00	(0.00)

Table 3. (Continued)

COMMON NAME	SCIENTIFIC NAME	TOTAL NUMBER	RELATIVE ABUNDANCE (%)	Low Cover		Medium Cover		Continuous Cover	
				MEAN	SE	MEAN	SE	MEAN	SE
Inshore lizardfish	*Synodus foetens*	3	0.00	0.02	(0.01)	0.01	(0.01)	0.00	(0.00)
Sheepshead	*Archosargus probatocephalus*	2	0.00	0.00	(0.00)	0.00	(0.00)	0.02	(0.01)
Spot	*Leiostomus xanthurus*	2	0.00	0.01	(0.01)	0.01	(0.01)	0.00	(0.00)
Hogchocker	*Trinectes maculatus*	2	0.00	0.00	(0.00)	0.02	(0.01)	0.00	(0.00)
Diamond killifish	*Adinia xenica*	1	0.00	0.00	(0.00)	0.00	(0.00)	0.01	(0.01)
Frillfin goby	*Bathygobius soporator*	1	0.00	0.00	(0.00)	0.01	(0.01)	0.00	(0.00)
Spadefish	*Chaetodipterus faber*	1	0.00	0.01	(0.01)	0.00	(0.00)	0.00	(0.00)
Bay whiff	*Citharichthys spilopterus*	1	0.00	0.00	(0.00)	0.01	(0.01)	0.00	(0.00)
Longnose killifish	*Fundulus similis*	1	0.00	0.00	(0.00)	0.00	(0.00)	0.01	(0.01)
Bluefish	*Pomatomus saltatrix*	1	0.00	0.01	(0.01)	0.00	(0.00)	0.00	(0.00)
Total Crustaceans		312,877	98.44	1655.10	(276.92)	1091.34	(216.01)	382.52	(52.11)
Arrow shrimp	*Tozeuma carolinense*	224,501	70.63	1374.91	(249.06)	894.34	(208.00)	74.39	(33.51)
Hippolytes Pleurocantha	*Hippolytes Pleurocantha*	57,107	17.97	208.16	(42.28)	133.11	(12.96)	171.30	(32.70)
American Prawn	*Palaemonetes vulgaris*	25,372	7.98	55.85	(11.42)	49.65	(6.56)	114.81	(17.75)
Brown / Pink shrimp	*Farfantepenaeus sp.*	5,134	1.62	13.95	(2.12)	12.61	(1.36)	19.10	(3.02)
Blue crab	*Callinectus sapidus*	595	0.19	1.56	(0.44)	1.15	(0.34)	2.57	(0.92)
White shrimp	*Litopenaeus setiferus*	106	0.03	0.52	(0.15)	0.27	(0.10)	0.19	(0.08)
Snapping shrimp	*Alpheus heterochaelis*	18	0.01	0.02	(0.01)	0.02	(0.01)	0.11	(0.04)
Sargassum Shrimp	*Latreutes parvulus*	17	0.01	0.03	(0.03)	0.11	(0.11)	0.00	(0.00)
Longnose spider crab	*Libinia dubia*	16	0.01	0.02	(0.01)	0.07	(0.03)	0.05	(0.03)
Combclaw shrimp	*Leptochela serratorbita*	11	0.00	0.09	(0.05)	0.01	(0.01)	0.01	(0.01)

Table 3. (Continued)

COMMON NAME	SCIENTIFIC NAME	TOTAL NUMBER	RELATIVE ABUNDANCE (%)	Low Cover		Medium Cover		Continuous Cover	
				MEAN	SE	MEAN	SE	MEAN	SE
FALL 2009		117,863							
Total Fishes		1,330	1.13	4.75	(1.20)	2.83	(0.55)	5.13	(0.67)
Gulf pipefish	*Syngnathus scovelli*	385	0.33	1.27	(0.41)	0.78	(0.17)	1.67	(0.31)
Darter Goby	*Gobionellus boleosoma*	222	0.19	0.34	(0.14)	0.27	(0.07)	1.37	(0.34)
Red drum	*Sciaenops occelatus*	191	0.16	1.21	(0.53)	0.43	(0.26)	0.33	(0.10)
Dusky pipefish	*Syngnathus floridae*	170	0.14	0.52	(0.17)	0.51	(0.18)	0.56	(0.17)
Spotted seatrout	*Cynoscion nebulosus*	156	0.13	1.01	(0.41)	0.37	(0.17)	0.23	(0.10)
Code goby	*Gobiosoma robustum*	117	0.10	0.09	(0.06)	0.14	(0.04)	0.75	(0.15)
Bay anchovy	*Anchoa mitcheli*	18	0.02	0.00	(0.00)	0.15	(0.13)	0.00	(0.00)
Blackcheek tonguefish	*Symphurus plagiusa*	15	0.01	0.07	(0.03)	0.06	(0.03)	0.02	(0.01)
Silver perch	*Bairdiella chrysoura*	13	0.01	0.13	(0.09)	0.04	(0.02)	0.02	(0.01)
Feather blenny	*Hypsoblennius hentz*	12	0.01	0.06	(0.02)	0.03	(0.02)	0.02	(0.01)
Spotfin mojarra	*Eucinostomus argenteus*	11	0.01	0.02	(0.01)	0.02	(0.02)	0.05	(0.04)
Dwarf seahorse	*Hippocampus zosterae*	8	0.01	0.00	(0.00)	0.02	(0.01)	0.05	(0.03)
Pinfish	*Lagodon rhomboides*	3	0.00	0.01	(0.01)	0.00	(0.00)	0.02	(0.02)
Striped Burrfish	*Chilomycterus shoepfi*	2	0.00	0.00	(0.00)	0.00	(0.00)	0.02	(0.02)
Chain pipefish	*Syngnathus louisianae*	2	0.00	0.00	(0.00)	0.00	(0.00)	0.02	(0.01)
Naked goby	*Gobiosoma bosc*	1	0.00	0.00	(0.00)	0.01	(0.01)	0.00	(0.00)
Rainwater killifish	*Lucania parva*	1	0.00	0.00	(0.00)	0.00	(0.00)	0.01	(0.01)
Inland silverside	*Menidia beryllina*	1	0.00	0.00	(0.00)	0.01	(0.01)	0.00	(0.00)
Southern kingfish	*Menticirrhus americanus*	1	0.00	0.01	(0.01)	0.00	(0.00)	0.00	(0.00)
Shrimp eel	*Ophichthus gomesii*	1	0.00	0.00	(0.00)	0.00	(0.00)	0.01	(0.01)

Table 3. (Continued)

COMMON NAME	SCIENTIFIC NAME	TOTAL NUMBER	RELATIVE ABUNDANCE (%)	Low Cover		Medium Cover		Continuous Cover	
				MEAN	SE	MEAN	SE	MEAN	SE
Total Crustaceans		116,533	98.87	397.92	(77.78)	258.91	(51.39)	432.20	(66.98)
Arrow shrimp	*Tozeuma carolinense*	79,315	67.29	354.21	(73.38)	213.57	(47.96)	198.36	(55.32)
Hippolytes Pleurocantha	*Hippolytes Pleurocantha*	31,841	27.02	39.85	(8.82)	40.60	(6.77)	195.75	(31.56)
American Prawn	*Palaemonetes vulgaris*	3,447	2.92	0.63	(0.20)	1.92	(0.60)	26.90	(10.66)
Brown / Pink shrimp	*Farfantepenaeus sp.*	1,450	1.23	2.13	(0.69)	1.90	(0.42)	8.84	(1.51)
Blue crab	*Callinectus sapidus*	307	0.26	0.31	(0.09)	0.54	(0.23)	1.78	(0.65)
White shrimp	*Litopenaeus setiferus*	135	0.11	0.62	(0.24)	0.35	(0.14)	0.41	(0.12)
Sargassum Shrimp	*Latreutes parvulus*	25	0.02	0.16	(0.06)	0.02	(0.01)	0.06	(0.02)
Combclaw shrimp	*Leptochela serratorbita*	11	0.01	0.01	(0.01)	0.01	(0.01)	0.08	(0.03)
Snapping shrimp	*Alpheus heterochaelis*	1	0.00	0.00	(0.00)	0.00	(0.00)	0.01	(0.01)
Longnose spider crab	*Libinia dubia*	1	0.00	0.00	(0.00)	0.00	(0.00)	0.01	(0.01)

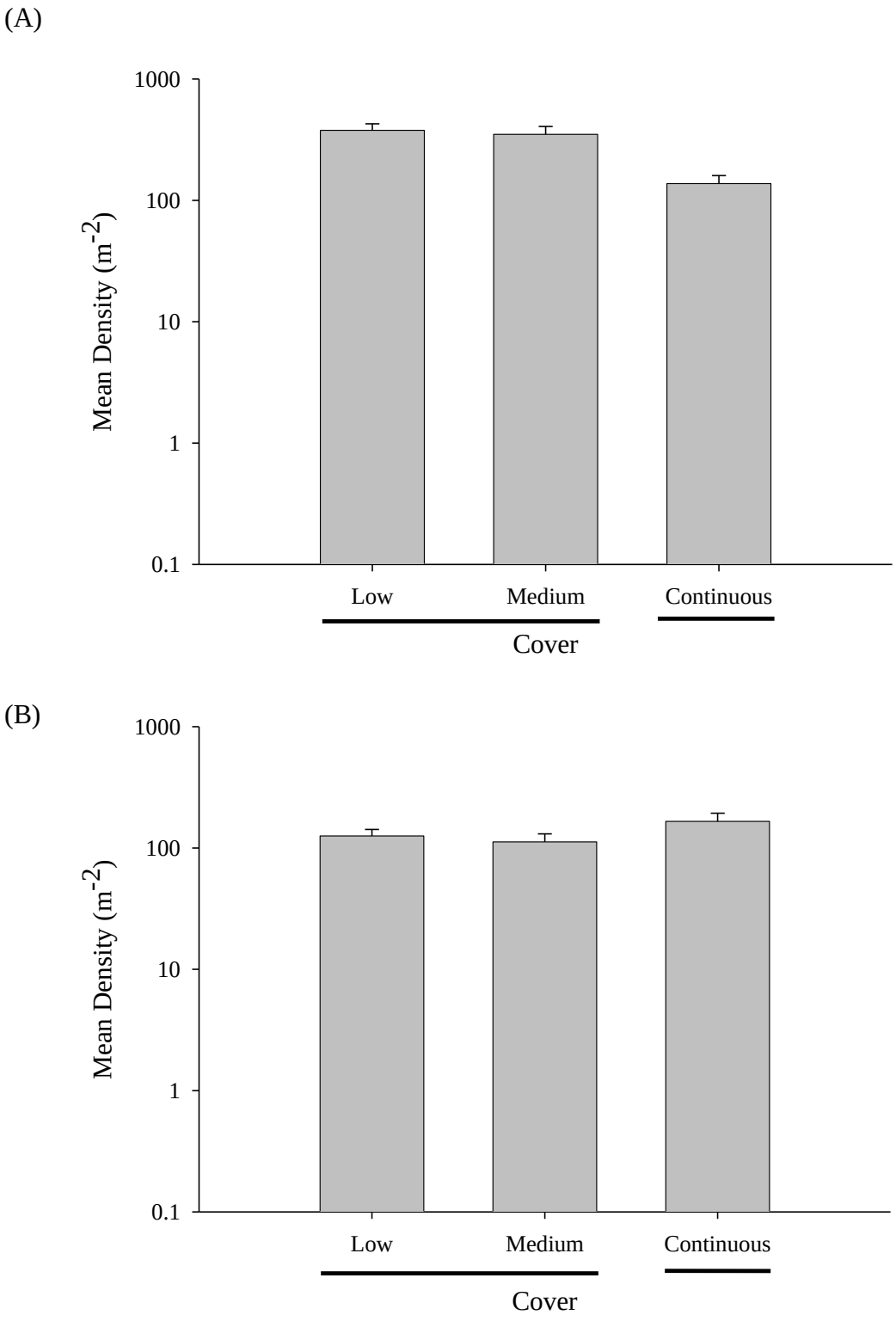
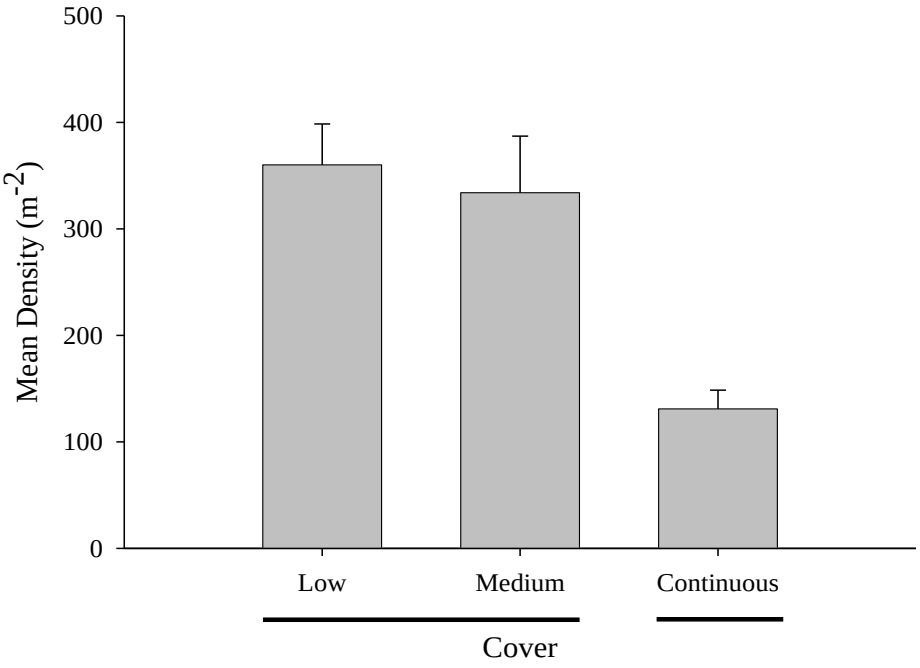


Figure 9. Total nekton density (m^{-2} ± SE) for three levels of cover in Corpus Christi Bay and Aransas Bay during (A) spring and (B) fall 2009. Bars that share a common horizontal line are not significantly different. In both seasons, densities of nekton found in continuous habitats were significantly different from fragmented habitats.

(A)



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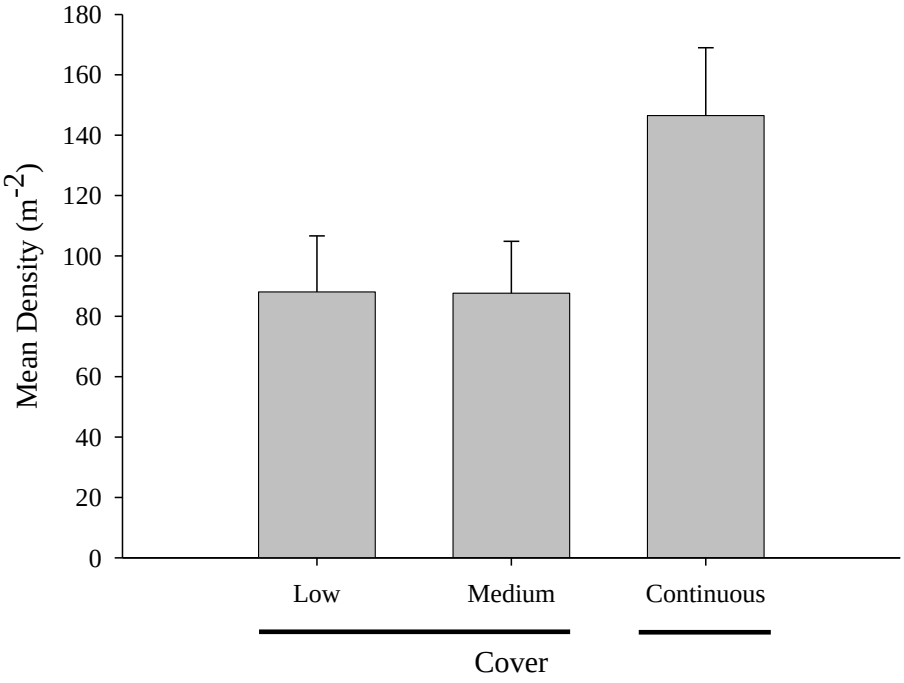
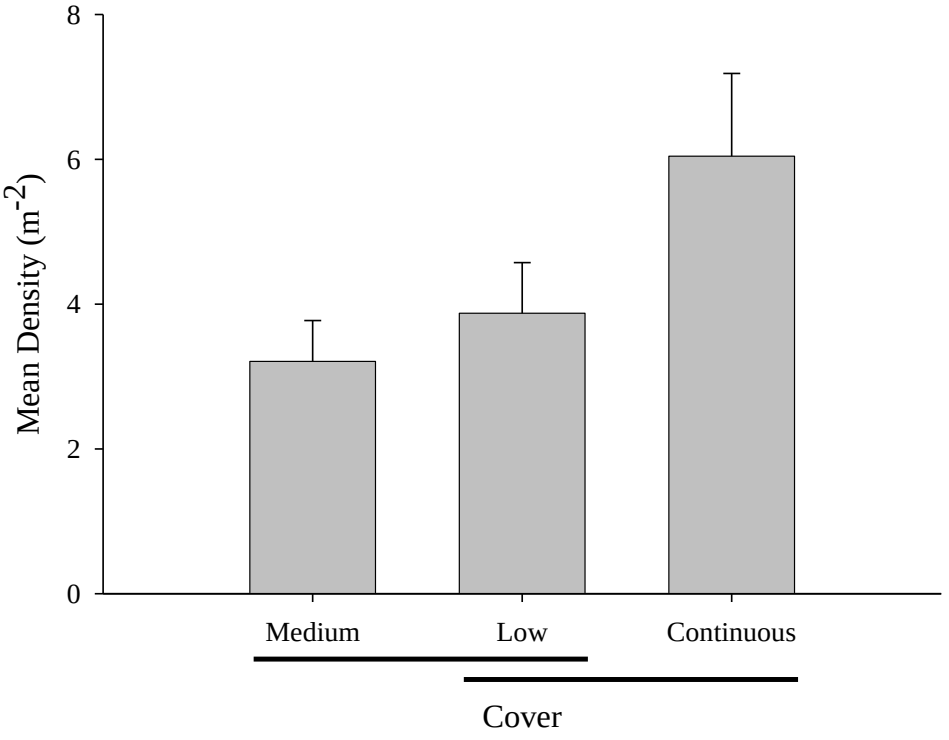


Figure 10. Total crustacean density (m^{-2} ± SE) for three levels of cover in Corpus Christi Bay and Aransas Bay during spring (A) and fall (B) 2009. Cover that share a common line are not significantly different. In both seasons, densities of nekton found in continuous habitats were significantly different from fragmented habitats.

(A)



(B)

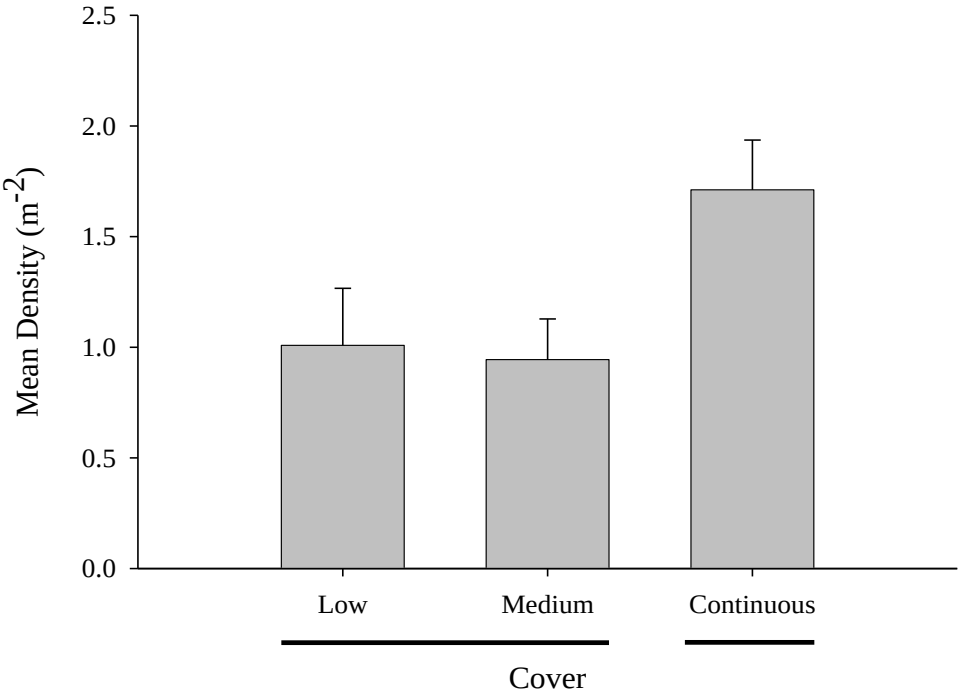


Figure 11. Fish density (m^{-2} ± SE) for three levels of cover in Corpus Christi Bay and Aransas Bay during fall 2009. Cover that share a common line are not significantly different.

Red drum densities and standard length.

Of the 117,863 organisms collected during fall epibenthic sampling, eighty two of them were newly settled juvenile red drum ranging in size from 6mm to 14.4mm ($\bar{x}$ = 7.8 mm SL). Red drum densities were not significantly different among three levels of cover ($F_{2,29}$ = 1.99; p= 0.1544) (Fig 12). However, the mean size (SL mm) of red drum per tow (n=43) was significantly different among low, medium, and continuous cover (F= 4.38; df= 2, 36; p= 0.02). *A priori* linear contrasts indicate that red drum lengths are significantly larger in continuous (9.2 mm ± 0.7) cover plots than in low (6.9 mm ± 0.2) cover plots. Red drum from medium (8.2 mm ± 0.7) cover plots did not differ significantly from continuous or low cover plots (Fig 13). Results suggest a gradual increase in SL of red drum from fragmented habitats to more continuous habitats.

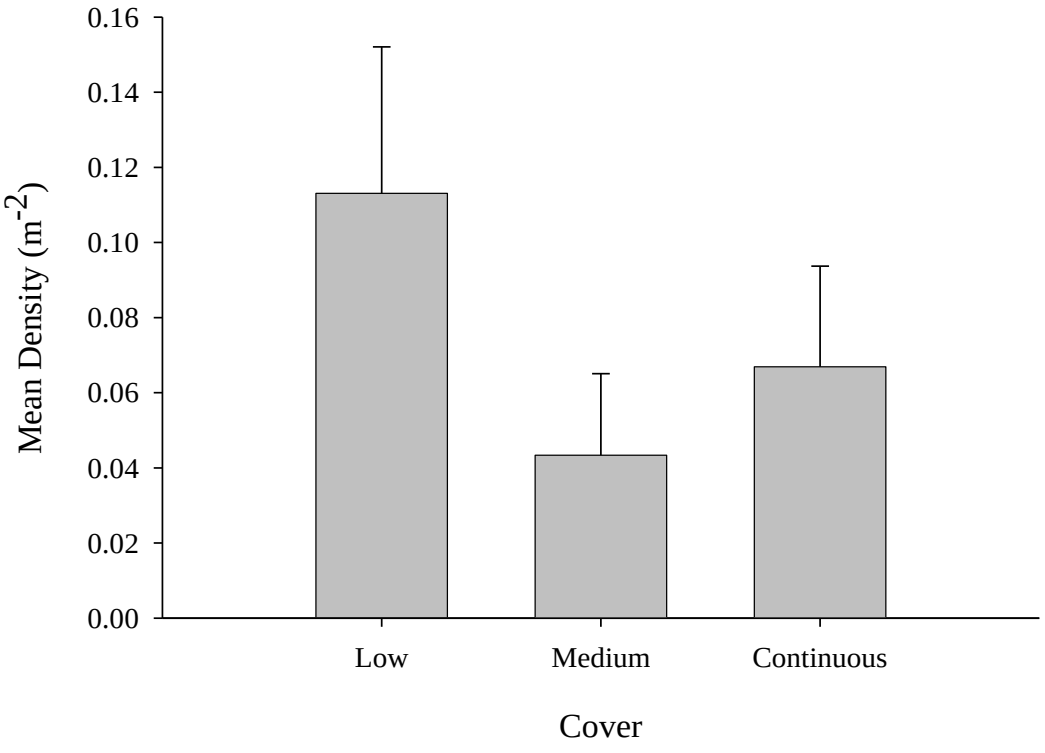


Figure 12. *Sciaenops ocellatus*. Red drum density (m^{-2} ± SE) for three levels of cover in Corpus Christi Bay and Aransas Bay during fall, 2009. No significant difference in red drum density among cover levels.

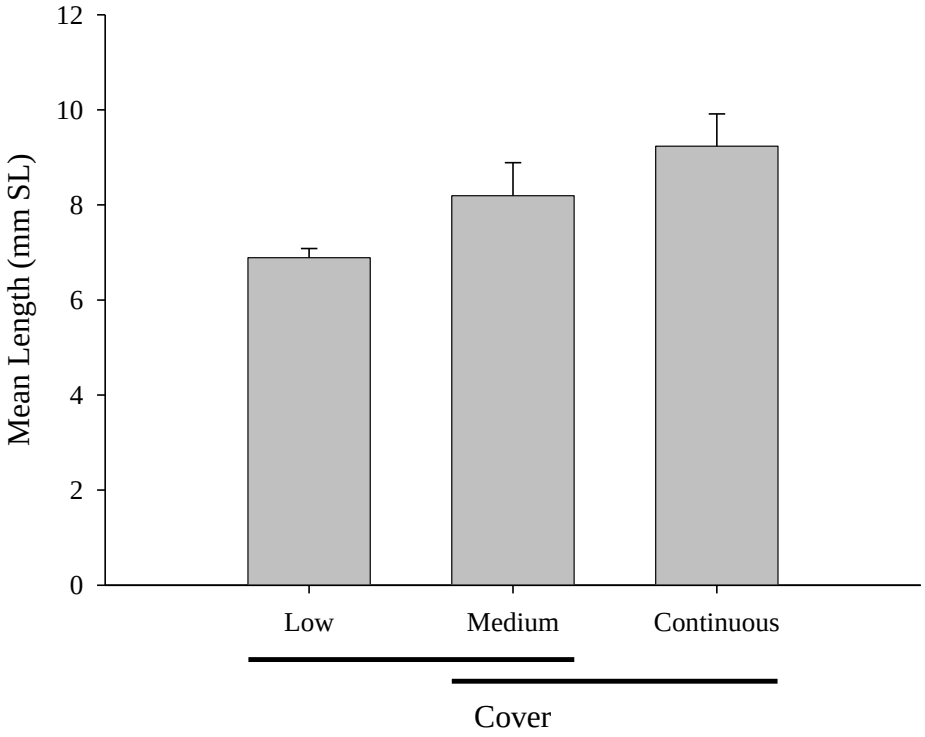


Figure 13. *Sciaenops ocellatus*. Mean red drum standard length (mm ± SE) for tows taken in three levels of cover in Corpus Christi Bay and Aransas Bay during fall 2009. Covers that share a common line are not significantly different.

Red drum growth

An ANCOVA using fish SL as the dependent variable and age in days as the covariate, found no significant interaction ($F_{1,60} = 0.00$; $p = 0.971$) indicating no difference in slopes of regression lines (growth rates) between habitats. There was a significant overall age-length relationship ($F_{1,60} = 224.93$; $p < 0.0001$) (Fig 14). The main effect of habitat fragmentation was not significant ($F_{1,60} = 0.02$; $p = 0.8867$), indicating no difference in size-at-age between habitats suggesting that red drum might settle equally into fragmented and non-fragmented habitats.

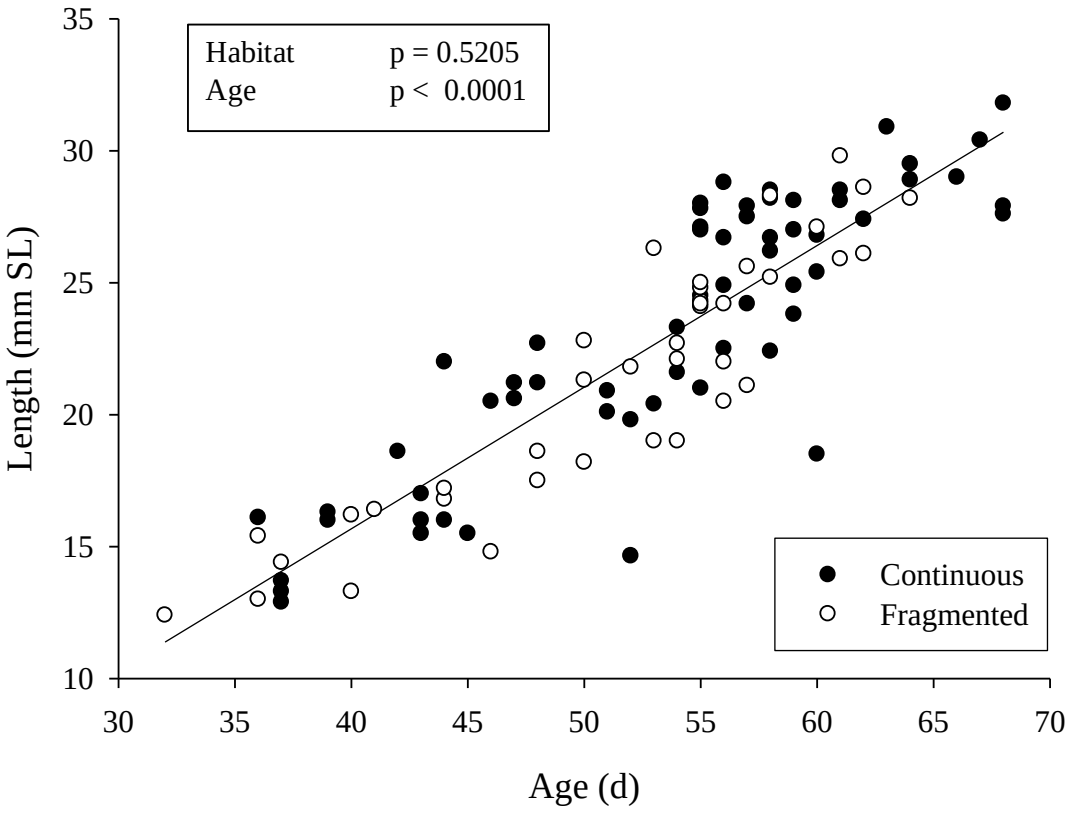


Figure 14. *Sciaenops ocellatus*. Relationship between age (d) and SL of red drum from continuous and fragmented habitats in Corpus Christi Bay. ANCOVA results show no significant differences between age-length regressions on fragmented habitats (n=64). Regression line is pooled from both habitats. SL = -5.8 + 0.54 (Age); $R^2 = 0.79$

RNA:DNA

Growth rates predicted by RNA:DNA ratio were similar among fragmentation levels suggesting that fragmentation did not influence recent growth rate. An ANCOVA model was run to test for a significant differences in RNA:DNA ratios between fragmented and non-fragmented habitats treating age as the covariate. There was no significant interaction between fragmentation and age ($F_{1,60}$ = 0.00; p= 0.9832), so the model was re-run without the interaction. Results indicate no significant difference in RNA:DNA ratios between fish taken from fragmented and non-fragmented plots ($F_{1, 61}$ = 1.36; p= 0.2489) (Fig 15).

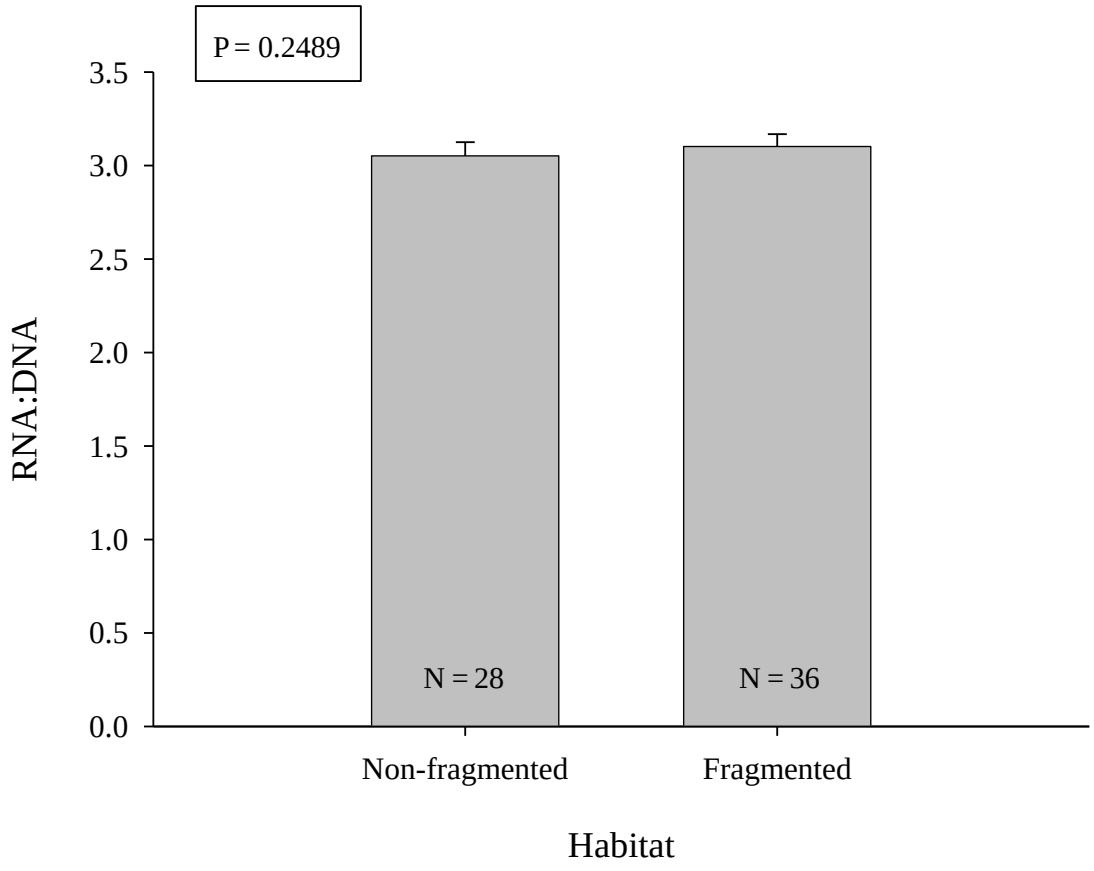


Figure 15. *Sciaenops ocellatus*. RNA:DNA ratios for red drum (n=64) sampled from fragmented and non-fragmented sites in Corpus Christi Bay during fall 2009.

Otolith microstructure

The distance from the otolith origin to the outer edge was measured in order to build a simple regression between fish SL and average (left, right) lapilli radius (Radius μm = 11.476x + 55.258; $r^2 = 0.96$), thus, enabling me to use otolith growth as a proxy for fish growth (Fig 16). There was no significant difference in age of red drum between fragmented (51.83 d^{-1} ± 1.36) and non-fragmented (54.89 d^{-1} ± 1.88) plots ($F_{1,62} = 1.82$; p= 0.1818). There was also no significant difference in mean growth (mmd^{-1}) of red drum between fragmented (0.41 mmd^{-1} ± 0.007) and non-fragmented (0.43 mmd^{-1} ± 0.0113) habitats ($F_{1,62} = 1.15$; p = 0.2871). A one way ANOVA indicated no significant difference in growth rates between fragmented and non-fragmented habitats for the last seven ($F_{1,62} = 1.75$; p= 0.1906), ten ($F_{1,62} = 0.72$; p= 0.3999), and fourteen ($F_{1,62} = 0.72$; p= 0.3996) days (Fig 17).

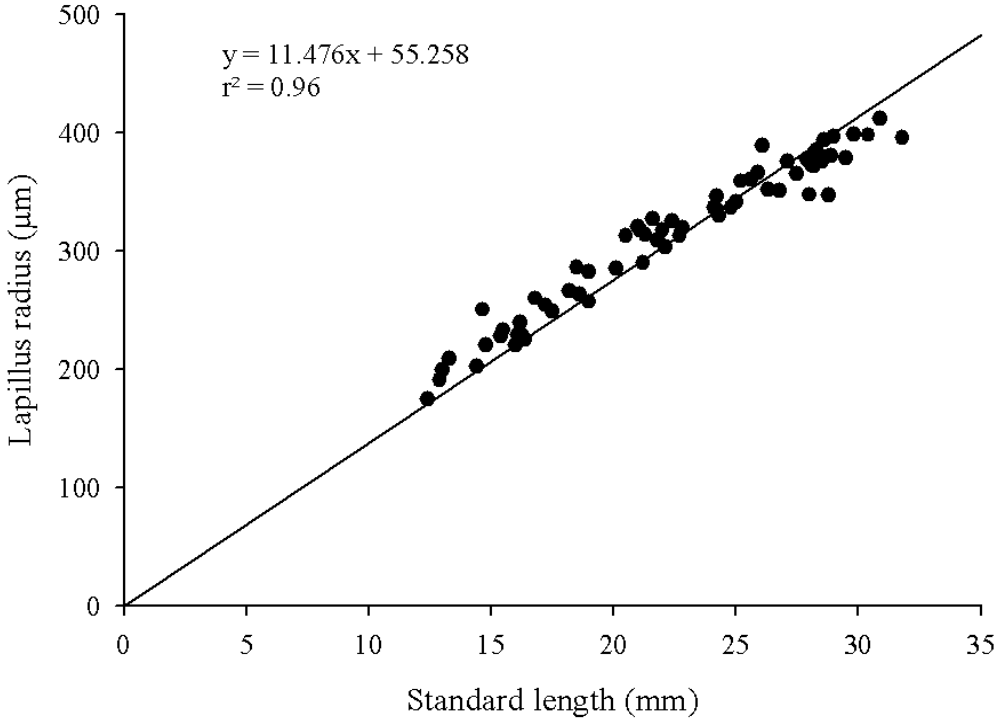
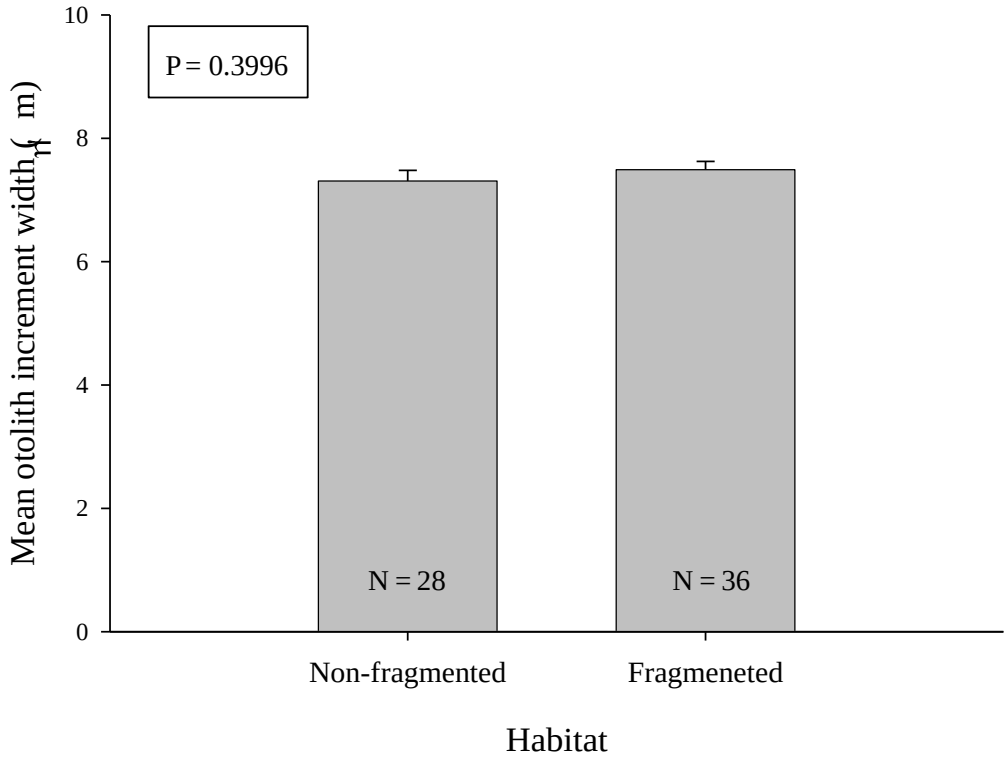
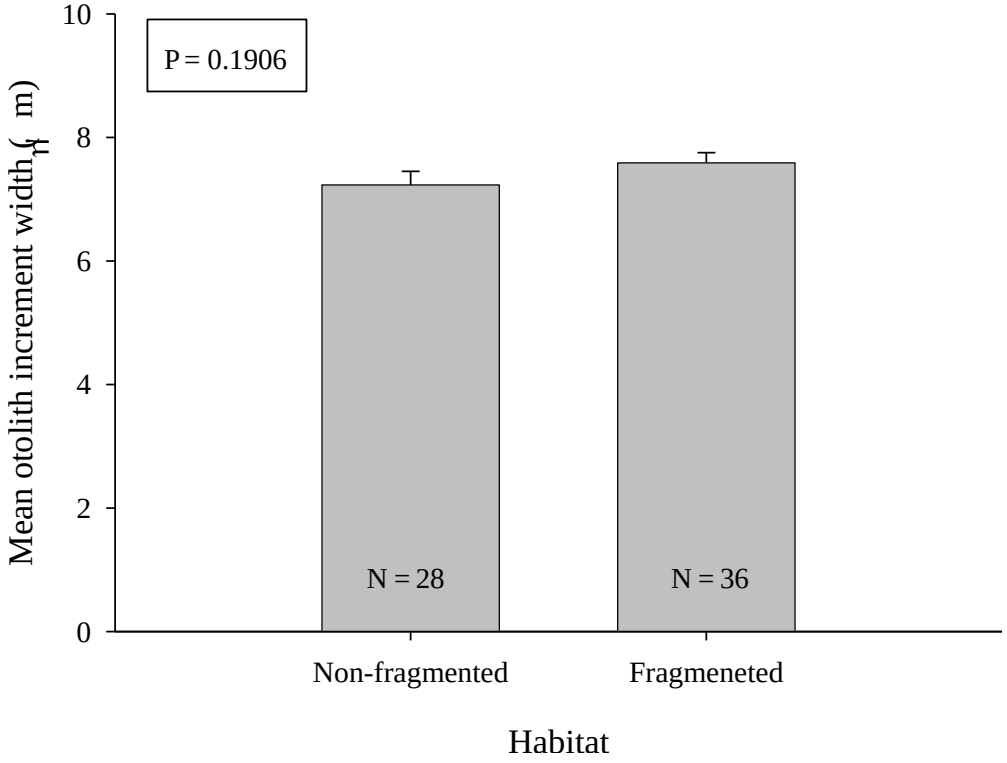


Figure 16. *Sciaenops ocellatus*. Linear relationship between lapillus radius (μm) and standard length of red drum (mm) sampled for growth analyses.

(A)


P = 0.3996
Mean otolith increment width (μm)
10
8
6
4
2
0
N = 28
N = 36
Non-fragmented
Fragmeneted
Habitat

(B)


P = 0.1906
Mean otolith increment width (μm)
10
8
6
4
2
0
N = 28
N = 36
Non-fragmented
Fragmeneted
Habitat

(C)

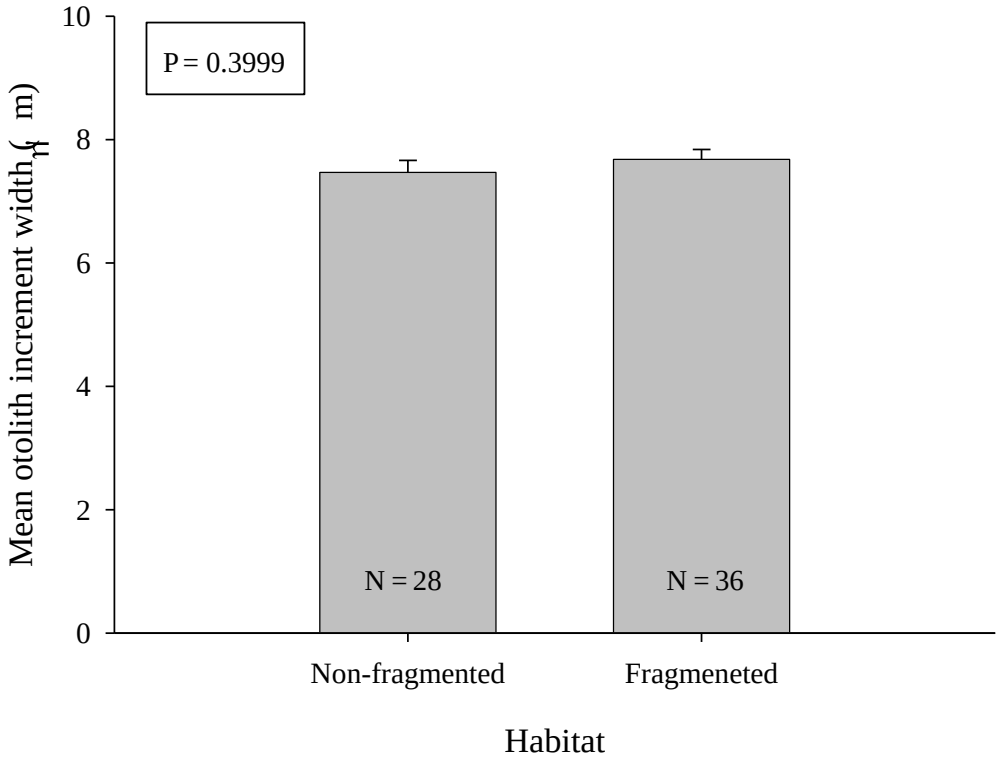


Figure 17. Mean otolith increment width (µm ± SE) for the last (A) 7d, (B) 10d, and (C) 14d of growth for newly settled red drum collected from fragmented and non-fragmented habitats in Corpus Christi Bay (n=64) during fall 2009.

Red drum movement

To asses fine scale movement of an estuarine dependent species within a highly fragmented seagrass network, 200 marked red drum were released into three highly fragmented (low cover) seagrass networks on November 26th, 2009. Fifty marked red drum were released into study site 1 while 75 marked red drum were released into sites 2 and 3. Although amounts of fish were different in each site, densities were similar but adjusted to reflect natural field densities (1.5 m^{-2}) determined from prior studies.

Subsequent intensive seining staring twenty-four hours after release, no marked fish were recaptured in the patches they were released in. Only one fish was recaptured

within a fragmented network. It was recaptured at site 1, 40 meters away from release point, which involved crossing two bare expanses measuring 2 and 3 meters. No wild red drum were captured at this site. No marked red drum or wild red drum were captured at sites 2 and 3. However, 5 marked red drum were recaptured within the neighboring continuous network 50 meters away from site 3 release point. In addition, 25 wild red drum were also caught in this continuous meadow. Forty-eight hours after release, no marked or wild red drum were captured at any of the sites. Seventy-two hours after release, 1 marked red drum was recaptured within site 1 in the same patch it was found two days prior. It's possible this was the same fish due to its growth from 18.6mm to 20.1mm in three days, resulting in a realistic growth rate (0.5 mm d^{-1}). No marked or wild red drum were found at site 2 or site 3. One marked red drum and 19 wild red drum were found in site 3's neighboring continuous meadow (Table 4).

Table 4. Recapture results for movement experiment carried out on 200 red drum in three highly fragmented sites in Corpus Christi Bay during fall 2009.

			Recapture Events					
			24 hrs		48 hrs		72 hrs	
Site	Fish Released	Distance from Continous Meadow	Marked	Wild	Marked	Wild	Marked	Wild
1	50	400 meters	1	0	0	0	1	0
2	75	120 meters	0	0	0	0	0	0
3	75	50 meters	5	25	0	0	1	19

DISCUSSION

The aim of this study was to locate and identify varying levels of fragmented seagrass beds (*Halodule wrightii*) in Aransas Bay and Corpus Christi Bay and investigate the effects of fragmentation on nekton densities growth and movement. I specifically looked at nekton densities among varying levels of fragmentation and chose to focus on red drum (*Sciaenops ocellatus*) as a model estuarine dependent species in terms of density, size distribution, growth rates, and movement. The project is unique in that a novel approach was used to delineate study sites using a handheld GeoXT Trimble unit. Overall, nekton densities in response to habitat fragmentation were varied between seasons and generally unclear. Crustaceans were found to be the driving force behind nekton densities, representing 98% of total nekton sampled. Fish densities were highest in more continuous meadows during spring and fall. However, without a closer look at individual species and size-class, results are variable. A more detailed investigation of newly settled red drum densities, size distributions, growth rates, and fine-scale movements help to paint a clearer picture of the effects of fragmentation on an estuarine-dependent species. Results suggest a temporal shift in size and densities of newly settled red rum from fragmented to more continuous habitats.

Mapping

The first goal of this research was to locate and delineate fragmented seagrass plots in Aransas Bay and Corpus Christi bays. Fragmented plots in close proximity to continuous meadows were chosen to minimize confounding site effects. Fragmented seagrass habitats in Corpus Christi Bay and Aransas Bay are most likely due to a combination of natural sediment deposition and anthropogenic activities like dredging

and shrimp trawling. According to Texas Parks and Wildlife, Aransas Bay has 14,000 acres of submerged seagrass and is considered the northern most stand of extensive seagrass beds. Mud Island was the only place within the Aransas bay system that showed a clear fragmentation of seagrass habitat. Mud Island is classified as a flying spit and known to be a site of high sediment deposition. Likewise, the southeast shoreline of Corpus Christi Bay has been shown to receive high levels of sediment deposition (Brown 2005). With an ever increasing human population along the Texas coast, anthropogenic effects are likely to increase. In 1999, there were 1,639 bay and bait shrimp vessels operating along the Texas coast (TPWD 2000). Increased sedimentation from dredging and trawling the bay bottom combined with daily tidal movements and wind born wave action could be responsible for sediment deposition on Mud Island and east Corpus Christi Bay. In general, seagrasses in this region experience relatively minor levels of fragmentation. Nevertheless, some fragmented areas were identified, and to accurately map these sites, this study took advantage of modern and relatively inexpensive GIS technology to map these areas.

The Trimble GeoXT GIS unit combines ground truthing in real time with geospatial analyses. It is one of the most accurate methods to date used to map fragmented habitats. I discovered several benefits to using this method, and future fragmentation or habitat ecology studies may benefit from this technology. Using the Trimble allowed me to increase the scope of my study by having larger study plots. This was necessary because fragmentation is a landscape issue and studying it involves mapping habitat patches and the matrix between them. Using the unit in conjunction with ArcMap also allowed better quantification of study sites and the individual patches within

them. These shapefiles and associated attribute tables can be archived and referenced for future studies. Another benefit to this methodology is creating a map which is more current than available flyover and satellite imagery. New satellite images are taken of the Texas coast each year by USDA-NAIP during the month of January. However, by this time seagrass patches have typically been reduced in size due to colder temperatures and feeding waterfowl (personal observation). The ephemeral nature of seagrass requires real-time truthing and using the Trimble makes this possible. What's more, using the Trimble in conjunction with ArcMap provides the option of investigating the effects of fragmentation on a multivariate level. Arcmap allows for an array of attributes to be derived from study sites. While I used percent cover as a proxy for fragmentation *per se*, there are indices for fragmentation like number of patches, area/perimeter ratio, and nearest neighbor (Sleeman 2005). These variables can be used to form a regression representing fragmentation. It is also possible to analyze the catch data by an individual patch sampled, taking into account relative size and distance to nearest neighbors. It is an extremely accurate method of *in situ* mapping that can be used in conjunction with satellite imagery. Finally, shapefiles along with associated databases from separate studies can be saved for future reference and data sharing, which could prove to be invaluable to documenting habitat changes and future managerial decisions.

Although it was outside the scope of my project, it would have been beneficial to map the entire north edge of Mud Island and the entire east side of Corpus Christi Bay. Once these areas are mapped, ArcMap can help describe relative fragmentation in terms of the local environment. Mapping the entire region also allows for a true random sampling design and can help to investigate the effects of fragmentation on the patch-

scale. Future studies in coastal ecology should consider using a Trimble as part of their studies. ArcMap helps to keep study sites organized and can offer additional perspective in terms of distance, area, and connectivity. Shapefiles of fragmented seagrass beds can be compared from season to season, year to year, and lab to lab.

Nekton

During the spring, higher densities of nekton were found in fragmented habitats compared to continuous meadows. Conversely, nekton densities during fall were highest in continuous meadows compared to high and medium fragmented plots. Interestingly, continuous plots differed from fragmented plots and fragmented plots did not differ between themselves during each season. These suggest that there may be some fundamental difference between varying levels of fragmentation and continuous habitats between seasons.

Differences in responses to fragmentation could be the result of factors such as habitat complexity, settlement, and "post-settlement" processes such as movement and differential mortality. During the fall season, I observed the seagrass patches to be generally larger and more dense than seagrass meadows in the spring. While the outer perimeters of these patches were mapped, core samples were not collected, thus, quantifiable metrics cannot be determined and would be an area for future studies. The link between fragmentation and habitat complexity is an area of fragmentation research which is often over looked and is considered an essential link from fragmentation to biotic response (Lindenmayer and Fisher, 2006). In a similar study, Johnson and Heck (2006) did not find a significant relationship between the size of an individual patch and above ground seagrass biomass. Habitat complexity may also influence settlement

patterns of certain nekton (Rozas and Minello 1998). Deposition may influence passively settling larvae by increasing settlement into edge habitat and a gradient may form with higher densities on edge habitats and fewer within the core of seagrass patches (Orth 1992). Habitats that are more fragmented typically have an increase in edge habitat (Forman 1995), and thereby an assumed increase in larval encounter rates compared to more continuous habitats with less edge. In addition to habitat complexity, "post-settlement" processes like habitat selection through movement and differential mortality might influence density values. Generalist species that have the ability to move throughout a habitat can lessen the negative effect of fragmentation in terms of densities and mortality (Bender et al., 1998). Many of the organisms collected (e.g., caridean shrimp) are highly mobile (Howard 1985), and previous experiments suggest that higher abundances of these organisms are due to predator avoidance rather than food availability (Coen et al. 1981, Orth et al. 1984). It is possible that nekton density results are a mere snapshot of a dynamic movement of nekton settlement, habitat selection, and mortality through predation.

Crustaceans comprised the majority of the organisms found in study plots during spring and fall, which suggests that the overall nekton response to fragmentation is driven by these crustacean densities. These results were comparable to previous experiments where the majority of nekton captured (> 80%), where found to be crustaceans common in south Texas seagrass beds (Rozas and Minello 1998, Reese et al. 2008, Gain 2009). *Tozeuma carolinense* (arrow shrimp), *Hippolytes Pleurocantha*, and *Palaemonetes vulgaris* (American prawn) compromise nearly 99% of total crustacean catch in this study. These organisms are commonly regarded as food source for juvenile fish, and

resulting densities of these crustaceans are likely to be influenced by movement and/or predator avoidance. To better understand the response of these organisms to fragmentation over a period of time, it is necessary to include size class in the analysis. This however, was not part of my particular role in what is a much larger investigation into nekton response to fragmentation.

Fish showed a varied response to fragmentation; the highest densities of fish occurred in continuous plots in both spring and fall samples. Gobies and pipefish account for the majority of the fish collected during spring and fall seasons. Gobies were mostly found in more continuous habitats while pipefish were more ubiquitous. These densities could also be the result of shelter and food availability (Wootton 1998). In highly fragmented areas, these fish are more susceptible to predation; therefore they preferentially seek more continuous beds. These results are similar to Kulczycki's (1981) where he showed significant increase in *Gobiosoma robustum* (code goby) and *Syngnathus scovelli* (gulf pipefish) in patches of drift algae within seagrass beds and suggests predator avoidance or habitat preference rather than food supply was responsible for increased densities around floating islands of drift algae. I found fish in all fragmentation levels suggesting that different groups of fish may prefer different levels of fragmentation based upon individual needs. For example, Fernandez (2005) found a higher abundance of adult small-sized schooling planktivorous fish living within more continuous seagrass habitats while there were higher numbers of nekton-benthic species and ambush predators in fragmented beds. Similarly, my results indicate higher densities of gobies in more continuous habitats while densities for *Cynoscion nebulosus* (spotted seatrout), *Bairdiella chrysoura* (silver perch), and *Anchoa mitcheli* (bay anchovy) appear

to be higher in more fragmented habitats. Therefore, depending on their feeding and behavior niches, varying levels of fragmentation may benefit certain species but not others. Also, it is difficult to make a definitive statement as to what is driving these differences without taking into account size class of individual species. Fernandez (2005) found no difference in fish abundance among fragmentation levels, but he did observe that overall size of fish were smaller in continuous plots. However, it is possible that in the current study, fish found in the more continuous beds are larger in size, a result of fast growth or increased predator avoidance due to habitat complexity. In general, this study shows that continuous habitats house the majority of fish species.

Ambiguous results in many fragmentation studies are often the result of different interpretations of the word "habitat." Lindenmayer and Fischer (2006) propose that fragmentation studies should aim to clarify the difference between general landscape vegetation and habitat as it relates to a particular species and it's occupancy within that environment. For example, fragmented seagrass beds may not necessarily mean fragmented habitats to some species. To investigate this point, one has to look at how an individual species might be using these habitats throughout various life stages. On a cursory level, fragmentation does not seem to greatly affect nekton and crustacean densities. It may however influence other aspects like survivability, growth, and movement. For these reasons, it is important to take a closer look at an individual species' use of these environments in order to determine if indeed they can truly be called functional 'habitats.' To further examine these processes at the individual species level, I investigated red drum densities, size distributions, growth rates, and movement patterns within local fragmented networks.

Red drum

Understanding how newly settled red drum respond to fragmented habitats requires an assessment of a variety of dynamic and interactive process namely, abundance/density, size structure, growth, and movement. I used red drum as a model to examine several ecological parameters associated with habitat fragmentation that may impact estuarine-dependent species. For example estuarine species use habitats that support energy gain (i.e., growth), provide shelter from predators, and minimize competition (Levin and Stunz 2005). Of these, starvation and predation are the two major forces influencing juvenile mortality, which in turn affects year-class strength (Houde 1986). Varying levels of density can reflect differential habitat selection and mortality due to resource availability and predation (Stunz et al. 2002). Clearly, these processes can be influenced by habitat fragmentation. Specifically, my goal was to characterize these responses to varying levels of fragmentation.

Holt (1983) found higher densities of red drum in the ecotone between bare patches and seagrasss beds. Because red drum are visual predators, they select for environments which increase foraging success. Stoner (1982) suggests that red drum are probably more successful at capturing prey in less vegetated areas. Thus, one might expect more red drum to be found in more fragmented areas with increased edge habitat, however, this trend not observed in this study. At the same time, red drum are a common prey item for larger faster fish and have a higher chance of surviving in more structurally complex environments (Rooker 1998, Stunz 2001). Holt (1983) suggests that more red drum were found along the edge of seagrasss beds because it provided shelter for hunting *and* refuge from predators. In this study, there was no significant difference in red drum

density among the varying levels of fragmentation (6mm – 14.4mm SL). Density results alone would indicate that all three levels of fragmented seagrass beds were suitable habitats for newly settled red drum. This however, would ignore a crucial component of red drum demographics within these habitats in terms of size-class.

Standard length of red drum varied significantly among fragmented networks, with larger fish being found in more continuous habitats. This could be the result of habitat selection (i.e., migration), increased predator avoidance due to habitat complexity, and/or faster growth rates. However, I suggest it is most like due to mortality or migration. Ten days after benthic sled samples were taken, I sampled all plots using a bag seine to collect red drum for growth analyses. While my sampling effort was not quantified, every patch within every network was seined at least twice. Fish ranged in size from 14.4 to 36.5 mm SL and averaged 23.2 mm, and no fish were found in highly fragmented, low cover, networks. Fish were only found in medium cover plots and continuous meadows suggesting initial settlement and then migration, predation, or both. Moreover, few fish were found within Mud Island fragmented study sites later during the study. Considering the presence and absence of red drum found from the growth collection, a "shift" from highly fragmented plots to more continuous habitats is observed. These patterns can be only be explained by two events. Red drum clearly settled to fragmented areas, but they either move to more continuous areas or were removed by predation. The size differences show that fish settling in more continuous areas grow and are observed in the larger size classes suggesting that these areas function better as nurseries.

Fast growth rates may reflect a healthy environment where resources are more abundant, while slower growth rates indicate an environment that may have limited food resources, which increases the time spent in the vulnerable juvenile life stage. However, I found no differences in growth rates between fragmented and non-fragmented habitats using both RNA:DNA analyses and otolith microstructure. RNA:DNA is a good indicator of growth and overall nutritional conditions of the fish (Buckley 1979), and a positive correlation between RNA:DNA ratios and growth rates have been shown by Westerman and Holt (1994). DNA is a species-specific constant while RNA increases with protein synthesis, i.e. growth. Starvation has been shown to decrease RNA:DNA levels in lab raised red drum within two days (Rooker 1996), and differences in RNA:DNA ratios among fish can give insight into the relative value of their environment as a providing habitat. Because there was no difference in RNA:DNA levels between habitats, I conclude that there is no lack of resources for red drum in fragmented or non-fragmented habitats.

Otolith microstructure also revealed no significant differences in age or growth rates between fragmented and non-fragmented habitats. These results are consistent with previous studies where no significant difference in RNA:DNA or growth, as determined through otolith microstructure, was found among study sites (Stunz 2002). The absence of fish in more fragmented habitats may be a combined result of slower growth rates and increased predation; it is possible that red drum are migrating and selecting habitats which maintain certain growth rates. Slower growth rates of red drum have been observed in lab and field caging experiments but not in natural collections (Hoff and Fuiman 1993, Holt 1993, Rooker and Holt 1997, Stunz et al. 2002, Reese 2008). A likely

explanation is that the slow-growing fish are removed from the population by predation, and my results are based upon growth rates from fish that have been in these habitats for several weeks. Undernourished red drum may have slower growth rates prolonging the critical phase of fish development, lowering chances for survival (Rice et al 1993). Fuiman (1994) also found that red drum survival increased substantially once they reached 20mm SL, clearly fish of this size would have lower predation risk and could more easily seek habitat that maximize their probability of surviving into larger size classes. This has been seen in pelagic fish that stay within certain temperature gradients, maintaining a constant growth rate throughout the year (Schuck 1951). Perhaps red drum are also moving to environments which maintain certain growth rates. However, the ecological trade-off for red drum to move from highly fragmented seagrass networks to more continuous habitats increases risk of predation by covering large expanses of non-vegetated bottom.

Movement can influence size and density patterns of red drum, and red drum are capable of large movements (Stunz et al. 2002; Bushon et al. 2007). It is known that older, larger juvenile red drum (>40mm) move into relatively deeper waters located within primary bays (Pearson 1928). Some work on juvenile and adult red drum movement has been done in the past (Bushon et al. 2007, McEachron et al. 1998), but none have investigated juvenile red drum movement within a fragmented network. Bushon et al. (2007) reported a marked red drum (25mm) traveling 200 meters within 72 hours of release. While movement of red drum within seagrass beds is somewhat understood, movement across bare substrate is not. Settle and stay hypothesis predicts that smaller fish are not willing to cross expanses of bare substrate due to increased

predation (Bell 1987). The results from my movement experiment however indicate that red drum are willing to travel across expanses of bare substrate in order to reach more continuous habitats. The general pattern I observed was that the majority of red drum were recaptured in a continuous meadow in the presence of other wild red drum. Two weeks past peak recruitment season, no wild red drum were found in any of the more isolated fragmented networks. During my movement experiment only one fish was recaptured within the fragmented network it was released. This fish was later recaptured 72 hours later. I believe that this red drum was effectively isolated from seagrass habitat and that the other marked red drum released into the same network were preyed upon.

Understanding how newly settled red drum respond to fragmented habitats requires a holistic view of densities, size structure, growth, and movement. Considering all of these factors and the results from the present research, I suggest that there is a temporal transition of red drum size and density from fragmented habitats to more continuous meadows. These data suggest red drum settle at relatively consistent densities among fragmented habitats but will either be preyed upon or migrate to more protective continuous meadows. Size distributions revealed significantly larger red drum occurring in more continuous habitats. This is most likely the result of increased predator avoidance. Using a seine net, which targeted larger fish, I sampled the same plots for growth and analyses and found the same trend; no fish were found in higher fragmented beds. Growth analyses on new recruits indicated no difference in growth between fragmented and non-fragmented habitats, suggesting there was no lack of resources in these habitats or detrimental conspecific competition due to density levels. Predator avoidance and/or habitat selection are likely reasons for the presence of larger fish in

more continuous beds along with no differential growth rates. Indeed, these two factors may be inter-related as selecting habitat with more complexity also increases predator avoidance thereby increasing survival (Legget and Deblois 1994, Rooker 1998b, Stunz 2001). For red drum to actively "select" for these more continuous habitats, they will have to be able to cross bare expanses of substrate, thereby being exposed to predation. My movement experiment shows that red drum are willing to cross expanses of bare substrate in order to reach more continuous habitats. Once again, I was unable to recapture marked red drum or capture wild red drum in highly fragmented networks, suggesting they were either preyed upon due to their isolation, or left the seagrass network altogether. Survival depends on a prey's ability to avoid predation, and complex habitat has been shown to reduce predation by decreasing detection and hindering predator movement (Rooker 1998b Stunz et al. 2002). In the case of red drum, it appears that a habitat can become too fragmented, whereby an individual patch can functionally be removed from its associated network, isolating newly settle red drum and decreasing chances for survival. At this point, the fragmented seagrasss beds cannot be considered viable habitat, because they fail to provide adequate refuge from predators. This however is not to say that these environments are not useful habitats for adult red drum, that may benefit from fragmentation through increased mobility and successful predation. Continuous habits are extremely valuable in terms of juvenile red drum growth and survival. Fragmented networks may still be useful habitats for adult red drum. The presence of adult red drum however depends upon the successful recruitment of larval red drum into adulthood. With this in mind, future efforts to conserve seagrass beds

would do well to minimize fragmentation while preserving healthy, continuous seagrass meadows.

Conclusions and future studies

It is apparent that estuarine nekton have varying response to habitat fragmentation. There is a flux in densities and size from some species across varying fragmentation levels. My investigation may be a mere snapshot in what otherwise is a dynamic system. Future studies should determine impacts of fragmentation from first year-class settlement to adulthood based upon individual species and their specific habitat requirements. A clear distinction should be made between landscape vegetation and true habitat. Nekton response to fragmentation varied between seasons and was largely driven by crustacean densities. Densities appear to be driven largely by "post-settlement' processes like habitat selection and predation, these habitats do not seem to be deplete of resources. Initial supply (i.e., settlement) appeared consistent among fragmentation levels, and later demographic impacts were observed via post-settlement causes. Analyses of red drum densities, size distribution, growth rates, and movements suggest a temporal shift in size and densities of red drum from fragmented habitats to more continuous habitats. Future studies should attempt to identify a fragmentation "threshold," whereby a habitat becomes fragmented to the point where distances between seagrass patches are too great and organisms living within them become isolated and are unable to move to larger more continuous habitats. At this point, isolated patches cannot be considered true habitat as far as juvenile red drum are concerned. Such isolation could impact recruitment success of this estuarine-dependent species. These estimations could

have implications for management if seagrass beds continue to become fragmented changing the functionality of this essential fish habitat.

Data from this study provides important information for managers for better conservation and management strategy and departure points for future habitat fragmentation studies. Clearly, I observed significant effect from fragmentation on the model estuarine species red drum. Given trends in human population growth fragmentation is likely to become a pervasive problem. Based on these findings I believe that while adult red drum may frequent these fragmented habitats, the young of the year require a greater connectivity within seagrass meadows and run the risk of predation through isolation if seagrass networks become too fragmented. Future studies would do well to identify this "threshold", thereby allowing for more informed conservation efforts based upon the specific needs of individual estuarine-dependent species.

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