3-17-2016

Distribution, Abundance and Movement of Fish among Seagrass and Mangrove Habitats in Biscayne Bay

Patrick C. Goebel
Nova Southeastern University, pg443@nova.edu

NSUWorks Citation

Patrick C. Goebel. 2016. *Distribution, Abundance and Movement of Fish among Seagrass and Mangrove Habitats in Biscayne Bay.* Master's thesis. Nova Southeastern University. Retrieved from NSUWorks, . (403)
https://nsuworks.nova.edu/occ_stuetd/403.

HALMOS COLLEGE OF NATURAL SCIENCES AND OCEANOGRAPHY

DISTRIBUTION, ABUNDANCE AND MOVEMENT OF FISH AMONG SEAGRASS AND MANGROVE HABITATS IN BISCAYNE BAY

By

Patrick C Goebel

Submitted to the Faculty of
Halmos College of Natural Sciences and Oceanography
in partial fulfillment of the requirements for
the degree of Master of Science with a specialty in:

Marine Biology

Nova Southeastern University

March, 2016

Distribution, Abundance and Movement of Fish Among Seagrass and Mangrove Habitats in Biscayne Bay

Submitted in Partial Fulfillment of the Requirements for the Degree of

Masters of Science:

Marine Biology

Patrick C Goebel
Nova Southeastern University
Halmos College of Natural Sciences and Oceanography

March 2016

Approved:

Thesis Committee

Major Professor :______________________________
David W. Kerstetter, Ph.D.

Committee Member :___________________________
Neil Hammerschlag, Ph.D.
University of Miami,
Rosenstiel School of Marine and Atmospheric Sciences

Committee Member :___________________________
James D. Thomas, Ph.D.

Abstract

Inshore tropical and subtropical estuaries harbor a relatively high abundance and diversity of organisms. Specifically within estuaries, mangrove and seagrass habitats provide shelter and food for a plethora of organisms, through some or all their life histories. Given the biological connection between offshore coral reefs and coastal estuaries, there is a critical need to understand the underlying processes that determine distribution and abundance patterns within mangrove-seagrass habitats. The predatory fish assemblage within the mangrove and seagrass beds of Biscayne Bay, Florida (USA), was examined over 24-hr. time periods along a distance and habitat gradient from the mangrove edge and nearshore environment (0–300 m) to farshore (301–700 m) seagrass beds. This thesis also investigated the occurrence, distribution and timing of reef fish movement between offshore coral reef habitat and inshore seagrass beds over 24-hr periods. Results indicate that fish predators differed over both the sampling period and with distance from mangrove edge. The results also demonstrated reef fishes move into Biscayne Bay at dusk and exit at dawn by utilizing Broad Creek Channel as a passageway. This work supports the idea of diel migration of selected reef fishes to inshore seagrass beds and highlights the importance of connective channels between habitats. The results suggest that the degradation or loss of seagrass habitat could differentially impact the life-history stages of reef fish species.

Keywords: seagrass beds, mangroves, coral reef fish, ecological connectivity, diel cycle, snapper, grunt, seagrass beds, barracuda, shark

Acknowledgements

I thank my advisors, Dr. Neil Hammerschlag and Dr. David Kerstetter for all their guidance and patience as well as the rest of my committee, Dr. James Thomas. You guys were always available, always helpful, and always demanding more. Neil, I will forever be grateful for the opportunity you granted me. You gave me a chance when no one else would. This thesis would never have been possible if it wasn't for you. I thank you not only for your assistance with this research project, but also for your education guidance and friendship. Dave, your guidance and feedback throughout this process was invaluable. The time and effort you put into this thesis will never be forgotten. Your attention to detail has strengthened my writing skills. I know pay attention to every detail, especially differentiating between fish and fishes. Thank you for your help.

I also thank my friends and family. Dad and mom, you encouraged me to follow my passion and supported me throughout this process. If it wasn't for your support, I wouldn't be in the position I am today. You have taught me so many life lessons and I will forever be grateful for your generosity and loving nature. To my lovely sister and genius brother-in-law, thank you both for all of your edits.

To my friends at the Shark Research and Conservation Program you will never be forgotten. We have shared a lot of great memories and I will cherish every day on the water and Wetlab, of course. Catherine, you looked out for me ever since I joined the program. I have learned so much from you on and off the boat. Thank you for getting the ball rolling on the thesis. Jake, your assistance with GIS was extremely helpful. Thank you all.

Additionally, I would like to thank Carl Hamp and the Florida Biodiversity Institute for providing the necessary equipment to complete this project. I also thank Bigblue Dive Lights for their generosity and support. I thank Evan D'Alessandro and the University of Miami for their generous hospitality as well.

To everyone who asked, "are you done yet?" Yes, I am done!

Table of Contents

List of Tables

List of Figures

Introduction

Mangrove and seagrass ecosystems are common coastal and estuarine habitats found broadly in tropical and subtropical latitudes (Saenger et al. 2013). Both habitats provide shelter, protection, and food for a plethora of organisms, through some or all of their life (Beck et al. 2001, Saenger et al. 2013). However, anthropogenic development continues to destroy these habitats on a global scale (Waycott et al. 2009). The urbanization of southeast Florida, and Miami in particular, has resulted in many changes, including habitat loss (Snedaker & Biber 1996) and a restructuring of the local hydrography through a series of canals and dikes. These resulting environmental modifications and degradation can have both direct and indirect effects on seagrass, mangrove, and coral reef fish species (Knip et al. 2010, Short & Wyllie-Echeverria 1996). Given the known biological connectivity among mangrove forests, seagrass beds, and coral reefs (e.g., Luo et al. 2009), the loss of habitat in one area can greatly affect fish populations in another. Thus, an understanding of a species distribution and the underlying processes that determine those patterns is a necessary component to define critical habitat and thus evaluating the potential effects of exploitation and anthropogenic change (Torres et al. 2006, Hannan et al. 2012).

Such studies are particularly important in Biscayne Bay (hereafter, simply "the Bay"), a large and semi-enclosed body of water located in southeast Florida, which continues to be impacted by anthropogenic effects, such as the urbanization of Miami. The Bay is a shallow, subtropical bay 65 km in length, up to 15 km wide, and with a 2 m average depth, except in dredged channels (Roessler and Beardsley, 1974, Serafy 2003). Up to 64% of the Bay bottom is covered with seagrasses because sediment depth and nutrients are sufficient, water depths are shallow, and water clarity is usually high (Markley and Milano 1985, Lirman and Cropper 2003, Browder et al. 2005). It is bordered to the west by mainland (Miami) and to the east by small mangrove islands. The eastern boundary of southern Biscayne Bay is relatively pristine. It has not lost a significant amount of mangrove forest and experiences minimal anthropogenic effects, thus making it an ideal study location.

There is a mixture of habitats in the Bay varying from dense seagrass beds to hardbottom to dredged channels (Roessler and Beardsley, 1974, Serafy et al. 2003, Lirman et al. 2008). The diverse habitats of the Bay support four types of seagrasses, over 500 species of fish and 800 species of invertebrates (Alleman et al. 1995, Lirman et al. 2008). The seagrass beds and mangrove habitats in Biscayne Bay act as a foraging and nursery ground for ecologically and economically important reef fishes in South Florida (Bohnsack and Ault, 1996). Several fish species, such as grunt and snappers, are known to use the Bay as a nursery area (Serafy et al. 1997, Serafy et al. 2003 Hammerschlag and Serafy 2010, Hammerschlag et al. 2010a,b) before making an ontogenetic migration to offshore coral habitats (Sedberry and Carter 1993, Ogden and Zieman 1997, Ley and McIvor 2002). Juvenile reef-fishes within the Bay are not evenly distributed and generally occur more frequently on the bay's more stable eastern shoreline (Serafy et al. 2003).

The overall objective of this thesis is to determine distribution, abundance and movement patterns of fish in southern Biscayne Bay. This thesis is split into two chapters, each focusing on different habitats within the Bay. The first chapter focuses on understanding the predatory fish assemblage in two critical fish nursery areas (mangrove and seagrass beds). Although commonly believed that these two nursery areas harbor lower abundances of predators, this perception has been recently challenged with studies reporting significant piscivore assemblages and high predation rates (Baker and Sheaves 2005, Baker and Sheaves 2006, Dorenbosh et al. 2009, Hammerschlag et al. 2010a,b). Nearshore predators are also presumed to be more active during crepuscular and nocturnal periods, yet studies investigating diel patterns of nearshore predators are few and little empirical evidence exists in support of increased predator activity during dark periods. Chapter 1 therefore examined and compared the predatory fish assemblage within the mangrove and seagrass beds of Biscayne Bay over 24-hr periods along a distance and habitat gradient from the mangrove edge and nearshore environment (0–300 m) to farshore (301–700 m) seagrass beds.

The second chapter focuses on understanding the movement of reef fish into and out of the Bay from seagrass beds (nocturnal feeding grounds) to adjacent coral reefs (diurnal refuging areas). It is commonly acknowledged that fishes leave coral reefs at night for shallow seagrass beds to feed on emerging invertebrates, although this conclusion is largely inferred from relative abundance of fishes in reefs and seagrass beds at different photoperiods (Robblee et al. 1984, Beets at al. 2003, Nagelkerken et al. 2000, Mumby et al. 2004, Berkström et al. 2013). However, there is little direct evidence of reef fish movement to and from nearby reefs into seagrass beds. Chapter 2 therefore investigated the flux (occurrence, distribution and timing) of reef fish movement between offshore coral reef habitat and inshore seagrass beds over 24-hr periods. The research specifically addressed diel differences in the flux of fish between an offshore coral reef and inshore bay through a channel in southern Biscayne Bay.

Baited Remote Underwater Video Surveys

Fish abundance and assemblages in inshore areas can be sampled using a variety of techniques, ranging from observation to extraction (see methods described in Taylor et al. 2013); the sampling method chosen is based on the focus of the particular study. The majority of inshore surveys have been done historically using one of two methods: underwater visual surveys and nets. However, there are limitations and therefor biases to underwater visual surveys and nets, such as either size limitation or capturing “shy” and cryptic species.

The use of underwater video systems has been used to investigate relative densities and species diversity of fish assemblages since 1967 (Isaacs 1969). As both the performance of digital camera technology has improved and the costs have decreased, these underwater systems have become more commonly used to monitor fish distributions. Cameras create a permanent record, which can be saved and used for either further studies or as a comparison for long term monitoring (Cappo et al. 2006). Camera surveys are also a cost-effective alternative to the more traditional net-based survey techniques, which typically require more personnel and field time (Brooks et al. 2011) and can be locally destructive to sensitive substrates.

The flexibility and non-destructive method of video cameras has allowed them to be used to investigate and determine fish abundances in a variety of habitats, including shallow coral reefs (Chapman et al. 2011), estuaries (Taylor et al. 2013, Gladstone et al. 2012), pelagic environments (Heagney et al. 2007), deep rocky reefs (Goetze et al. 2011), and shallow rocky reefs (Broad et al. 2010). This technique allows for the detection of fishes of any size within environments of any rugosity and depth, as well as across long time periods (Cappo et al. 2011, Harvey et al. 2012). Underwater video stations have a vast depth at which they can be deployed and the use of lights allow these stations to be deployed at night and at depths were light does not penetrate (Cappo et al. 2004, Harvey et al. 2012). Underwater visual surveys and traditional methods are species and size selective (Lowry et al. 2012, Harvey et al 2012). Specifically, “shy” species, cryptic species, and large predators, such as sharks, will actively avoid divers (Brock 1982, Watson et al. 2005), which can result in biased surveys. Also, underwater visual surveys, hook-and-line, trap, and trawl gear sampling methods are all limited by depth, fish behavior, seafloor rugosity, and size selection (Cappo et al. 2006). By adding a baited component near the camera, it allows the inclusion of the top predators, which are attracted to the camera by either the bait plume itself or the aggregation of smaller fishes in the area (Taylor et al. 2013, Watson et al. 2005). Studies of top predators are often logistically challenging and relatively expensive, and it is important to develop appropriate, yet cost-effective methods for identifying their spatial distribution. This study used baited remote underwater videos to investigate the predatory fish assemblages in seagrass beds and mangrove fringe habitats in the southern portion of Biscayne Bay.

1. Distribution, abundance, and movement of predators among seagrass and mangrove habitats in a subtropical coastal bay

1.1 Background

Inshore tropical and subtropical waters are biologically productive and often harbor a relatively high abundance and diversity of fish and invertebrates. In particular, seagrass beds and mangrove forests provide food and shelter for numerous economically and ecologically important fishes through some or all of their life (Beck et al. 2001, Adams et al. 2006, Nagelkerken et al. 2008). It is widely accepted that such habitats harbor lower abundances of piscivorous fishes (Patterson and Whitfield 2000). Recently, this model has been challenged, due to significant piscivore assemblages and high predation rates observed in some of these habitats (Baker and Sheaves 2005, Baker and Sheaves 2006, Dorenbosh et al. 2009, Hammerschlag et al. 2010a, Hammerschlag et al. 2010b). In particular, predation risk to juvenile fishes may actually be high, especially under low light conditions such as nocturnal and crepuscular periods when fishes leave refuges to forage (Danilowicz and Sale 1999, Unsworth et al. 2007, Hammerschlag et al. 2010a,b). However, due to minimal nocturnal sampling in prior studies, the actual predator assemblages in inshore tropical habitats may be underestimated in nearshore habitats (Baker and Sheaves 2006). Accurately characterizing the predator assemblages in these habitats is important as predators can directly influence habitat use and fitness of their prey, which can in turn indirectly impact ecosystem dynamics (Madin et al. 2015), including carbon sequestration (Atwood et al. 2015).

Numerous studies have been conducted investigating the distribution of fishes within the mangrove and seagrass habitats of Biscayne Bay, Florida (USA) (Serafy et al. 1997, Serafy et al. 2003, Serafy et al. 2007, Faunce and Serafy 2008, Serrano et al. 2010, Hammerschlag and Serafy 2010). For example, Serafy et al. (2003) noted that more fish species were collected from salinity-stable versus variable-salinity areas within the Bay, and that the mangrove shorelines on the seaward side of the islands consistently harbored higher numbers of fish taxa than those on the mainland side. However, large mobile predatory fishes have often been absent from these studies, presumably because the methods used in the studies (e.g., seine nets) typically under-represent large mobile predators, which have greater avoidance ability than smaller schooling fish (Brock 1982). Indeed, tethering experiments revealed high predation rates on juvenile fishes at night near the mangrove-seagrass ecotone in Biscayne Bay (Hammerschlag et al. 2010a,b). However, the composition and structure of the predator assemblage and how it may change spatially with distance from shore or temporally over a diel period in this area remains limited.

In the present chapter, BRUVS were used to investigate the composition and structure of the predator assemblage in Biscayne Bay, Florida, along a distant gradient spanning from the mangrove edge across adjacent seagrass beds at different times of day. Specifically, this study examined the distribution of fish predators (species composition, relative abundance, and size structure) from 0-700 m from mangrove edge and evaluated if and how these patterns varied by diel period (day, night, dusk, dawn).

1.2 Methods

1.2.1 Study Site

Biscayne Bay is a shallow, subtropical bay 65 km in length, up to 15 km wide, and with a 2 m average depth, except in dredged channels (Figure 1.1; Roessler and Beardsley, 1974). Up to 64% of the Bay bottom is covered with seagrasses because sediment depth and nutrients are sufficient, water depths are shallow, and water clarity is high (Markley and Milano 1985, Browder et al. 2005). The eastern boundary of southern Biscayne Bay is relatively pristine. It has not lost a significant amount of mangrove forest and experiences minimal anthropogenic effects, thus making it an ideal study location. This study was conducted along the eastern (leeward) side of Totten Key in the southern part of the bay (Figure 1.1).

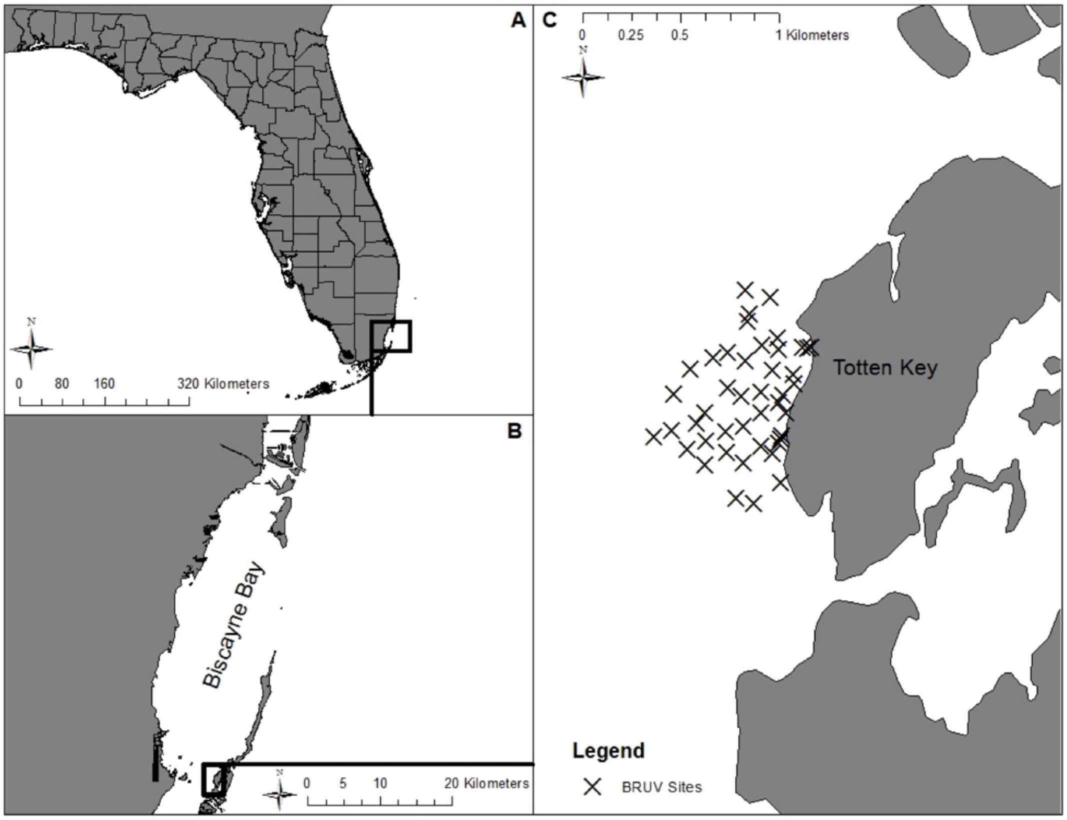


Figure 1.1 Study area: A) Location of Biscayne Bay in Florida; B) location of study area within Biscayne Bay; and C) position of study area on leeward side of Totten Key within Biscayne Bay. X's represent the 45 deployment sites

1.2.2 Data collection

Baited underwater video surveys (BRUVS) were used to sample predator distribution and abundance. BRUVS were composed of a polyvinyl chloride (PVC) frame containing a video camera (GoPro; Hero3) and bait crate. The bait crate was placed at the end of 0.75 m long bait arm held in a horizontal orientation to the substratum (Klages et al. 2014). A thirty-centimeter ruler was placed at the end of the bait arm, which was used to estimate fish size (total length) (Figure 2.1). Prior to each deployment, the bait crate was filled with 0.8 kg of chopped frozen pilchards, *Sardinella aurita* (Valenciennes, 1874). For nocturnal and crepuscular sampling, BRUVS were equipped with four underwater red lights (Bigblue; model AL900 XWP) within a waterproof housing. Similar to the design used by Harvey et al. (2012), each red light was attached to the PVC frame to the extreme right and left and center of the top crossbar. Red lights are predicted to have no impact on fish behavior (Harvey et al. 2012).

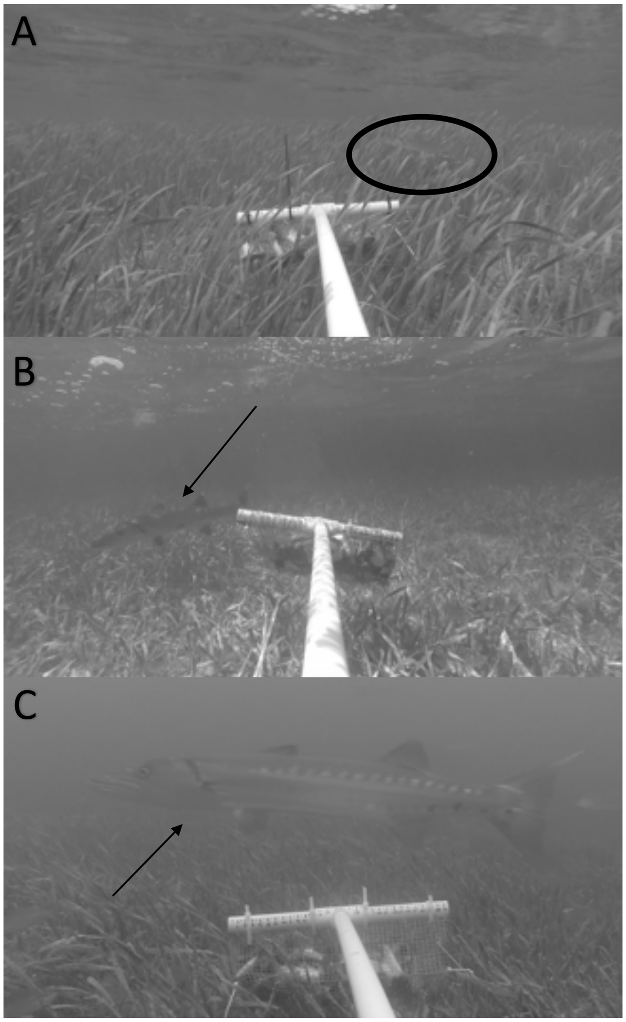


Figure 1.2. Life-history stages of Sphyraena barracuda estimated using a 30 cm scale bar. A) Juvenile, found within the oval; B) Late Juvenile; and C) Adult.

Sampling was conducted from August through October 2014. To discriminate diel patterns in fish activity, this study divided the day into four time categories; day (1200-1300 hrs), dark (2300-0000 hrs), dusk (30 min before and after sunset), and dawn (30 min before and after sunrise). Cameras were set facing the shoreline and left to record for 1 hour per sampling session.

To compare the distribution and abundance of predators between the mangrove-seagrass ecotone and adjacent seagrass habitats, sites were selected at random ranging from the mangrove-seagrass ecotone to 669 m from shore (Figure 1.1), which were separated into a nearshore (0-300 m) and farshore zone (301-700 m) based on a depth contour increase at 300 m from shore (Table 1.1). During sampling, random sites were sampled simultaneously at each time period, and the order of sites being sampled within the study area was randomly selected. A total of 45 random sites were chosen and sampled 4 times (day, dusk, dark, and dawn). A set of three BRUVS, each separated by 200 m, were deployed within each of the four time periods. A distance of 200 m separated each camera, to minimize the possibility of attracting fish from one camera to another (see Cappo et al. 2001).

Despite significantly low variation in water quality and benthic habitat between and among sites (Serafy et al. 2003), several abiotic conditions were measured biweekly in the study area. Environmental conditions measured included temperature (via thermometer), salinity (via refractometer), and depth via ruler. Additionally, benthic habitat characteristics (vegetation cover percentage and canopy height) at each site were measured once during the duration of the sampling period. To accomplish this, a 0.5 m x 0.5 m quadrat was randomly tossed four times within 15 m of each site and used to record vegetation cover percentage and canopy height following the approach of Hammerschlag et al. (2010b). Vegetation cover percentage, as defined in this study, was the fraction of the total quadrat area that was viewed from directly above. Vegetation height (cm) was measured within each quadrat, where the measurement point was selected randomly to minimize bias. Finally, the distance to shore was measured (via ArcGIS) from each randomly selected site.

1.2.3 BRUVS Analysis

Following the approach of Cappo et al. (2003), three metrics from each video recording were used: 1) number of species, used as a measure of diversity, 2) estimated length of fish species, and 3) maximum number of each individual species viewed at any one time (*MaxN*). This last metric reduces “double counts” of individuals and provides a conservative estimate of abundance.

Digital imagery recorded by the camera was downloaded to a laptop computer and stored at its original resolution (720p) as a .mov file. The length of the bait arm (0.75 m) was used to standardize the field of view from the footage. Fish observed outside this range were excluded from the data count to allow accurate species identification and control for the effect of visibility (Taylor et al. 2013). The 30 cm scale bar attached to the bait arm was used to estimate total fish length. Estimated length was used to assign fish to one of three life-history stages (see Table 1.2). Individuals smaller than "size at Age 1" from published age-and-growth studies were designated as juveniles, individuals larger than "size at Age 1" but smaller than "size at maturity" were classified as late juveniles, and individuals larger than "size at maturity" were classified as adults (Faunce and Serafy 2007). Based on published diet data, this study considered the following fishes tertiary consumers: great barracuda *Sphyraena barracuda* (Rafinesque, 1815, nurse shark *Ginglymostoma cirratum* (Bonnaterre, 1788), and lemon shark *Negaprion brevirostris* (Poey, 1868); whereas the following species were considered secondary consumers: mangrove snapper *Lutjanus griseus* (Linnaeus, 1758), lane snapper *Lutjanus synagris* (Linnaeus, 1758), yellowtail snapper *Ocyurus chrysurus* (Bloch, 1791), jack crevalle *Caranx hippos* (Linnaeus, 1766), and grunts, combined into *Haemulon* sp.

Species	Source ID	Size (cm)	Source	Edits	Input Size
Age-1					
L. griseus	1	9.51 TL	Manooch III & Matheson III, 1981	none	9.51 TL
L. synagris	2	13.4 TL	Manooch III & Mason, 1984	none	13.4 TL
O. chrysuru	3	16.0 TL	Garcia et al. 2003	none	16.0 TL
Haemulon sp.	4	10.5 FL	Billings & Munro, 1974	Mean 4 & 5	10.35 FL
	5	10.2 FL	Garcia-Arteaga, 1992		
C. hippos	6	20.0 FL	Snelson, 1992	none	20.0 FL
S. barracuda	7	37.8 FL	De Sylva, 1963	none	37.8
N.brevirostris	8	60.0 PCL	Gruber & Stout, 1983	none	60.0 PCL
G. cirratum	9	60.0 PCL	Castro, 2000	none	60.0 PCL
Size at maturity					
L. griseus	10	18.2 SL	Males: Domeier et al. 1996	Mean 12 & 13	19.5 SL
	11	19.8 SL	Females: Domeier et al. 1996		
	12	19.0 SL	Mean 10 & 11		
	13	20.0 SL	Both sexes: Starck, 1971		
L. synagris	14	18.0 FL	Thompson & Munro, 1983	Mean 14 & 17	21.0 FL
	15	24.5 FL	Females: Luckhurst et al. 2000		
	16	23.5 Fl	Males: Luckhurst et al. 2000		
	17	24.0 FL	Mean 15 & 16		
Haemulon sp	18	22.0 FL	Billings & Munro, 1974	none	22.0 FL
O. chrysuru	19	26.0 FL	Males: Thompson & Munro, 1974	none	28.0 FL
	20	30.0 FL	Females: Thompson & Munro, 1974		
	21	28.0 FL	Mean 19 & 20		
C. hippos	22	55.0 FL	Males: Thompson & Munro, 1974	none	60.5 FL
	23	66.0 FL	Females: Thompson & Munro, 1974		
	24	60.5 FL	Mean 22 & 23		
S. barracuda	25	58.0 FL	Females: de Sylva, 1963	none	52.0 FL
	26	46.0 FL	Males: de Sylva, 1963		
	27	52.0 FL	Mean 25 & 26		
N. brevirostris	28	225 TL	Males: Brown & Gruber, 1988	none	232.5
	29	240 TL	Females: Brown & Gruber, 1988		
	30	232.5 TL	Mean 28 & 29		
G. cirratum	31	227 TL	Females: Castro, 2000	none	220.6 TL
	32	214.3 TL	Males: Castro, 2000		
	33	220.6 TL	Mean 31 & 32		

Table 1.1. Information used to determine cut-off sizes for life-history stages used in analyses. Individuals less than the size at age-1 were defined as juveniles, those larger than the size at maturity were defined adults, and those in-between were defined as late juveniles.

1.2.4 Data Analysis

1.2.4.1 Environmental factors vs distance from shore

Generalized linear models (GLMs), were applied to examine abiotic and physical environmental variables; temperature, salinity, vegetation cover, canopy height, and depth on distance from shore. Models were first fitted to a Poisson distribution and then tested for over-dispersion. If models tested positive for over-dispersion, a negative binomial model was applied. GLMs were performed in statistical package R 3.2.2. (R Development Core Team 2008).

1.2.4.2 Distance gradient & diel period

Spatial patterns of fishes along the distance gradient (0-300 m from mangroves) were evaluated by comparing the *MaxN* of tertiary and secondary guild levels, species, and life-history stages of the following species: a) juvenile and late juvenile *L. griseus*, b) juvenile *O. chrysurus*, c) juvenile *L. synagris*, d) juvenile *C. hippos*, e) juvenile and late juvenile *S. barracuda*, f) juvenile *N. brevirostris*, and g) late juvenile *G. cirratum*. Data were positively skewed and zeros inflated, thus it was unfit for use in conventional parametric statistical analyses. Therefore, guild levels, species, and life-history stages for each diel period and distance combination were determined using a delta-distribution mean estimator (Fletcher et al. 2005), a measure of fish that separately considers the proportion of samples positive for a given assemblage component (i.e., frequency of occurrence) and the assemblage component mean when present (i.e., concentration). This approach was previously used to examine fish patterns in Biscayne Bay (e.g., Faunce and Serafy 2007, Serafy et al. 2007, Faunce and Serafy 2008, Hammerschlag and Serafy 2010). Using SAS statistical software (SAS Institute; Cary, NC, USA), this study regressed frequency of occurrence and concentration against distance from shore. Statistical significance was assessed at the $\alpha = 0.05$ level.

1.2.4.3 Abundance vs diel period

Zero-inflated negative binomial models, a class of generalized linear mixed models (GLMMs), were applied to examine effects of time periods (dark, dawn, day, and dusk) and distance zones (nearshore and farshore) on the abundance (*MaxN*) of all species, life-history stages and trophic levels. Models were first fitted to a Poisson distribution and then tested for over-dispersion. If models tested positive for over-dispersion, a negative binomial model was applied. Site was included as a random effect for models investigating abundance (*MaxN*) between time periods to account for any inherent differences among sites. To avoid spurious significance from the series of pairwise tests between time periods, the Bonferroni correction was applied to the significance level.

1.2.4.4 Abundance nearshore vs farshore zones

Zero-inflated negative binomial models, were applied to examine effects of zones (nearshore and farshore) on the abundance (*MaxN*) of trophic guilds, species, and life-history stages. Models were first fitted to a Poisson distribution and then tested for over-dispersion. If models tested positive for over-dispersion, a negative binomial model was applied. For models testing nearshore versus farshore sites, time and site were included as random effects to account for any unwanted variance in time of day within the nearshore versus farshore samples. To avoid spurious significance from the series of pairwise tests between time periods, the Bonferroni correction was applied to the significance level. All zero-inflated negative binomial models were performed in statistical package R 3.2.2 (R Development Core Team 2008).

1.3 Results

1.3.1 General

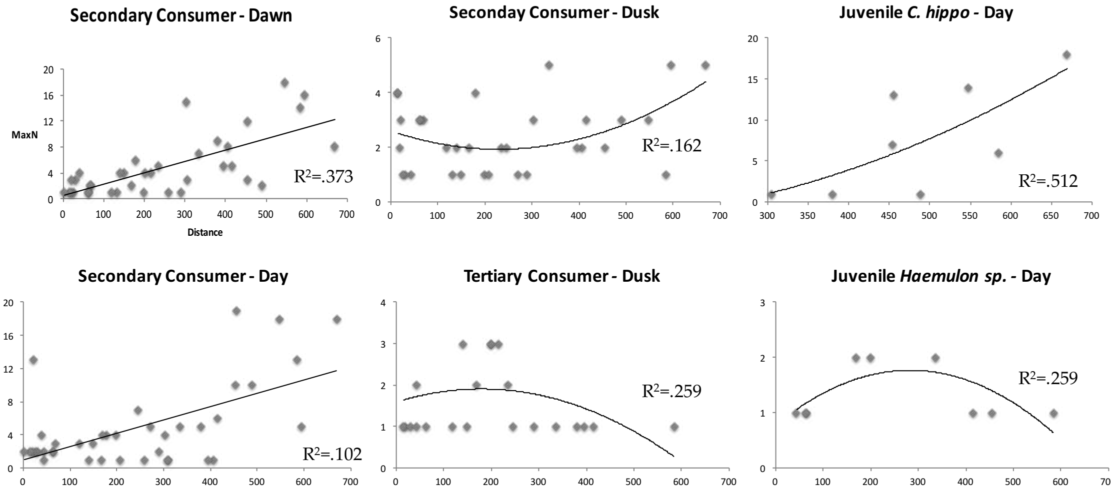
A total of 171 BRUV deployments (43 dusk, 43 day, 42 dawn, and 43 dark, with nine excluded due to mechanical error) yielded 1,339 fishes observed: 136 tertiary consumers, 475 secondary consumers, and 723 primary consumers (five fish could not be identified). GLMs revealed no significant difference in relationships between environmental factors and distance from shore, with the only exception being depth (Table 1.1). Consistency in the physical an abiotic factors permitted us to reduce dissimilarity in environmental factors that could confound predator distribution.

	Environmental variables				
	Temperature ©	Salinity	Vegetation cover (%)	Canopy height (cm)	Depth (cm)
Total	31.6 (.05)	37 (.03)	83.6 (1.7)	30.4 (1.05)	**71.5 (3.7)**
0-300	31.9 (.06)	37.6 (.09)	82.19 (2.5)	30.4 (1.6)	**57.4 (3.2)**
300-700	31.3 (.08)	36.4 (.1)	86.36(4.6)	30.4 (1.3)	**100 (3.2)**

Table 1.2. Mean of abiotic and physical environmental variables measured at the study area; temperature, salinity, vegetation cover, canopy height, and depth. GLMs revealed no significant difference in relationships between environmental factors and distance from shore, with the only exception being depth. Bolded means indicate significance (p<0.05), and standard error is listed in parentheses.

1.3.2 Distance gradient & diel period

This chapter analyzed the concentration and frequency of occurrence of two trophic guilds, nine species, with two species having multiple life-history stages across the distance gradient at dawn, day, dusk and dark resulting in a total of 101 comparisons. For 88 of 101 comparisons, this study found that fish concentration and frequency of occurrence were uniform across the distance. The exceptions were the following for concentration-distance patterns: (I) increased linearly – secondary consumers (dawn, day), juvenile *C. hippos* (day); (II) parabolic – secondary consumers (dusk), tertiary consumer (dusk), *Haemulon* sp. (day). The exceptions of frequency of occurrence-distance patterns were the following: (I) increasing linearly – juvenile *O. chrysurus* (dawn), juvenile *L. synagris* (dawn, day), juvenile *C. hippos* (dawn, day), (II) decreasing linearly – *L. griseus* (day), late juvenile *L. griseus* (day). A summary of the results can be found in Tables 1.3-1.7, as well as in Figures 1.3 and 1.4.


Secondary Consumer - Dawn

Seconday Consumer - Dusk

Juvenile *C. hippo* - Day

Secondary Consumer - Day

Tertiary Consumer - Dusk

Juvenile *Haemulon sp.* - Day

Figure 1.3. Relative density-distance patterns for significant models determined at the alpha level (p<0.05). All other trophic guilds, species and species life-history stages not depicted below are uniformly distributed over the seagrass gradient.

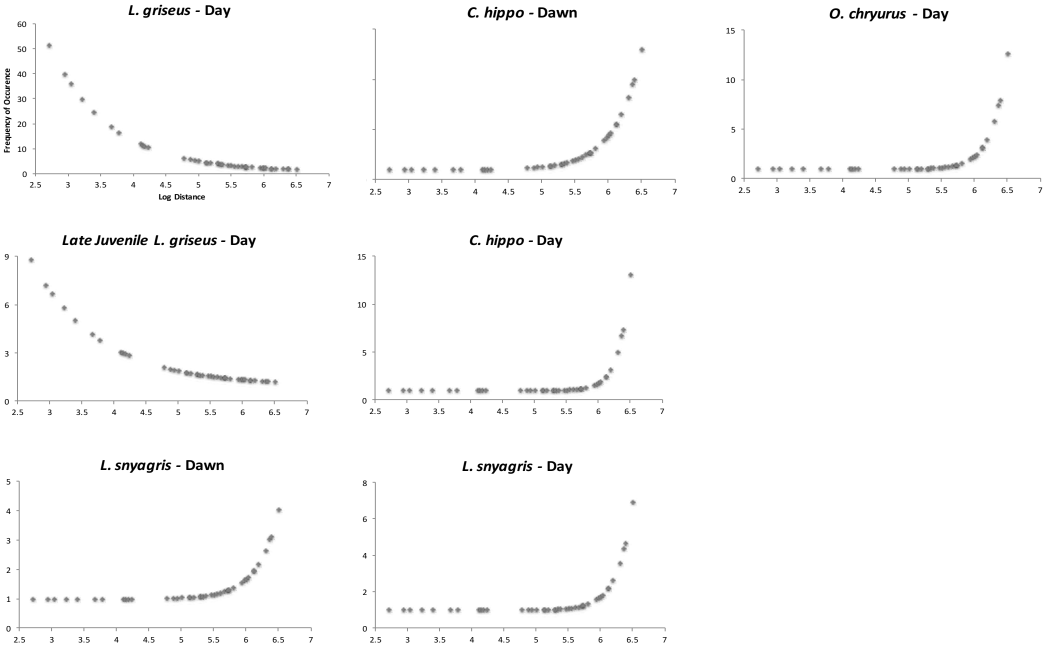


Figure 1.4. Frequency of occurrence patterns for significant models determined at the alpha level (p<0.05). All other trophic guilds, species, and species life-history stages not depicted below are uniformly distributed over the seagrass gradient

1.3.3 Abundance vs diel period

There was a significant decline in total predators recorded during dark and dusk in comparison to dawn and day. Tertiary consumers were more abundant (*MaxN*) during the day than at dark. *Sphyraena barracuda* were more abundant during day than at any other time period. Juvenile *S. barracuda* abundance did not change over the sampling period, while late juveniles were more abundant during the day than at any other time period. *Ginglymostoma cirratum* and *N. brevirostris* abundances did not significantly change over the sampling period, but small peaks at dawn and dusk were observed. A summary of the results can be found in Figures 1.5-1.7.

Secondary consumers were more abundant (*MaxN*) during dawn and day than dusk and dark. *Lutjanus griseus* were more abundant (*MaxN*) during dawn, day, and dusk compared to dark. Late juvenile *L. griseus* abundance (*MaxN*) followed this same pattern. Juvenile *L. griseus* were more abundant (*MaxN*) during day than at dark. Juvenile *O. chrysurus* were more abundant (*MaxN*) during dawn than dusk and dark while juvenile *L. synagris* and were more abundant (*MaxN*) during dawn and day compared to dark. Juvenile *C. hippos* also showed this same pattern and were more abundant (*MaxN*) during dawn and day. The abundance (*MaxN*) of *Haemulon* sp. did not significantly change over the sampling period, but small peaks at day and dusk were observed. A summary of these results can found in Figures 1.5, 1.8, and 1.9.

Tropic Guild	Time	Metric	Model	c-value F-value	Significance	Relationship
Secondary Consumer	Dark	Occurrence	logistic	0.624	ns	
		Concentration	linear	0.94	ns	
		Concentration	parabolic	0.5	ns	
	Dawn	Occurrence	logistic	0.565	ns	
		Concentration	linear	20.9	***	Positive
		Concentration	parabolic	13.95	***	Negative
	Day	Occurrence	logistic	0.635	ns	
		Concentration	linear	4.21	*	Positive
		Concentration	parabolic	4.31	ns	
	Dusk	Occurrence	logistic	0.573	ns	
		Concentration	linear	0.11	ns	
		Concentration	parabolic	2.9	*	Positive
Tertiary Consumer	Dark	Occurrence	logistic	0.695	ns	
		Concentration	linear	1.51	ns	
		Concentration	parabolic	0.81	ns	
	Dawn	Occurrence	logistic	0.541	ns	
		Concentration	linear	1.1	ns	
		Concentration	parabolic	2.78	ns	
	Day	Occurrence	logistic	0.638	ns	
		Concentration	linear	0.06	ns	
		Concentration	parabolic	0.4	ns	
	Dusk	Occurrence	logistic	0.431	ns	
		Concentration	linear	0.01	ns	
		Concentration	parabolic	3.34	*	Negative

Table 1.3. Summary of linear and quadratic models results for secondary and tertiary consumers regressed against distance from shore by time period for secondary and tertiary consumers. Asterisks indicate significance level (* = p<0.05, ** = p<0.01, *** = p<0.001), non-significant models are labeled as “ns”.

Species	Time	Metric	Model	c-value F-value	Significance	Relationship
L. griseus	Dark	Occurrence	logistic	0.515	ns	
		Concentration	linear	0.75	ns	
		Concentration	parabolic	0.34	ns	
	Dawn	Occurrence	logistic	0.637	ns	
		Concentration	linear	0.01	ns	
		Concentration	parabolic	0.66	ns	
	Day	Occurrence	logistic	0.746	*	Negative
		Concentration	linear	2.37	ns	
		Concentration	parabolic	1.57	ns	
	Dusk	Occurrence	logistic	0.488	ns	
		Concentration	linear	3.19	ns	
		Concentration	parabolic	1.97	ns	
Juvenile	Dark	Occurrence	logistic	0.504	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
	Dawn	Occurrence	logistic	0.599	ns	
		Concentration	linear	0.04	ns	
		Concentration	parabolic	0.48	ns	
	Day	Occurrence	logistic	0.537	ns	
		Concentration	linear	0.59	ns	
		Concentration	parabolic	0.99	ns	
	Dusk	Occurrence	logistic	0.591	ns	
		Concentration	linear	0.96	ns	
		Concentration	parabolic	0.55	ns	
Late Juvenile	Dark	Occurrence	logistic	0.53	ns	
		Concentration	linear	0.82	ns	
		Concentration	parabolic	0.37	ns	
	Dawn	Occurrence	logistic	0.593	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	0.4	ns	
	Day	Occurrence	logistic	0.766	**	Negative
		Concentration	linear	0.05	ns	
		Concentration	parabolic	0.6	ns	
	Dusk	Occurrence	logistic	0.543	ns	
		Concentration	linear	1.85	ns	
		Concentration	parabolic	0.89	ns	

Table 1.4. Summary of linear and quadratic models results regressed against distance from shore by time period for L. griseus, juvenile L. griseus, and late juvenile L. griseus. Asterisks indicate significance level (* = $p<0.05$, ** = $p<0.01$, *** = $p<0.001$), and non-significant models are labeled as “ns”. Number signs (#) indicate models that could not detect differences because an abundance of one across the distant gradient.

1.3.4 Abundance nearshore vs farshore zone

The total abundance (*MaxN*) of predators was higher in the farshore zone than the nearshore zone, which is influenced by the higher abundance (*MaxN*) of secondary consumers present in the farshore zone. Tertiary consumers were more abundant (*MaxN*) in the nearshore zone. *Lutjanus griseus* and late juvenile *L. griseus* were more abundant (*MaxN*) in the nearshore zone, while juvenile *L. griseus* abundance (*MaxN*) did not differ between zones. Juvenile *O. chrysurus*, *L. synagris*, and *C. hippos* were all significantly more abundant (*MaxN*) in the farshore zone. *Haemulon* sp. were evenly distributed between the two zones. Juvenile and late juvenile *S. barracuda* were evenly distributed across zones, although adult *S. barracuda* were more abundant (*MaxN*) in the farshore zone (n=7; 6 farshore versus 1 nearshore). *Ginglymostoma cirratum* and *N. brevirostris* were more abundant (*MaxN*) in the nearshore zone. A summary of the results can be found in Tables 1.8 and 1.9.

Species	Time	Metric	Model	c-value F-value	Significance	Relationship
Juvenile *O. chrysurus*	Dark	Occurrence	logistic	$		
		Concentration	linear	$		
		Concentration	parabolic	$		
	Dawn	Occurrence	logistic	0.846	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
	Day	Occurrence	logistic	0.889	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
	Dusk	Occurrence	logistic	$		
		Concentration	linear	$		
		Concentration	parabolic	$		
Juvenile *L. synagris*	Dark	Occurrence	logistic	$		
		Concentration	linear	$		
		Concentration	parabolic	$		
	Dawn	Occurrence	logistic	0.75	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
	Day	Occurrence	logistic	$		
		Concentration	linear	$		
		Concentration	parabolic	$		
	Dusk	Occurrence	logistic	$		
		Concentration	linear	$		
		Concentration	parabolic	$		
Juvenile *Haemulon sp*	Dark	Occurrence	logistic	0.433	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
	Dawn	Occurrence	logistic	0.61	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
	Day	Occurrence	logistic	0.487	ns	
		Concentration	linear	46.5	**	Positive
		Concentration	parabolic	37.34	ns	
	Dusk	Occurrence	logistic	0.672	ns	
		Concentration	linear	1.58	ns	
		Concentration	parabolic	0.98	ns	
Late Juvenile *C. hippos*	Dark	Occurrence	logistic	$		
		Concentration	linear	$		
		Concentration	parabolic	$		
	Dawn	Occurrence	logistic	0.8	ns	
		Concentration	linear	0.06	ns	
		Concentration	parabolic	0.18	ns	
	Day	Occurrence	logistic	$		
		Concentration	linear	$		
		Concentration	parabolic	$		
	Dusk	Occurrence	logistic	0.704	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	

Table 1.5. Summary of linear and quadratic models results regressed against distance from shore by time period for juvenile O. chrysurus, juvenile L. synagris, juvenile Haemulon sp., and juvenile C. hippos. Asterisks indicate significance level (* = p<0.05, ** = p<0.01, *** = p<0.001), and non-significant models are labeled as “ns”. Number signs (#) indicate models that could not detect differences because an abundance of one across the distant gradient. Dollar signs ($) indicate models where fish were absent for that time period.

1.4 Discussion

Diel sampling along a distance gradient from the mangrove edge across adjacent seagrass habitat (0-700 m) revealed that distribution patterns (concentrations and frequency of occurrences) for trophic guilds, species, and species life-history stages are mostly uniform or increase linearly with distance from shore, (88 of 103 comparisons). My results contradict the findings of Jelbart et al. (2007) and Unsworth et al. (2008), who reported patterns of decreasing density with increasing distance from shore. The difference may be attributed to the focus on predator-species compared to the whole fish assemblage investigated in the other studies. This chapter did find that for frequency of occurrence, *L. griseus* and late juvenile *L. griseus* were the only secondary consumer species and life-history stages to decrease with distance from shore, a result consistent with prior diurnal studies of this species in Biscayne Bay (e.g., Luo et al. 2009).

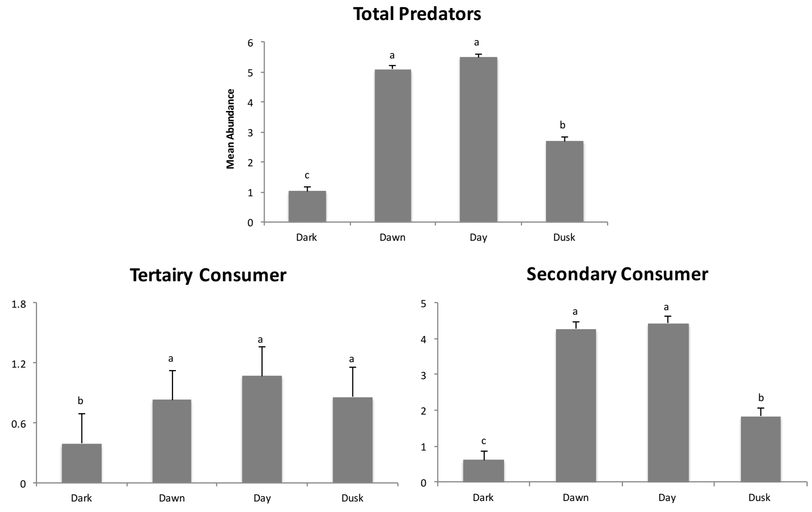


Figure 1.5. Abundance (MaxN) rates for all predators (Total Abundance), secondary, and tertiary consumers across diel periods. Letters indicate significantly different groupings at P<0.05 adjusted with the Bonferroni correction for multiple comparisons; same letter=no difference.

The lack of secondary consumer abundance (*MaxN*) at dark (2300-0000 hrs) in the seagrass beds was surprising. *Haemulon* sp. and *Lutjanus* sp. are supposedly nocturnal foragers that have been shown to migrate from the mangroves at sunset to feed into adjacent seagrass beds at night (Nagelkerken et al. 2001, Appeldoorn et al. 2009, Luo et al. 2009, Hammerschlag and Serafy 2010). The high abundance (*MaxN*) of *Haemulon* sp. observed during the day can be attributed to schools of small early juveniles (>3 cm, displaying juvenile livery) harboring in the seagrass beds. Although in lower abundance (*MaxN*), larger early juveniles (< 3 cm, displaying adult livery) were only documented at dusk and dark and conspicuously absent during the day. Therefor, the abundance (MaxN) of large juvenile *Haemulon* sp. offshore may reflect an offshore migration, which has been documented by Ogden and Quinn (1984) and Burke (1995). *Lutanus* sp. were more abundant (*MaxN*) at dawn and day, suggesting that *Lutjanus* sp. are not migrating into the study site to feed at night. However, on reef habits, *L. griseus* snapper feed up to 1.6 km from diurnal resting spots, which is beyond the distance limits

Species	**Time**	**Metric**	**Model**	**c-value F-value**	**Significance**	**Relationship**
S. barracuda	Dark	Occurrence	logistic	0.512	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
	Dawn	Occurrence	logistic	0.616	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
	Day	Occurrence	logistic	0.538	ns	
		Concentration	linear	0.69	ns	
		Concentration	parabolic	0.51	ns	
	Dusk	Occurrence	logistic	0.698	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	0.74	ns	
Juvenile	Dark	Occurrence	logistic	0.512	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
	Dawn	Occurrence	logistic	0.603	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
	Day	Occurrence	logistic	0.607	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
	Dusk	Occurrence	logistic	0.554	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
Late Juvenile	Dark	Occurrence	logistic	$	ns	
		Concentration	linear	$	ns	
		Concentration	parabolic	$	ns	
	Dawn	Occurrence	logistic	0.488	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
	Day	Occurrence	logistic	0.537	ns	
		Concentration	linear	0.91	ns	
		Concentration	parabolic	0.79	ns	
	Dusk	Occurrence	logistic	0.756	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	

Table 1.6. Summary of linear and quadratic models results regressed against distance from shore by time period for S. barracuda, juvenile S. barracuda, and late juvenile S. barracuda. Asterisks indicate significance level (* = p<0.05, ** = p<0.01, *** = p<0.001), and non-significant models are labeled as "ns". Number signs (#) indicate models that could not detect differences because an abundance of one across the distant gradient. Dollar signs ($) indicate models were fish where absent for that time period.

of this study (Starck & David 1966). Hence, it is possible that late juvenile gray snapper may be harboring in the seagrass beds during dawn and day and traveling further offshore at night beyond the distance of the study area investigated.

Species	**Time**	**Metric**	**Model**	**c-value F-value**	**Significance**	**Relationship**
Juvenile *N. brevirostris*	Dark	Occurrence	logistic	0.6	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
	Dawn	Occurrence	logistic	0.449	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
	Day	Occurrence	logistic	0.417	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
	Dusk	Occurrence	logistic	0.062	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	#	ns	
Late Juvenile *G. cirratum*	Dark	Occurrence	logistic	0.695	ns	
		Concentration	linear	1.87	ns	
		Concentration	parabolic	0.8	ns	
	Dawn	Occurrence	logistic	0.663	ns	
		Concentration	linear	0.13	ns	
		Concentration	parabolic	0.07	ns	
	Day	Occurrence	logistic	0.607	ns	
		Concentration	linear	#	ns	
		Concentration	parabolic	0.4	ns	
	Dusk	Occurrence	logistic	0.658	ns	
		Concentration	linear	1.5	ns	
		Concentration	parabolic	0.67	ns	

Table 1.7. Summary of linear and quadratic models results regressed against distance from shore by time period for juvenile N. brevirostris and late juvenile G. cirratum. Asterisks indicate significance level (* = p<0.05, ** = p<0.01, *** = p<0.001), and non-significant models are labeled as "ns". Number signs (#) indicate models that could not detect differences because an abundance of one across the distant gradient. Dollar signs ($) indicate models were fish where absent for that time period.

The high abundances (*MaxN*) at dawn and day in the farshore zone and the distribution patterns, frequency of occurrence increasing with distance, displayed by *C. hippos, L, synagris, and O. chrysurus* were unexpected, as previous studies have found lower densities of fish species within or near the mangroves at dark compared to day (Rooker and Dennis 1999, Nagelkerken et al. 2000, Christian 2003). *Caranx hippos*, which feed on benthic invertebrates and fish, are considered diurnal predators (Kwei 1978, Saloman and Naughton 1984). *Ocyurus chrysurus* and *L. synagris*, both of which feed on benthic invertebrates and fish, are considered nocturnal predators (Pauly and Froese 1996, Franks and VanderKooy 2015). In this study, all three species were more abundant (*MaxN)* during dawn and day and absent at dark. Stomach content analysis of *L. synagris* and *O. chrysurus* detailed by Starck (1971) revealed that both species had stomach contents throughout the day time period, indicating diurnal feeding. This may indicate that the farshore zone has a higher density of this prey, but based on the results of Hammerschlag et al. (2010a), these species are most likely occupying the farshore zone to avoid predation by tertiary consumers in the nearshore zone, such as sharks. In addition, the lack of small juvenile *Haemulon* sp. and *Lutjanus* sp. at dark (2300-0000 hrs) suggests that small juvenile *Haemulon* sp. and *Lutjanus* sp. are diurnal and seek refuge in the seagrass bed at dark. This chapter's observations highlight the importance of seagrass beds as daytime feeding habitats for small juvenile *Lutjanus* sp. and *Haemulon* sp. within the mangrove-seagrass continuum.

Model	**Parameter**	**Coefficient**	**SE**	**Wald Z**	***p* value**
total abundance	Intercept	2.203	0.277	7.96	1.80E-15
	Nearshore	-0.468	0.119	-3.93	**8.60E-05**
secondary consumers	Intercept	1.242	0.402	3.09	0.002
	Nearshore	-1.006	0.179	0.179	**2.10E-08**
tertiary consumers	Intercept	-0.965	0.265	-3.64	0.0003
	Nearshore	0.89	0.24	3.71	**0.0002**

Table 1.8. Abundances (MaxN) for all predators (Total Abundance), secondary, and tertiary consumers compared across nearshore and farshore zones. Predators overall were more abundant in the farshore, but tertiary consumers as a guild were more abundant in the nearshore.

In this study, tertiary and secondary consumer abundances (*MaxN*) differed greatly between zones. Likewise, at dusk, tertiary and secondary consumer distributions (concentration) showed an opposite parabolic relationship, with a peaks and lows between 200 – 300 m (Figure 1.10). I believe, this area between 200 – 300 m is an area of high predation and may act as transition zone between the shallow, nearshore seagrass and the deeper, farshore seagrass beds. At the study site, the depth is fairly uniform till 300 m from shore. But, after 300 m, the depth increases with distance (Figure 1.11). Transition zones are high risk areas acting as hunting corridors for predators in an assortment of settings and have been previously reported in mangrove-seagrass habitats (Hammerschlag et al. 2010a).

Model	**Parameter**	**Coefficient**	**SE**	**Wald Z**	***p* value**
L. griseus	Intercept	-0.427	0.314	-1.36	0.174
	Nearshore	0.4	0.199	2.01	**0.045**
(juvenile)	Intercept	-1.577	0.391	-4.03	5.50E-05
	Nearshore	0.123	0.324	0.38	0.7
(late juvenile)	Intercept	-0.785	0.319	-2.46	0.014
	Nearshore	0.486	0.247	1.97	**0.049**
S. barracuda	Intercept	-1.317	0.477	-2.76	0.0058
	Nearshore	-0.211	0.281	-0.75	0.4515
(juvenile)	Intercept	-2.979	0.577	-5.16	2.50E-07
	Nearshore	0.745	0.646	1.15	0.25
(late juvenile)	Intercept	-1.593	0.289	-5.52	3.50E-08
	Nearshore	-0.181	0.369	-0.49	0.62
O. chrysuru	Intercept	-0.337	1.004	-0.34	0.74
(juvenile)	Nearshore	-2.956	0.597	-4.95	**7.50E-07**
C. hippos	Intercept	-0.813	1.288	-0.63	0.53
(juvenile)	Nearshore	-3.16	0.667	-4.74	**2.20E-06**
L. synagris	Intercept	-1.548	0.646	-2.4	0.1652
(juvenile)	Nearshore	-3.567	1.026	-3.48	**0.0005**
Haemulon sp	Intercept	-1.422	0.331	-4.29	1.80E-05
(juvenile)	Nearshore	-0.382	371	-1.03	0.3
N. brevirostris	Intercept	-16.3	626.3	-0.03	0.98
(juvenile)	Nearshore	15.2	626.3	0.02	0.98
G. cirratum	Intercept	-3.03	0.68	-4.45	8.40E-06
(late juvenile)	Nearshore	2.2	0.64	3.44	**0.0006**

Table 1.9. Abundance (MaxN) rates for the most common species compared across nearshore and farshore zones. Lutjans griseus and S. barracuda were both modeled for all life-history stages combined, and then separated for juveniles and late juveniles. A model for adult S. barracuda did not converge due to low sample size (n = 7). Bolded p-values indicate significance.

The low abundance of tertiary consumers (*MaxN*) at dark (2300-0000 hrs) appeared to be largely driven by the absence of *S. barracuda* compared to their high daytime (1200 - 1300 hrs) abundance in the seagrass. In contrast, *N. brevirostris* and *G. cirratum* were present throughout the diel cycle. In a nearby seagrass bed within Biscayne Bay, Hammerschlag et al. (2010a) reported that predation rates on tethered fish were twice as high during the night than during the day. It is thus possible that these increased predation rates at night may not be due to an increase in predator abundance, but rather increased feeding activity by predators at night optimizing probability of prey capture (Smith et al. 2011).

Sphyraena barracuda in all three life-history stages were seen at dark (2300-0000 hrs) in the seagrass beds (pers. obs.), but only documented on the BRUVS cameras twice. Thus, this study concludes that *S. barracuda* are present in the area, but not actively foraging at night, most likely because their ability to hunt is impaired by low light conditions (de Sylva 1963). Adult *S. barracuda* were not analyzed as a separate life-

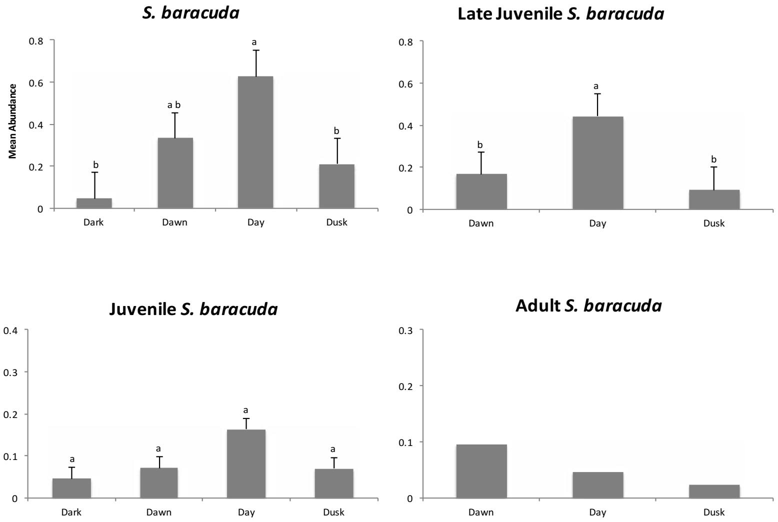


Figure 1.6. Mean abundance (MaxN) rates for S. barracuda, juvenile S. barracuda, late juvenile S. barracuda and adult S. barracuda compared across diel periods. Late juvenile S. barracuda and adult were not present at dark. Letters indicate significantly different groupings at the alpha level (0.05) adjusted with the Bonferroni correction for multiple comparisons; same letter=no difference. A model for adult S. barracuda did not converge due to low sample size (n = 7).

history stage due to a small sample size (n=7), but clear patterns emerged as over 80% of adult *S. barracuda* were documented offshore at dawn. Therefore, adults were more abundant (*MaxN*) offshore at dawn, while juvenile and late juvenile *S. barracuda* were evenly distributed over the seagrass bed (Table 1.9). Adult *S. barracuda* are most likely positioning themselves to take advantage of prey such as the emerging abundance of *O. chrysurus*, *L. synagris*, and *C. hippos* that appear offshore in the morning (Figure 1.3 and 1.4).

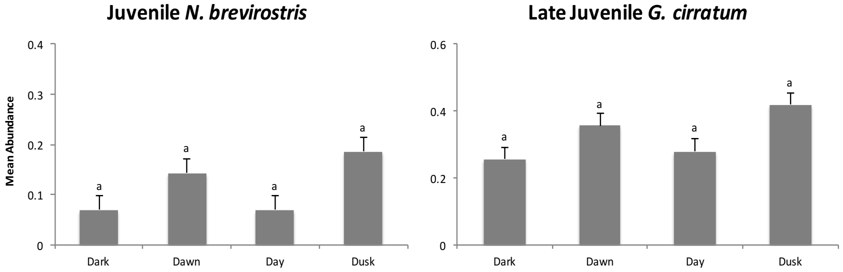


Figure 1.7. Mean abundance (MaxN) rates for juvenile Negaprion brevirostris and late juvenile G. cirratum sharks compared across diel periods. Letters indicate significantly different groupings at the alpha level (0.05) adjusted with the Bonferroni correction for multiple comparisons; same letter=no difference.

The recent decline in *S. barracuda* populations in the south Florida region is now being investigated by the Florida Fish and Wildlife Commission. It has been well documented that juvenile *S. barracuda* occur in high densities in Biscayne Bay (Serafy et al. 2003, Faunce and Serafy 2008). After reaching a certain size (*ca.* 500 mm), individuals are thought to move from the mangrove-seagrass to reef-dominated habitat (Christine 2010). However, in this study, *S. barracuda* of all three life-history stages were documented and not uniformly distributed with distance from shore. This highlights the importance of the mangrove-seagrass habitat not only as a nursery habitat for this species, but also as a juvenile and adult habitat. Therefore, any degradation or loss of seagrass habitat in Biscayne Bay will impact all three life-history-stages of this species. For that reason, both mangroves and seagrass beds may need to be considered for Essential Fish Habitat designation in any potential federal management plans *S. barracuda*.

G. cirratum and *N. brevirostris* are generally considered nocturnal predators that forage at night (Tanka 1973, Correia et al. 1995). However, my results do not support this claim. Both *G. cirratum* and *N. brevirostris* were present throughout the diel cycle and had abundances (*MaxN*) that were lowest at dark (2300-0000 hrs) and highest during dawn and dusk (Figure 1.7). Thus, in the shallow seagrass habitats, these sharks are more likely crepuscular than nocturnal, which is consistent with the observations of Gruber (1982). *N. brevirostris* were the only species that were not documented in both nearshore and farshore zones, appearing solely in the nearshore zone. Late juvenile *G. cirratum* also avoided the deeper water offshore, as they were not documented past 380 m. In addition to potential increased prey capture by predators at the mangrove-seagrass ecotone (Hammerschlag et al. 2010a,b), the nearshore zone is shallower (Table 1.1) and may reduce their vulnerability to large predatory sharks. *N. brevirostris* in other nursery areas have been shown to harbor close to the mangroves to avoid predation from larger sharks (Franks 2007, Stump 2013).

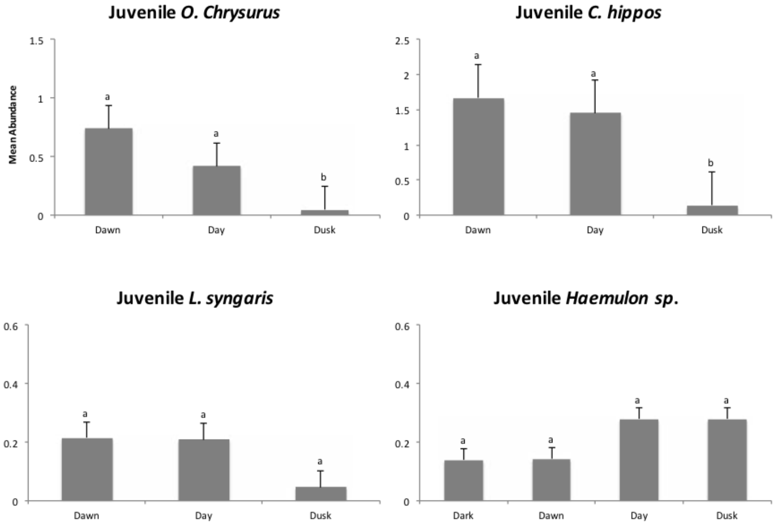
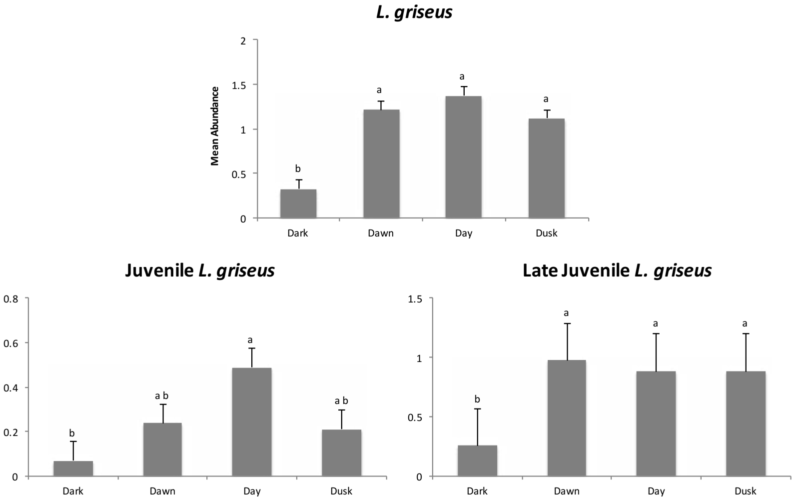


Figure 1.8. Mean abundance (MaxN) rates for juvenile Ocyurus chrysurus, juvenile Lutjanus synagris, juvenile C. hippos and juvenile Haemulon sp. compared across diel periods. Letters indicate significantly different groupings at the alpha level (0.05) adjusted with the Bonferroni correction for multiple comparisons; same letter=no difference.


L. griseus

Juvenile *L. griseus*

Late Juvenile *L. griseus*

Figure 1.9. Mean abundance (MaxN) rates for total L. griseus, juvenile L. griseus and late L. griseus abundances compared across diel periods. Letters indicate significantly different groupings at the alpha level (0.05) adjusted with the Bonferroni correction for multiple comparisons; same letter=no difference.

It is worth noting that the sampling occurred at distinct times in the diel cycle. Thus, it is possible that fish distributions during these times may not be the same as later or earlier in the sampling period. For example, fish distribution during the nocturnal sampling window, between 12 PM and 1 AM local time, may differ earlier or later in the night. This caveat applies to the other sampling periods as well. It is also worth considering that the results are based on sampling that used the presence of light to illuminate the field of view. Hence, it is possible that fish were deterred or attracted by the presence of artificial light during dawn, dusk, and dark (Carazo et al. 2013, Fitzpatrick et al. 2013). Specifically, the lower abundance (*MaxN*) of fish at dark (2300-0000 hrs) may be a species specific reactions to artificial lighting (Harvey et al. 2012). For example, *Solea senegalensis* (Kaup, 1858) exposed to high intensity red lights at night prompted behaviors of fear and escape (Carazo et al. 2013). Furthermore, plume dispersal or area of attraction is an unknown confounding factor in this study.

In summary, the investigation into the distribution and abundance of predators among seagrass and mangrove habitats revealed that the predatory fish assemblage varies over both the diel cycle and mangrove-seagrass distance gradient. Studies that do not

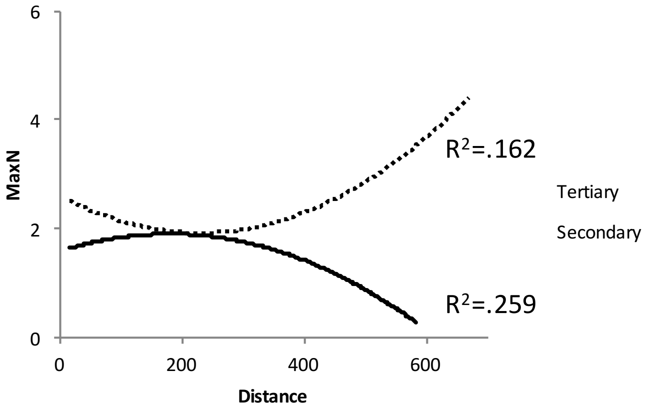


Figure 1.10. Relative density-distance patterns for tertiary and secondary consumer models at dusk, models determined at the alpha level ($p<0.05$).

take these spatiotemporal patterns into account may generate inaccurate estimates for both population abundance and utilization of specific habitats, which could negatively affect the effectiveness of subsequent management measures. Seagrass beds provide a very important nursery function for a variety of economically and ecologically important fishes. However, anthropogenic development continues to destroy these habitats on a global scale (Waycott et al. 2009). Understanding how fish are distributed across the mangrove-seagrass habitat is vital to the conservation and management of species. The results of this chapter, suggest the degradation or loss of seagrass habitat along a distance gradient could differentially impact the life-history stages of species differently.

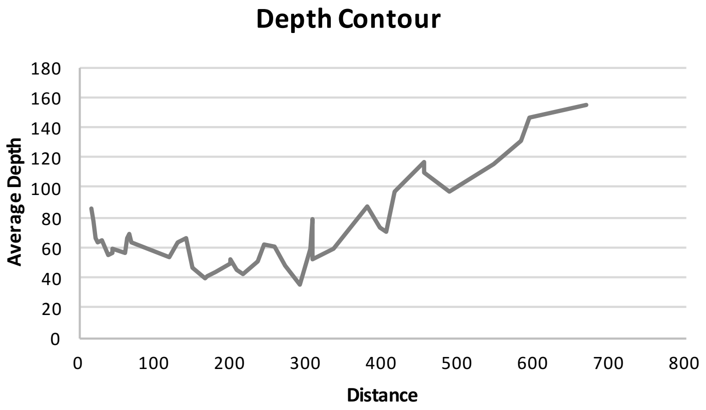


Figure 1.11. Average depth across the distance gradient. The depth is fairly uniform till 300 m from shore. But, after 300 m, the depth increases with distance.

2. Diel patterns of fishes through a channel linking offshore coral reefs and inshore seagrass habitats

2.1 Background

Understanding the connectivity between offshore coral reefs and coastal estuaries is a research priority for the conservation management of tropical fishes and coastal ecosystems (e.g., Chin et al. 2013). In particular, seagrass beds and mangrove forests serve as nurseries for juvenile fishes, providing food and shelter for numerous economically and ecologically important reef species (Beck et al. 2001, Nagelkerken et al. 2000, Nagelkerken et al. 2001, Nagelkerken et al. 2002, Serafy et al. 2003, Adams et al. 2006, Nagelkerken et al. 2008, Berkström et al. 2013a,b). After reaching a certain size, some fishes leave the protection of the nursery habitat and make an ontogenetic shift to offshore coral reefs, serving as adult habitat (Rooker and Dennis 1991, Kimirei et al. 2011). The proximity of mangrove and seagrass habitats to coral reefs has a positive influence on fish abundances that occupy the reef habitat (Nagelkerken et al. 2002, Dorenbosh et al. 2004, Mumby et al. 2004). In the extreme case, loss of mangrove habitats can lead to local extinction (Mumby et al. 2004). Moreover, Honda et al. (2013) reported that over a third of the commercial fish species they documented in their study utilized seagrass or mangrove or both habitats or one those habitats in combination with coral reefs.

Many nocturnally active reef fishes shelter on coral reefs by day and migrate into adjacent feeding sites at night (Hobson 1965, Ogden and Ehrlich 1977, Rooker and Dennis 1991, Nagelkerke et al. 2000); whereas, many diurnally active reef fishes shelter on coral reefs at night and migrate into adjacent feeding sites during the day (Ogden and Zieman 1977, Maciá & Robinson 2005, Krumme 2009, Garcia et al. 2015). It is generally believed that inshore tropical seagrass beds provide feeding grounds for numerous reef species, such as lutjanid snappers and haemulid grunts at night (Jelbart et al. 2007, Luo et al. 2009). However, this presumption is primarily inferred from relative abundance of reef fishes in seagrass beds (Robblee et al. 1984, Beets at al. 2003, Nagelkerken et al. 2000, Mumby et al. 2004, Berkström et al. 2013a), stable isotope analysis (Nagelkerken et al. 2000, Nagelkerken et al. 2008), and gut content analysis (Nagelkerken et al. 2008, Berkström et al. 2013a). Despite higher abundances of reef fishes in seagrass beds at night, there are only a few studies which have provided direct evidence of reef fish movements to and from nearby reefs into seagrass beds (Beets et al. 2003, Friedlander and Monaco 2007, Hitt et al. 2011b). It is believed that the timing of diel migrations to seagrass beds from coral reefs or mangroves generally occurs around crepuscular periods although relatively few direct measurements of such behaviors have been obtained (Hobson 1965, Ogden and Ehrlich 1979, McFarland et al. 1979, Luo et al. 2009, Appeldoorn et al. 2009, Hitt et al. 2011a,b).

Efforts to protect habitats with high connectivity require an understanding of fish migrations and pathways. Given the biological connection between offshore coral reefs and coastal estuaries, any degradation in one habitat can have consequences well beyond its boundaries. Thus, there is a critical need to understand the underlying processes that determine movement patterns between coastal estuaries and coral reef habitats. Such studies are particularly important in Biscayne Bay, a large and semi-enclosed body of water located in subtropical southeast Florida, USA (Figure 2.1). Given its location near the large urban region of Miami, the Bay continues to be impacted by several anthropogenic effects, including the restructuring of historical feeder waterways from the Everglades, the destruction of fringing mangrove stands, and contamination via street and runoff drainages (Parker et al. 1955, Teas 1977, Browder et al. 2005, Zhang et al. 2009). In addition to these general anthropogenic stressors, the Bay and nearby reefs are also targeted heavily year-round by numerous recreational fisheries (Bohnsack et al. 1994).

The purpose of this study was to measure the flux of reef fishes into and out of Biscayne Bay through a channel directly linking offshore coral reefs and inshore mangrove and seagrass beds. Specifically, the present study investigated the flux (occurrence, distribution and timing) of reef fishes in Broad Creek Channel in southern Biscayne Bay. Broad Creek Channel is one of just two main channels in southern Biscayne Bay linking the nearby reefs and with inshore seagrass beds and mangroves, which makes this an ideal study site. I tested the hypothesis that reef fishes will enter the Bay from adjacent Coral reefs through Broad Creek Channel at dusk, and likewise, exit the Bay through the same channel at dawn. This study focused on the fishes within the families Lutjanidae (snappers) and Haemulidae (grunts) because these species are abundant, economically and ecologically important and are believed to exhibit nocturnal migrations between coral reefs and adjacent seagrass or mangrove habitats (e.g. Odgen and Ehrlich 1977, Rooker and Dennis 1991, Nagelkerken et al. 2000, Hammerschlag and Serafy 2010).

2.2 Methods

2.2.1 Study Site

Biscayne Bay (hereafter, simply "the Bay") is a shallow, subtropical bay 65 km in length, up to 15 km wide, and with a 2 m average depth, except in dredged channels (Figure 2.1; Roessler and Beardsley, 1974). Up to 64% of the Bay bottom is covered with seagrasses because sediment depth and nutrients are sufficient, water depths are shallow, and water clarity is high (Markley & Milano 1985, Browder et al. 2005). The southern portion of the Bay is wide and is protected from the open ocean by a relatively pristine mangrove forest, and water conditions (dissolved oxygen, temperature, and salinity) are relatively stable in this area (Serafy et al. 2003). This part of the Bay is connected to the sea by two relatively short and deep channels (Broad Creek and Caesar Creek). To maximize sampling opportunities, I focused exclusively on Broad Creek channel.

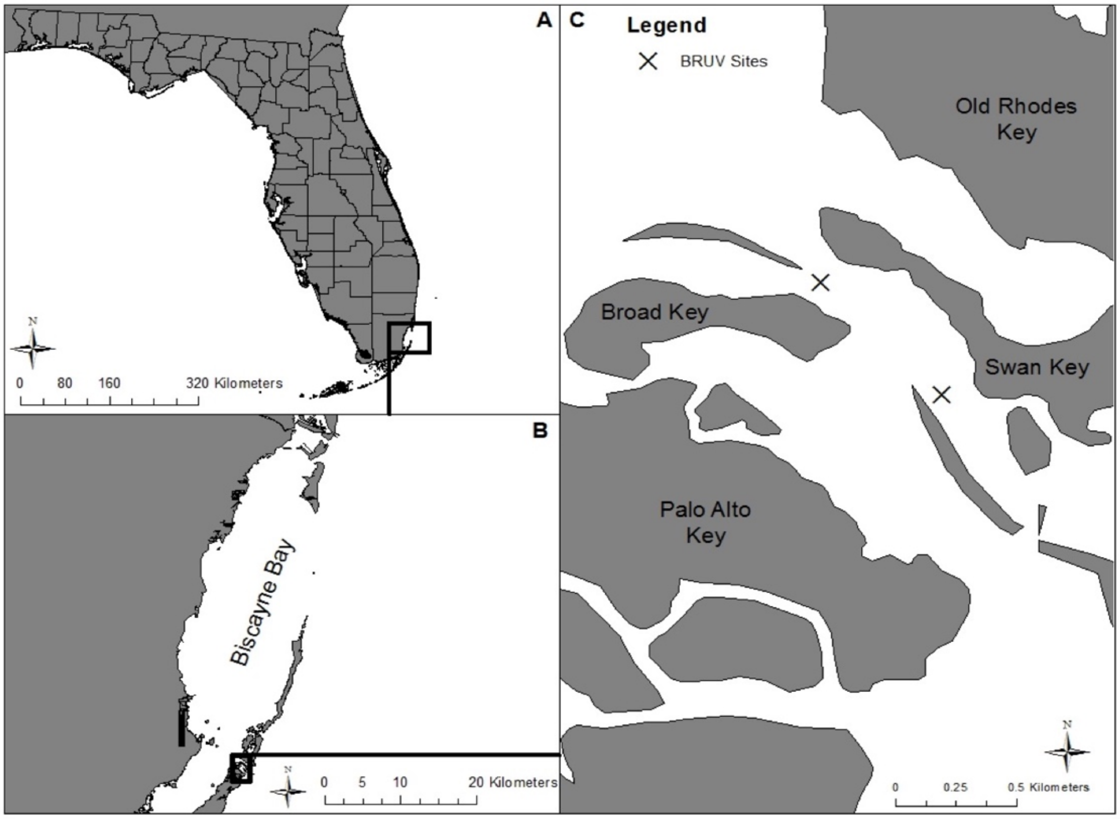


Figure 2.1. Study area. A) location of Biscayne Bay in Florida; B) location of of study area within Biscayne Bay; C) position of Broad Creek Channel. X's represents deployment sites of the BRUVS.

2.2.2 Baited Remote Underwater Video Stations

Baited remote underwater video station (BRUVS) were used to investigate the flux of fishes in and out of Broad Creek Channel. The use of underwater video systems has been widely used to investigate the species diversity and relative densities of fish assemblages since 1967 (Isaacs 1969). The flexibility and non-destructive method of cameras has permitted investigations of fish abundances in a variety of habitats, including shallow coral reefs (Chapman et al. 2011), estuaries (Taylor et al. 2013, Gladstone et al. 2012), pelagic environments (Heagney et al. 2007), deep rocky reefs (Goetze et al. 2011), and shallow rocky reefs (Broad et al. 2010). Underwater video stations have a vast depth at which they can be deployed and the addition of lights allows these stations to be deployed at night and at depths were sunlight does not penetrate (Cappo et al. 2004, Harvey et al. 2012). Underwater visual surveys (UVS) do not allow for the complete detection of “shy” species, cryptic species, and large predators, such as sharks, as many fishes will actively avoid divers (Brock 1982, Watson et al. 2005), which can result in biased surveys. Also, UVS, hook-and-line, trap, and trawl gear sampling methods can all be additionally limited by depth, fish behavior, seafloor rugosity, and size selection (Cappo et al. 2006). Baited remote underwater video surveys (BRUVS) are a type of remote sampling technique that is non-destructive and non-intrusive (Cappo et al. 2004). Furthermore, by adding a baited component near the camera, it allows the inclusion of top predators that are attracted to the camera by either the bait itself or aggregation of smaller fishes in the area (Taylor et al. 2013, Watson et al. 2005).

2.2.3 Video Collection

BRUVS were constructed of a polyvinyl chloride (PVC) frame containing a video camera (Hero3 model; GoPro, Inc.: San Mateo, CA, USA) and a bait crate. The bait crate was placed at the end of 0.75 m long bait arm held in a horizontal orientation to the substratum (Klages et al. 2014). Prior to each deployment, the bait crate was filled with 0.8 kg of chopped frozen pilchards *Sardinella aurita* (Valenciennes, 1874).

Sampling was conducted from August through October 2014. In an effort to discriminate diel patterns in fish activity, the 24-hr sampling period (also referred to as *time* period) was divided up into four time categories; day (1200-1300 hrs), night (2300-2400 hrs), dusk (30 min before and after sunset), and dawn (30 min before and after sunrise). My dawn and dusk sampling periods were chosen based on previous studies reporting that diel migrations commence around sunset and ends around sunrise (Ogden and Ehrlich 1979, McFarland et al. 1979, Luo et al. 2009, Hitt et al. 2011a,b). Two cameras were also placed at two choke points of the channel in an effort to determine flux into versus out of the Bay, one entering and one exiting, at 25.378 deg N x 80.261 deg W (Bay side) and 25.375 deg N x 80.256 deg W (ocean side) (Figure 2.1). During sampling, both zones were sampled simultaneously during each time period. BRUVS in each zone in the channel were replicated five times, totaling 40 BRUVS deployments each lasting 1 hour.

Environmental or abiotic conditions can have an effect on the distribution of fishes, therefore the following variables were measured at each deployment: temperature (via thermometer), salinity (via refractometer), current speed and direction, and tide.

Current speed was estimated upon collection of each BRUV using a drogue deployed to 1.5 m depth. As the surface buoy had drifted to the extent of its anchor line at the end of each BRUV deployment, it was considered a stationary point. The drogue was deployed next to the stationary buoy and attached to it via a 10 m line. I estimated current speed from the time taken for the drogue to travel to the extent of the 10 m line.

2.2.4 BRUV Analysis

Digital imagery recorded by the camera was downloaded and stored at its original resolution (720p) as a .mov file. The length of the bait arm (0.75 m) was used to standardized the field of view from the footage. Fishes observed outside this range were excluded from the data count to allow accurate species identification and control for the effect of visibility (Taylor et al. 2013). Each taxa observed in the video was identified to species and then grouped into family. Following the approach of Cappo et al. (2003), the following data were extracted from each video for snappers and grunts separately: 1) time of arrival for (*TOA*) and (2) abundance, i.e. the maximum number of individuals at any one time (*MaxN*; see Cappo et al. 2003). This last metric reduces "double counts" of individuals and provides a conservative estimate of abundance.

2.2.5 Data Analysis

2.2.5.1 Effects of Environmental Factors & diel period

Generalized linear models (GLMs) were applied to examine the effects of Ocean side vs Bay side, tide, time (sampling period), current speed, and the direction of water movement (incoming outgoing) on total abundance (*MaxN*) of Haemulidae spp. and Lutjanidae spp. Models were first fitted to a Poisson distribution and then tested for over-dispersion. If models tested positive for over-dispersion, a negative binomial model was applied (Ver Hoef & Boveng 2007). Model selection was conducted using an Akaike Information Criterion (AIC)-based approach, which involves sequentially removing parameters from the saturated model and selecting the model with the lowest AIC score (Logan 2010). In models that detected the variable *time* as significant, a pairwise test was used to determine difference between time periods. To avoid spurious significance from the number of pairwise tests, Bonferroni corrections were applied to these data analyses.

2.2.5.2 TOA and Abundance (MaxN) vs Time Side Interaction

Two-way factorial analyses of variance (ANOVA) was applied to examine the effects of time, channel entrance side (ocean vs bay) and a time x side interaction on *TOA* and abundances (*MaxN)* of Haemulidae spp. and Lutjanidae spp. If the results of ANOVAs indicated a significant effect at the 0.05 probability level, a post-hoc Tukey test was used to determine which means were significantly different. Statistical significance was declared at the $P < 0.05$. This study used *TOA* to determine entry and exit out of the channel (i.e., lower *TOA* at the Ocean side at dusk, higher *TOA* at the Bay side, and the reverse pattern at dawn). All statistical analyses were performed using the statistical package R (version 3.2.2; R Development Core Team 2008).

2.3 Results

2.3.1 General

A total of 40 BRUVS deployments (10 dusk, 10 day, 10 dawn, and 10 dark) yielded a total of 522 individual fishes observed: 361 were identified as Haemulidae spp., 67 were identified as Lutjanidae spp. Haemulidae spp., and Lutjanidae spp. combined composed for more than 80% of the fish assemblage. A complete list of species can be found in Table 2.1.

Trophic Level	Family	Genus Species	Common name	Dawn	Day	Dusk	Night	Total MaxN
3.8	Haemulidae	*Haemulon plumierii*	White grunt	98	89	116	32	335
3.5	Haemulidae	*Haemulon sciurus*	Bluestriped grunt	6	0	8	0	14
3.6	Haemulidae	*Anisotremus virginicus*	Porkfish	5	3	2	0	10
4.4	Haemulidae	*Haemulon aurolineatum*	Tomate	2	0	0	0	2
								361
4.2	Lutjanidae	*Lutjanus griseus*	Gray snapper	9	2	9	3	23
4.0	Lutjanidae	*Ocyurus chrysurus*	Yellowtail snapper	16	9	19	0	44
								67
4.1	Balistidae	*Balistes capriscus*	Gray triggerfish	10	10	2	0	22
4.1	Carangidae	*Caranx crysos*	Blue runner	1	12	1	0	14
	Scaridae	*Scarus* sp.	Parrotfish	1	8	3	0	12
3.5	Epinephelidae	*Epinephelus morio*	Red grouper	4	1	3	0	8
4.5	Sphyraenidae	*Sphyraena barracuda*	Great barracuda	1	3	2	0	6
4.3	Serranidae	*Mycteroperca bonaci*	Black grouper	1	2	2	0	5
4.2	Carangidae	*Caranx latus*	Horse-eye jack	3	0	2	0	5
2.0	Kyphosidae	*Kyphosus sectartix*	Bermuda chub	1	3	0	0	4
3.5	Dasyatidae	*Dasyatis sabina*	Atlantic stingray	1	2	0	0	3
3.2	Pomacanthidae	*Pomacanthus arcuatus*	Gray angelfish	0	1	2	0	3
4.5	Muraenidae	*Gymnothorax moringa*	Spotted moray	0	0	1	1	2
2.8	Monacanthidae	*Aluterus scriptus*	Scrawled filefish	0	2	0	0	2
4.2	Labridae	*Lachnolaimus maximus*	Hogfish	0	1	0	0	1
4.2	Ginglymostomatidae	*Ginglymostoma cirratum*	Nurse shark	0	0	0	1	1
3.6	Urotryhonidae	*Urobatis jamaicensis*	Yellow stingray	0	0	1	0	1
2.5	Pompacentridae	*Stegastes variabilis*	Cocoa damselfish	0	1	0	0	1
2.0	Acanthuridae	*Acanthurus coeruleus*	Blue tang	0	1	0	0	1
unidentified fishes				0	0	2	1	3
		Total		159	150	175	38	522

Table 2.1. Species list for all fishes observed across diel periods during baited underwater video surveys from August through October 2014 in southern Biscayne Bay, Florida (USA).

2.3.2 Effects of Environmental Factors

Generalized linear models (GLMs) applied to examine the effects of Ocean side vs Bay side of the channel, tide state, time (sampling period), current speed, and the direction of water movement (incoming outgoing) on total abundance (*MaxN*) of fishes, revealed that Lutjanidae and Haemulidae spp. abundances (*MaxN*) were affected by dawn, day, and dusk GLM, N=40, P>0.01). Further, snappers and grunts were overall more abundant on the ocean side (GLM, N=40, P>0.001; Table 2.2).

2.3.3 Abundance vs Diel Period

Lutjanidae spp. abundance (*MaxN*) peaked at dawn and dusk and was lowest at dark (GLM, N=40, P>0.05; Table 2.3, Figure 2.2). Haemulidae spp. abundance (*MaxN*) showed a significant decline at dark compared to dawn, day and dusk (GLM, N=40, P>0.0001; Table 2.3, Figure 2.2).

Model	Parameter	Coefficient	SE	Wald Z	*p* value
Haemulidae spp.	Intercept	0.963	0.245	3.93	8.50E-05
	Dawn	1.219	0.267	4.56	5.10E-06*
	Day	1.156	0.271	4.26	2.00E-05*
	Dusk	1.488	0.267	5.57	2.6E-08*
	Ocean	0.339	0.166	2.04	0.041*
Lutjanidae spp.	Intercept	-1.618	0.607	-2.67	0.00767
	Dawn	2.337	0.612	3.82	0.00013*
	Day	1.305	0.658	1.98	0.04739*
	Dusk	2.225	0.615	3.62	0.0003*
	Ocean	0.706	0.261	2.7	0.00685*

Table 2.2. General linear models for Haemulidae spp. and Lutjanidae spp abundances (MaxN). comparing environmental factors. Asterisks (*) indicate significance at p < 0.05.

2.3.4 TOA vs Time Side Interaction

There was no difference detected in *TOA* for Haemulidae spp; whereas, *TOA* for Lutjanidae spp. showed two time/side periods that differed. The *TOA* at the bay side at dusk was lower than the *TOA* at the bay side at day for snapper. The *TOA* at the ocean side at dawn was higher than *TOA* at the bay side at dusk for snappers. A summary of these results can be found in Tables 2.4-2.5.

Model	Parameter	Coefficient	SE	Wald Z	*p* value
Lutjanidae spp.	Intercept	-1.204	0.592	-2.03	0.04208
	Dark - Dawn	2.335	0.633	3.69	0.00022*
	Dark - Day	1.299	0.678	1.92	0.05519
	Dark - Dusk	2.234	0.636	3.51	4.40E-04*
	Dawn – Day	-1.036	0.398	-2.61	0.00917*
	Dawn – Dusk	-0.102	0.321	-0.32	0.75103
	Day – Dusk	0.934	0.402	2.32	0.0201
Haemulidae spp.	Intercept	2.299	0.166	13.77	2E-16
	Dark - Dawn	1.161	0.275	4.23	2.40E-05*
	Dark – Day	1.118	0.280	3.98	6.80E-05*
	Dark - Dusk	1.425	0.275	5.18	2.30E-07*
	Dawn – Day	-0.042	0.228	-0.19	0.85
	Dawn – Dusk	0.264	0.221	1.19	0.23
	Day – Dusk	0.306	0.229	1.34	0.18

Table 2.3. Haemulidae spp., and Lutjanidae spp. abundances compared across diel periods. Asterisks (*) indicate significance at $p < 0.05$, subsequently adjusted with Bonferroni corrections for multiple comparisons.

2.3.5 Abundance (MaxN) vs Time Side Interaction

A difference was detected in the abundance for Haemulidae spp and showed three time/side periods that differed. The abundance (*MaxN)* at the bay side at dusk was significantly higher than the abundance at the bay side at dark for grunts. Similarly, the abundance (*MaxN)* at the ocean side at dusk was significantly higher than the abundance at the bay side at dark for grunts. The abundance (*MaxN)* at the ocean side at day was significantly higher than the abundance at the bay side at dark for grunts. Lutjanidae spp. also detected a difference between seven time/side periods. The abundance (*MaxN)* at dawn on the ocean side was higher than the abundance at dark on the bay and ocean side.

Model	Df	Sum Sq	Mean Sq	F-value	*p* value
only Haemulidae spp.	10	2039	203.9	1.208	0.31
	49	8271	168.8		
only Lutjanidae spp.	7	3293	470.4	2.666	0.0125*
	152	26817	1764		

Table 2.4. Results of ANOVA analyses comparing 30 Time: Side combinations for TOA for Haemulidae spp., and Lutjanidae spp. Asterisks (*) indicate significance at $p < 0.05$.

Alike, the abundance (*MaxN)* on the ocean side at dusk was higher than the abundance on the bay side at dark, day, and dusk. The abundance (*MaxN)* of Lutjanidae spp. on the ocean side at dusk was also higher than the abundance on the abundance on the ocean side at dark and day. A summary of these result can be found in Tables 2.6-2.8.

2.4 Discussion

Diel sampling in Broad Creek Channel, a relatively short and deep channel in southern Biscayne Bay, revealed abundances of snappers and grunts that support the diel migration of reef fishes into seagrass beds at crepuscular periods. However, varying TOA patterns of reef fishes didn't reflect fishes moving into and out of the channel at dusk and dawn. TOA of all species was twice as low at dusk on the bay side than at dawn on the bay side. The difference in TOA could be impacted by the sampling period. Sampling occurred at distinct times in the diel cycle: dusk (30 min before and after sunset), and dawn (30 min before and after sunrise). It is possible that TOA during these times may not be the same as later or earlier in the sampling period. For example, Lou et al. (2009) noted fish returning to their daytime resting place frequently extended into the late morning hours. Therefor, the deployments may have missed the initial flux of the migration. In addition, TOA may have been altered by the presence of predators in the channel. Anti-predatory behavioral responses are major factors influencing habitat use in many animals. Large predatory fish alter the behavior of smaller prey and have been shown to have a significant impact on the results of BRUV studies (Klages et al. 2014). Consequently, the presence of large predators may have increased the TOA at some time periods.

Time x Side	diff	lwr	upr	p adj	
dawn:bay-dark:bay	2.737	-19.247	24.722	0.999	
day:bay-dark:bay	6.329	-15.492	28.151	0.986	
dusk:bay-dark:bay	-6.131	-28.322	16.059	0.989	
dark:ocean-dark:bay	0.620	-25.731	26.973	1.000	
dawn:ocean-dark:bay	6.068	-15.753	27.891	0.989	
day:ocean-dark:bay	-0.695	-23.056	21.666	1.000	
dusk:ocean-dark:bay	-3.403	-25.275	18.469	0.999	
day:bay-dawn:bay	3.592	-7.641	14.825	0.976	
dusk:bay-dawn:bay	-8.869	-20.803	3.0653	0.309	
dark:ocean-dawn:bay	-2.116	-20.676	16.442	0.999	
dawn:ocean-dawn:bay	3.331	-7.902	14.565	0.984	
day:ocean-dawn:bay	-3.432	-15.680	8.815	0.988	
dusk:ocean-dawn:bay	-6.140	-17.472	5.190	0.709	
dusk:bay-day:bay	-12.461	-24.092	-0.829	0.026	*
dark:ocean-day:bay	-5.708	-24.074	12.657	0.979	
dawn:ocean-day:bay	-0.260	-11.171	10.650	1.000	
day:ocean-day:bay	-7.024	-18.977	4.927	0.616	
dusk:ocean-day:bay	-9.732	-20.744	1.278	0.125	
dark:ocean-dusk:bay	6.752	-12.050	25.555	0.955	
dawn:ocean-dusk:bay	12.200	0.569	23.831	0.032	*
day:ocean-dusk:bay	5.436	-7.176	18.049	0.888	
dusk:ocean-dusk:bay	2.728	-8.997	14.454	0.996	
dawn:ocean-dark:ocean	5.448	-12.918	23.814	0.984	
day:ocean-dark:ocean	-1.315	-20.319	17.687	0.999	
dusk:ocean-dark:ocean	-4.024	-22.450	14.401	0.997	
day:ocean-dawn:ocean	-6.763	-18.716	5.188	0.661	
dusk:ocean-dawn:ocean	-9.472	-20.483	1.539	0.148	
dusk:ocean-day:ocean	-2.708	-14.752	9.336	0.997	

Table 2.5. Results of post-hoc Tukey test comparing Lutjanidae spp. time of arrival (TOA) across diel periods and side (ocean vs bay). Asterisks (*) indicate significance at p

In South Florida, Haemulidae spp. and Lutjanidae spp. are abundant and have substantial economic importance (Murphy et al. 1999, Rutherford et al. 1989). My results of abundance (MaxN) indicate that both of these species groups are moving into the channel at dusk (Figure 2.2). My finding concurs with Beets et al. (2003), who noted the migration from reef to seagrass generally occurs during twilight periods; the high abundance at dawn is a result of these fishes moving back to their diurnal resting place. The abundance at day that I observed was not expected, as fish should have returned to their resting place by morning. In this case, the Haemulidae spp. documented in the BRUVS videos during the day may use Broad Creek Channel as a daytime resting place due to anthropogenic structures (e.g., sunken household appliances and boats) that may reduce predation risk in much the same way as natural patch reefs do (Ogden & Ehrlich

1997). In fact, Haemulidae spp. have been documented sheltering in channels and other structurally complex habitats (Rooker and Dennis 1991, Nagelkerken et al. 2000, Verweij and Nagelkerken 2007). Lutjanidae spp. abundance also reflected the migration of coral reef-associated species into adjacent seagrass beds at night. Their abundance was higher at dusk on the ocean side than at dusk on the bay side. Their abundance was also highest during crepuscular periods, similar to the distinct diel migration pattern documented by Luo et al. (2009), in which the movement of Lutjanus spp. to the seagrass beds began at dusk, and the return began at dawn and extended into the morning. Haemulidae spp. and Lutjanidae spp. showed similar patterns in the channel and highlight channels as a connectivity route between inshore seagrass beds and offshore reefs.

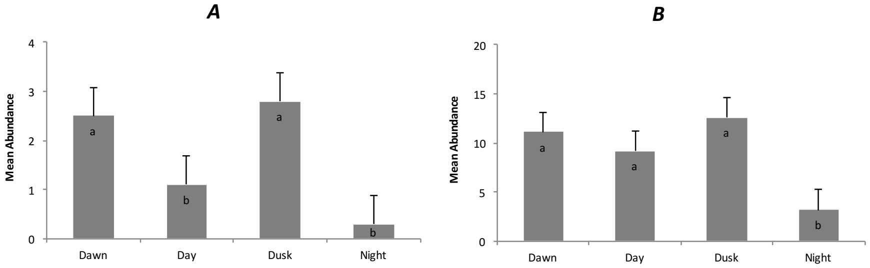


Figure 2.2. Comparison of mean abundance (MaxN) rates for: A) Lutjanidae spp; B) Haemulidae spp. across diel periods. Letters indicate significantly different groupings at P<0.05 adjusted with the Bonferroni correction for multiple comparisons; same letter=no difference.

Tide, water movement, and current speed did not have an effect on family group abundances. Tide has been shown to affect species abundance and richness in shallow water creeks and channels as fish are forced to move from habitats due to a decrease in water level at low tide. (e.g., Dorenbosh et al. 2004). Broad Creek Channel is roughly 3-5 m deep; thus, areas of the channel and bay are accessible throughout the tidal cycle. The movement and speed of the current moving through the channel also did not impact species abundances. Current direction and speed have been shown to impact BRUVS studies, as the bait plume can be carried further distances and attract more fishes (Taylor et al. 2013). However, in this study, these two factors did not significantly affect the abundance of the trophic guilds and species I investigated. The ocean side demonstrated a higher abundance of Haemulidae spp., and Lutjanidae spp. than the bay side. However, the direction of the current did not affect this abundance. On both incoming and outgoing tides, the ocean side had a higher abundance of fishes. Therefore, Haemulidae spp. and Lutjanidae spp. are moving into and through the channel at distinct times despite possible changes in tide, current velocity, and movement.

Model	Df	Sum Sq	Mean Sq	F-value	*p* value
Haemulidae spp.	7	570.8	81.54	2.679	0.026*
	32	974	30.44		
Lutjanidae spp.	7	89.5	12.79	5.683	0.00025*
	32	72	2.25		

Table 2.6. Results of ANOVA analyses comparing 30 Time:Side combinations for Haemulidae spp., and Lutjanidae spp. abundances (MaxN). Asterisks (*) indicate significance at $p < 0.05$.

It has been hypothesized that locally migrating fishes make a tradeoff between an increase in food abundance and higher predation risk (Hammerschlag et al. 2010a,b). At night, seagrass beds harbor higher densities of preferred food (Nagelkerken et al. 2000), such that these fishes are increasing their risk of predation in exchange for maximizing their energetic gains from prey. In channels, which should be an area of high predation, fishes are forced to swim through a narrow opening as they move from the ocean to the bay and later return. In my study, great barracuda Sphyraena barracuda (Edwards, 1771) and groupers (Subfamily Epinephelidae) both showed similar patterns to Haemulidae spp., and Lutjanidae spp in the channel. I hypothesize these predators are positioning themselves in the channel at dawn and dusk to take advantage of small reef fishes moving through the channel. However, more research is needed to determine whether S. barracuda and Epinephelidae spp. abundances are higher in the channel at dawn and dusk. Predation rates in the channel should also be investigated to determine if they correlate with an increase in S. barracuda and Epinephelidae spp. abundance. If predation risk is higher in the channel, then reef fishes are making a tradeoff between an increase in food abundance and higher predation risk. Further studies should investigate how prey fish species react to the presence of predators in the channel. For example, Dorenbosh et al. (2004) predicted that juvenile Lutjanus spp. will avoid small channels at high tide to avoid large predators. I believe the presence of large predators in the channel will have an impact on the movement of prey species moving through the channel. But, more research is needed to determine how predators affect the migration of reef species through channels interlinking tropical and subtropical inshore feeding grounds and offshore coral reefs.

Time x Side	diff	lwr	upr	p adj	
dawn:bay-dark:bay	2.200	-0.873	5.273	0.314	
day:bay-dark:bay	8.000	-2.273	3.873	0.988	
dusk:bay-dark:bay	8.000	-2.273	3.873	0.988	
dark:ocean-dark:bay	2.000	-2.873	3.273	0.999	
dawn:ocean-dark:bay	3.600	0.526	6.673	0.012	*
day:ocean-dark:bay	6.000	-2.473	3.673	0.998	
dusk:ocean-dark:bay	4.200	1.126	7.273	0.002	*
day:bay-dawn:bay	-1.400	-4.473	1.673	0.814	
dusk:bay-dawn:bay	-1.400	-4.473	1.673	0.814	
dark:ocean-dawn:bay	-2.000	-5.073	1.073	0.431	
dawn:ocean-dawn:bay	1.400	-1.673	4.473	0.814	
day:ocean-dawn:bay	-1.600	-4.673	1.473	0.695	
dusk:ocean-dawn:bay	2.000	-1.073	5.073	0.431	
dusk:bay-day:bay	-1.665	-3.073	3.073	1.000	
dark:ocean-day:bay	-6.000	-3.673	2.473	0.998	
dawn:ocean-day:bay	2.800	-0.273	5.873	0.095	
day:ocean-day:bay	-2.000	-3.273	2.873	0.999	
dusk:ocean-day:bay	3.400	0.326	6.473	0.021	*
dark:ocean-dusk:bay	-6.000	-3.673	2.473	0.998	
dawn:ocean-dusk:bay	2.800	-0.273	5.873	0.095	
day:ocean-dusk:bay	-2.000	-3.273	2.873	0.999	
dusk:ocean-dusk:bay	3.400	0.326	6.473	0.021	*
dawn:ocean-dark:ocean	3.400	0.326	6.473	0.021	*
day:ocean-dark:ocean	4.000	-2.673	3.473	0.999	
dusk:ocean-dark:ocean	4.000	0.926	7.073	0.004	*
day:ocean-dawn:ocean	-3.000	-6.073	0.073	0.059	
dusk:ocean-dawn:ocean	6.000	-2.473	3.673	0.998	
dusk:ocean-day:ocean	3.600	0.526	6.673	0.012	*

Table 2.7. Results of post-hoc Tukey test comparing Lutjanidae spp. abundance (MaxN) across diel periods and side (ocean vs bay). Asterisks (*) indicate significance at $p < 0.05$.

It is worth noting that my sampling occurred at distinct times in the diel cycle. Thus, it is possible that the abundances of fishes and TOA during these times may not be the same as later or earlier in the sampling period. For example, Lou et al. (2009) noted fish returning to their daytime resting place frequently extended into the late morning hours. Tulevech and Recksiek (1994), found movements to generally occur during dusk, but some exceptions were noted. Therefore, fish distribution during the dawn and dusk sampling window, may differ earlier or later in the morning. This caveat applies to the other sampling periods as well. It is also worth considering that my results are based on sampling that used the presence of light to illuminate the field of view. Hence, it is possible that fishes were deterred or attracted by the presence of artificial light during dawn, dusk, and night (Fitzpatrick et al. 2013). Specifically, the lower abundance (MaxN) of fishes at dark may be a species specific reactions to artificial lighting (Carazo et al. 2013, Harvey et al. 2012). For example, Senegalese sole Solea senegalensis (Kaup, 1858) exposed to high intensity red lights at night prompted behaviors of fear and escape (Carazo et al. 2013). Furthermore, plume dispersal or area of attraction is an unknown confounding factor in this study.

Time x Side	Diff	lwr	upr	p adj	
dawn:bay-dark:bay	9.05	-1.188	19.288	0.113	
day:bay-dark:bay	5.45	-4.788	15.688	0.664	
dusk:bay-dark:bay	10.85	0.611	21.088	0.031	*
dark:ocean-dark:bay	3.05	-7.188	13.288	0.975	
dawn:ocean-dark:bay	9.65	-0.588	19.888	0.075	
day:ocean-dark:bay	12.50	1.707	23.292	0.014	*
dusk:ocean-dark:bay	13.00	2.207	23.792	0.010	*
day:bay-dawn:bay	-3.60	-13.253	6.053	0.920	
dusk:bay-dawn:bay	1.80	-7.853	11.453	0.998	
dark:ocean-dawn:bay	-6.00	-15.653	3.653	0.482	
dawn:ocean-dawn:bay	0.60	-9.053	10.253	0.999	
day:ocean-dawn:bay	3.45	-6.788	13.688	0.951	
dusk:ocean-dawn:bay	3.95	-6.288	14.188	0.906	
dusk:bay-day:bay	5.40	-4.253	15.053	0.609	
dark:ocean-day:bay	-2.40	-12.053	7.253	0.991	
dawn:ocean-day:bay	4.20	-5.453	13.853	0.841	
day:ocean-day:bay	7.05	-3.188	17.288	0.355	
dusk:ocean-day:bay	7.55	-2.688	17.788	0.276	
dark:ocean-dusk:bay	-7.80	-17.453	1.853	0.184	
dawn:ocean-dusk:bay	-1.20	-10.853	8.453	0.999	
day:ocean-dusk:bay	1.65	-8.588	11.888	0.999	
dusk:ocean-dusk:bay	2.15	-8.0888	12.388	0.996	
dawn:ocean-dark:ocean	6.60	-3.053	16.253	0.364	
day:ocean-dark:ocean	9.45	-0.788	19.688	0.086	
dusk:ocean-dark:ocean	9.95	-0.288	20.188	0.061	
day:ocean-dawn:ocean	2.85	-7.388	13.088	0.982	
dusk:ocean-dawn:ocean	3.35	-6.888	13.588	0.958	
dusk:ocean-day:ocean	0.50	-10.292	11.292	0.999	

Table 2.8. Results of post-hoc Tukey test comparing Haemulidae spp. abundance (*MaxN*) across diel periods and side (ocean vs bay). Asterisks (*) indicate significance at $p < 0.05$.

In summary, my results revealed evidence of reef fishes utilizing a channel as a passageway for moving into the bay at dusk and out of the bay at dawn. The high abundance of Haemulidae spp. and Lutjanidae spp. at dusk reinforces the timing of diel migrations to seagrass beds to feed commencing at crepuscular periods (Hobson 1965, Ogden and Ehrlich 1979, McFarland et al. 1979, Luo et al. 2009, Appeldoorn et al. 2009, Hitt et al. 2011ab). The distance from inlets or channels to seagrass beds and inlets or channels to coral reef habitats has been shown to effect species abundances and diversity at the larval stage (Ford et al. 2010) and nursery stage (Faunce and Serafy 2007). This study adds to the important biological role channels play by interlinking nocturnal feeding grounds and diurnal resting grounds. I believe the location of seagrass beds relative to the distance of inlets or channels within a tropical and subtropical bay, may have a significant effect on the abundance and distribution of reef fishes feeding at night within seagrass beds. Furthermore, seagrass beds on the windward side of Biscayne Bay are documented foraging grounds for reef species (Luo et al. 2009), but my study shows that reef fishes are similarly feeding in the seagrass beds within the Bay as well. Efforts to conserve stocks of important reef fishes should therefore include protection of important associated inshore feeding grounds. Given the importance of connectivity between habitats to economically and ecologically important fishes, channels should be given high conservation importance for management protection.

Conclusion

The overall objective of this thesis was to improve the understanding of the distribution, abundance and movement patterns of fish in inshore subtropical estuaries. In Chapter 1, I investigated the temporal and spatial predator assemblages within the mangrove-seagrass nursery habitat continuum. Large mobile predatory fishes have been mainly absent from surveys in mangrove and shallow seagrass habitats, presumably because the methods used in the studies (e.g., underwater visual surveys and seine nets) (Harvey et al. 2007, Brock 1982). This thesis investigated the predator assemblage by using baited underwater video surveys. The results indicate that the fish predator assemblage is not uniformly distributed neither temporally over a 24 hr period nor spatially over a distance gradient from shore. This thesis establishes distribution patterns of predatory fish species and life-history stages within the inshore mangrove-seagrass nursery habitat.

The results suggest that large predators play an ecological role in the distribution of juvenile fishes within the mangrove-seagrass nursery area. Large predators directly influence the habitat use of juvenile of fishes by forcing juvenile Lutjanidae spp. and Haemulidae spp. to utilize seagrass beds further form shore. Recent studies by Hammerschlag et al. (2010) and Dorenbosh et al (2009) have reported high predation rates and subsequently hypothesized this distribution may be a result of an increase in predator abundance. My results suggest that this distribution may be a result of an increase in feeding activity by predators, thereby optimizing their probability of prey capture, rather than simply a numerical increase in predator abundance.

Mangrove and seagrass beds in southern Biscayne Bay should be considered for Essential Fish Habitat designation in any potential state management plans for *S. barracuda, Lutjanus* spp*, and Haemulon spp.* The southern portion of Biscayne bay is relatively pristine while the northern portion of the bay is an urbanized bay, that has almost lost its entire mangrove shore line (DERM, 1981). Given the ecologically and economically importance of *S. barracuda*, *Lutjanus* spp*., and Haemulon spp.* in south Florida it is important to protect the remaining seagrass and mangroves habitats within the Bay (Bohnsack et al. 1994, Bohnsack and Ault, 1996). Based on the result of my thesis, *Lutjanus* spp. and *Haemulon* spp., settle in southern Biscayne Bay's seagrass beds as exclusive diurnal predators. Once the individuals grow large enough, they begin to shelter in the mangrove habitat and feed in the seagrass beds only at night. *S. barracuda* of all three life-history stages were documented and not uniformly distributed in the seagrass-mangrove habitat. This highlights the importance of this area not only as a nursery habitat for this species, but also as a juvenile and adult habitat. A recent decline in *S. barracuda* populations in the Western Atlantic Ocean is now being investigated by the Florida Fish and Wildlife Commission. I believe the mangrove-seagrass habitat should be given high conservation importance for any management protection plan for *S. barracuda*.

Chapter 2 described the diel movement patterns of fish through a channel linking offshore coral reefs and inshore seagrass habitats. The work presented in this thesis has built upon previous studies and attained a more comprehensive understanding of the timing, occurrence and distribution of reef fish movements from coral reefs to inshore feeding grounds. This chapter highlights the biological connectivity between the seagrass beds and coral reefs, with the seagrass beds in Biscayne Bay serving as foraging grounds for ecologically and economically important reef fishes in south Florida (Bohnsack et al. 1994, Bohnsack and Ault, 1996).

The present study highlights the necessity of better fundamental knowledge of the connectivity between estuaries and adjacent marine areas. The lack of adult coral reef species in chapter one, such as grunts and snappers suggests that reef fishes are moving through Broad Key Channel and not foraging in the study site of Chapter 1. I believe the location of seagrass beds relative to the distance of inlets or channels within a tropical and subtropical bay, may have a significant effect on the abundance and distribution of reef fishes feeding at night within seagrass beds. More research is needed to determine where these fishes are foraging after they enter the channel as they can travel long distances between resting and foraging locations (Beets et al. 2003). Understanding fish movement and migrations between seagrass habitats and coral reefs will facilitate more informed ecosystem-level management (Unsworth et al. 2007).

Through the use of baited remote underwater video surveys, this thesis has increased the general knowledge of fish distribution, abundance and movement patterns in inshore subtropical estuaries. The work presented in this thesis has built upon the importance of mangrove-seagrass habitats as well as the connectivity between offshore reefs and inshore seagrass beds. It is clear from this study that various species forage in seagrass beds at distinctly different times. Studies that do not take these spatiotemporal patterns into account may generate inaccurate estimates for both population abundance and utilization of specific habitats. This could negatively affect the effectiveness of subsequent management measures, which are critical in the mangrove and seagrass habitats as they continue to be impacted by anthropogenic effects. The results of this thesis suggest that the degradation or loss of seagrass habitat along a distance gradient from mangrove forests and distance from channels could differentially impact the life-history stages of reef fish species.

Future Studies

The original goal of this study was to sample in a dry and wet season. However, due to insufficient funding and boat logistics, I was only able sample during a two-month period that occurred during the wet season. Based on previous studies (Hammerschlag and Serafy 2010), I believe the predator assemblage will vary between the wet and dry season. In this study, tertiary and secondary consumer abundances (*MaxN*) differed greatly between zones, highlighting the ecological role large predators play within the mangrove-seagrass nursery habitat. If large predators are absent or there is a shift in the predator assemblage due to seasonal change it will have direct impact on the mangrove-seagrass nursery ecosystem. Therefore, future studies should investigate the predator assemblage in both the wet and dry season (just don't do it at Nova Southeastern University).

There is limited knowledge of adult great barracuda within the mangrove-seagrass habitat. After reaching a certain size (*ca.* 500 mm), individuals are thought to move from the mangrove-seagrass to reef-dominated habitat (Christine 2010). However, in this study, adult *S. barracuda* were documented and not uniformly distributed with distance from shore. Adult barracuda should be implanted with small acoustic telemetry transmitters or satellite tags to understand the habitat utilization within the seagrass-mangrove habitat. Receivers could be placed at two entry and exit points within southern Biscayne Bay (Broad Creek Channel and Caesar Creek) to determine if adults are moving between inshore habitats and offshore habitats. Receivers could also be placed along a distance gradient from the mangrove shore into adjacent seagrass beds to determine how these fish are distributed spatial and temporal in the mangrove-seagrass continuum.

In chapter 2, great barracuda and groupers both showed similar patterns to Haemulidae spp., and Lutjanidae spp in the channel. I hypothesize these predators are positioning themselves in the channel at dawn and dusk to take advantage of small reef fishes moving through the channel. However, more baited remote underwater video deployments are needed to determine whether great barracuda and grouper abundances are higher in the channel at dawn and dusk. Predation rates in the channel should also be investigated to determine if they correlate with an increase in great barracuda and grouper abundance. A series of diel tethering experiments in Broad Key Channel could be conducted to compare predation rates on reef fishes moving through the channel at different photoperiods. If predation risk is higher in the channel, then reef fishes are making a tradeoff between an increase in food abundance and higher predation risk.

References

Adams AJ, Dahlgren CP, Kellison GT, Kendall MS, Layman CA, Ley JA, Nagelkerken I, Serafy JE. 2006. Nursery function of tropical back-reef systems. Mar Eco Prog Ser. 318:187-301

Allemann RW. 1995. An update to the surface water improve- ment and management plan for Biscayne Bay. South Florida Water Management District. West Palm Beach, Florida.

Appeldoorn RS, Aguilar-Perera A, Bouwmeester BLK, Dennis, GD, Hill RL, Merten W, Recksiek CW, Williams SJ, 2009. Movement of fishes (Grunts: Haemulidae) across the coral reef seascape: A review of scales, patterns and processes. *Caribbean Journal of Science, 45(2-3)*:304-316.

Atwood TB, Connolly RM, Ritchie EG, Lovelock CE, Heithaus MR, Hays GC, Fourqurean JW, Macreadie PI. 2015. Predators help protect carbon stocks in blue carbon ecosystems. Nature Climate Change. 5(12):1038-45.

Baker R, Sheaves M. 2005. Redefining the piscivore assemblage of shallow estuarine nursery habitats. Mar Ecol Prog Ser. 291:197-213.

Baker R, Sheaves M. 2006. Visual surveys reveal high densities of large piscivores in shallow estuarine nurseries. Mar Ecol Prog Ser. 323:75-82.

Beck MW, Heck Jr KL, Able KW, Childers DL, Eggleston DB, Gillanders BM, Halpern B, Hays CG, Hoshino K, Minello TJ, Orth, RJ. 2001. The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates: A better understanding of the habitats that serve as nurseries for marine species and the factors that create site-specific variability in nursery quality will improve conservation and management of these areas. Bioscience. 51(8):633-641.

Beets J, Muehlstein L, Haught K, Schmitges H. 2003. Habitat connectivity in coastal environ- ments: Patterns and movements of Caribbean coral reef fishes with emphasis on bluestriped grunt, Haemulon scirus. Gulf. Caribb. Res. 14(2):29-42.

Berkström C, Lindborg R, Thyresson M, Gullström M. 2013a. Assessing connectivity in a tropical embayment: fish migrations and seascape ecology. Biological conservation. 166:43-53.

Berkström C, Jörgensen T, Hellström M. 2013b. Ecological connectivity and niche differentiation between two closely related fish species in the mangrove-seagrass-coral reef continuum. Mar Ecol Prog Ser, 477:201-215.

Billings VC, Munro JL. 1974. The biology, ecology, exploitation and management of Caribbean reef fishes: Pomadasydae (grunts). Part 5. Research Reports of the Zoology Department of the University of the West Indies 3:1-128

Bohnsack JA, Ault JS. 1996. Management strategies to conserve marine biodiversity. Oceanography 9:73–82.

Bohnsack JA, Harper DE, McClellan DB. 1994. Fisheries trends from Monroe County, Florida. Bulletin of Marine Science 54:982–1018.

Broad A, Knott N, Turon X, Davis AR. 2010. Effects of a shark repulsion device on rocky reef fishes: no shocking outcomes. Mar Eco Prog Ser. 408:295-298

Brock RE. 1982. A critique of the visual census method for assessing coral reef fish populations. Bulletin of Marine Science. 32(1):269-276.

Brooks EJ, Sloman KA, Sims DW, Danylchuk AJ. 2011. Validating the use of baited remote underwater video surveys for assessing the diversity, distribution and abundance of sharks in the Bahamas. Endangered Species Research. 13(3):231-243.

Browder JA, Alleman R, Markley S, Ortner P, Pitts PA. 2005. Biscayne Bay conceptual ecological model. Wetlands. 25(4):854-869.

Brown CA, Gruber SH. 1988. Age assessment of the lemon shark, Negaprion brevirostris, using tetracycline validated vertebral centra. Copeia. 747-753.

Burke NC. 1995. Nocturnal foraging habitats of French and bluestriped grunts, Haemulon flavolineatum and H. sciurus, at Tobacco Caye, Belize. Environmental Biology of Fishes. 42(4):365-374.

Cappo M, Harvey E, Malcolm H, Speare P. 2003. Potential of video techniques to monitor diversity, abundance and size of fish in studies of marine protected areas. In: Beuner JP, Grant A, Smith DC, Mahon D, editors. Aquatic Protected Areas-What works best and how do we know? Proceedings of the World Congress on Aquatic Protected Areas. Cairns, August 2002. 455-464.

Cappo M, Harvey E, Shortis M. 2006. Counting and measuring fish with baited video techniques-an overview. Australian Society for Fish Biology Workshop Proceedings. 101-114.

Cappo M, Speare P, De'ath G. 2004. Comparison of baited remote underwater video stations (BRUVS) and prawn (shrimp) trawls for assessments of fish biodiversity in inter-reefal areas of the Great Barrier Reef Marine Park. Journal of Experimental Marine Biology and Ecology. 302(2):123-152.

Cappo M, Speare P, Wassenberg TJ, Harvey E, Rees M, Heyward A, Pitcher R. 2001. Use of baited remote underwater video stations (BRUVS) to survey demersal fish—how deep and meaningful. In: Harvey ES, Cappo M, editors. Video sensing of the size frequency and abundance of target and non-target fauna in Australian fisheries – a national workshop. Fisheries Research and Development Corporation, Australia, 63-71.

Carazo I, Norambuena F, Oliveira C, Sánchez-Vázquez FJ, Duncan NJ. 2013. The effect of night illumination, red and infrared light, on locomotor activity, behaviour and melatonin of Senegalese sole (Solea senegalensis) broodstock. Physiology and Behavior. 118:201-207.

Castro JI. 2000. The biology of the nurse shark, Ginglymostoma cirratum, off the Florida east coast and the Bahama Islands. Environ Biol Fish. 58(1):1-22.

Chapman D, Babcock E, Abercrombie D, Bond M, Pikitch E. 2011. The elasmobranchs of Glover's Reef Marine Reserve. Fisheries Centre Research Reports. 19(6):38.

Chin A, Heupel MR, Simpfendorfer CA, Tobin AJ. 2013. Ontogenetic movements of juvenile blacktip reef sharks: evidence of dispersal and connectivity between coastal habitats and coral reefs. Aquatic Conservation: Marine and Freshwater Ecosystems, 23(3):468-474.

Christian J. 2003. Evaluating the energetic value and structural contributions of mangroves and seagrasses to local fish populations: the importance of habitat connectivity. Masters, University of Alabama.

Christine AO. 2010. The physiological ecology and behavior of an apex predatory fish, the great barracuda (Sphyraena barracuda). Doctoral Dissertation. Carleton University Ottawa.

Correia J, De Marignac J, Gruber SH. 1995. Young lemon shark behavior in Bimini Lagoon. Bahamas J Sci. 10, 2-8.

Danilowicz BS, Sale PF. 1999. Relative intensity of predation on the French grunt, Haemulon flavolineatum, during diurnal, dusk, and nocturnal periods on a coral reef. Marine Biology. 133(2):337-343.

De Sylva DP. 1963. Systematics and life history of the great barracuda Sphyraena barracuda. Studies in Tropical Oceanography. Univeristy of Miami. 1:1-179

Domeier ML, Koenig C, Coleman F. 1996. Reproductive biology of the gray snapper (*Lutjanus griseus*), with notes on spawning for other western Atlantic snappers (Lutjani- dae). In: F. Arreguin-Sanchez J L, Munro MC, Blagos, Pauly D, editors. Biology, series and culture of tropical snappers and groupers. ICLARM Conf. proc. Intl. Ctr. Liv. Aquat. Res. Mgt., Manila. 48:189–201

Dorenbosch M, Grol MG, Groene A, Velde VDG, Nagelkerken I. 2009. Piscivore assemblages and predation pressure affect relative safety of some back-reef habitats for juvenile fish in a Caribbean bay. Mar Ecol Prog Ser. 379:181 -196.

Dorenbosch M, Verweiji MC, Nagelkerken I, Jiddawi N, van der Velde G. 2004. Homing and daytime tidal movements of juvenile snappers (Lutjanidae) between shallow-water nursery habitats in Zanzibar, western Indian Ocean. Environmental Biology of Fishes, 70(3):203-209.

Faunce CH, Serafy JE. 2006. Mangroves as fish habitat: 50 years of field studies. Mar Eco Prog Ser, 318:1-18.

Faunce CH, Serafy JE. 2007. Nearshore habitat use by gray snapper (Lutjanus griseus) and bluestriped grunt (Haemulon sciurus): environmental gradients and ontogenetic shifts. Bull Mar Sci. 80:473–495.

Faunce CH, Serafy JE. 2008. Growth and secondary production of an eventual reef fish during mangrove residency. Estuar Coast Shelf Sci. 79(1):93-100.

Fitzpatrick C, McLean D, Harvey ES. 2013. Using artificial illumination to survey nocturnal reef fish. Fish Res. 146:41-50.

Fletcher D, MacKenzie D, Villouta E. 2005. Modeling skewed data with many zeros: A simple approach combining ordinary and logistic regression. Environ Ecol Stat.12:45–54.

Ford JR, Williams RJ, Fowler, AM, Cox DR, Suthers IM. 2010. Identifying critical estuarine seagrass habitat for settlement of coastally spawned fish. Marine Ecology Progress Series. 408:181-193.

Franks BR. 2007. The spatial ecology and resource selection of juvenile lemon sharks (Negaprion brevirostris) in their primary nursery areas. Doctoral dissertation. Drexel University. Philadelphia, Pennsylvania.

Franks JS, VanderKooy KE. 2015. Feeding habits of juvenile lane snapper Lutjanus synagris from Mississippi coastal waters, with comments on the diet of gray snapper Lutjanus griseus. Gulf Caribb Res. 12(1):11-17.

Friedlander AM, Monaco ME. 2007. Acoustic tracking of reef fishes to elucidate habitat utilization patterns and residence times inside and outside marine protected areas around the island of St. John, USVI.

Garcia ER, Potts JC, Rulifson RA, Manooch CS. 2003. Age and growth of yellowtail snapper, Ocyurus chrysurus, from the southeastern United States. Bull Mar Sci. 72(3):909-921.

Garcia J, Mourier J, Lenfant P. 2015. Spatial behavior of two coral reef fishes within a Caribbean marine protected area. Marine environmental research. 109:41-51.

García-Arteaga JP. 1992. Edad y crecimiento del ronco amarillo, *Haemulon sciurus* (Shaw) (pisces: haemulidae) en la plataforma suroccidental de Cuba. Cienc Biol. 25:104–116.

Gladstone W, Lindfield S, Coleman M, Kelaher B. 2012. Optimisation of baited remote underwater video sampling designs for estuarine fish assemblages. Journal of Experimental Marine Biology and Ecology. 429:28-35.

Goetze JS, Langlois TJ, Egli DP, Harvey ES. 2011. Evidence of artisanal fishing impacts and depth refuge in assemblages of Fijian reef fish. Coral Reefs. 30(2):507-517

Gruber SH, Stout RG. 1983. Biological materials for the study of age and growth in a tropical marine elasmobranch, the lemon shark, Negaprion brevirostris (Poey). In Proceedings of the international workshop on age determination of oceanic pelagic fishes: tunas, billfishes, and sharks. NOAA Tech Rep NMFS. 8:193-205.

Gruber SH. 1982. Role of the lemon shark, Negaprion brevirostris (Poey), as a predator in the tropical marine environment: a multidisciplinary study. Fla Sci. 45:46-75

Hammerschlag N, Heithaus MR, Serafy JE. 2010b. Influence of predation risk and food supply on nocturnal fish foraging distributions along a mangrove-seagrass ecotone. Mar Eco Prog Series. 414:223-235.

Hammerschlag N, Morgan A, Serafy JE. 2010a. Relative predation risk for fishes along a subtropical mangrove-seagrass ecotone. Marine Ecology Progress Series. 401:259–267.

Hammerschlag N, Ovando D, Serafy, JE. 2010c. Seasonal diet and feeding habits of juvenile fishes foraging along a subtropical marine ecotone. Aquatic Biology. 9:279–290.

Hammerschlag N, Serafy JE. 2010. Nocturnal fish utilization of a subtropical mangrove-seagrass ecotone. Marine Ecology. 31(2):364-374.

Hannan KM, Driggers III, William B, Hanisko DS, Jones LM, Canning AB. 2012. Distribution of the nurse shark, Ginglymostoma cirratum, in the northern Gulf of Mexico. Bulletin of Marine Science. 88(1):73-80.

Harvey ES, Butler JJ, McLean DL, Shand J. 2012. Contrasting habitat use of diurnal and nocturnal fish assemblages in temperate Western Australia. J Exp Mar Biol Ecol. 426:78-86.

Harvey ES, Butler JJ, McLean DL, Shand J. 2012. Contrasting habitat use of diurnal and nocturnal fish assemblages in temperate Western Australia. J Exp Mar Biol Ecol. 426:78-86.

Heagney EC, Lynch TP, Babcock RC, Suthers IM. 2007. Pelagic fish assemblages assessed using mid-water baited video: standardising fish counts using bait plume size. Marine Ecology Progress Series. 350:255.

Hitt S, Pittman SJ, Brown KA, 2011a. Tracking and mapping sun-synchronous migrations and diel space use patterns of Haemulon sciurus and Lutjanus apo- dus in the U.S. Virgin Islands. Environ. Biol. Fish. 92:525-538.

Hitt S, Pittman SJ, Nemeth RS. 2011b. Diel movements of fishes linked to benthic seascape structure in a Caribbean coral reef ecosystem. Marine Ecology Progress Series, 427:275-291.

Hobson ES. 1965. Diurnal-nocturnal activity of some inshore fishes in the Gulf of California. Copeia. 291-302.

Honda K, Nakamura Y, Nakaoka M, Uy W H, Fortes MD. 2013. Habitat use by fishes in coral reefs, seagrass beds and mangrove habitats in the Philippines. Plos one. 8(8):65735.

Isaacs JD. 1969. The nature of oceanic life. Scientific American. 221(3):146.

Jelbart JE, Ross PM, Connolly RM. 2007. Fish assemblages in seagrass beds are influenced by the proximity of mangrove forests. Marine Biology. 150(5):993-1002.

Kimirei IA, Nagelkerken I, Griffioen B, Wagner C, Mgaya YD. 2011. Ontogenetic habitat use by mangrove/seagrass-associated coral reef fishes shows flexibility in time and space. Estuarine, Coastal and Shelf Science. 92(1):47-58.

Klages J, Broad A, Kelaher BP, Davis AR. 2014. The influence of gummy sharks, Mustelus antarcticus, on observed fish assemblage structure. Environmental Biology of Fishes. 97(2):215-222.

Knip DM, Heupel MR, Simpfendorfer CA. 2010. Sharks in nearshore environments: models, importance, and consequences. Marine Ecology Progress Series. 402:1-11.

Krumme U. 2009. Diel and tidal movements by fish and decapods linking tropical coastal ecosystems. In *Ecological connectivity among tropical coastal ecosystems*. pp 271-324. Springer Netherlands.

Kwei EA. 1978. Food and spawning activity of Caranx hippos off the coast of Ghana. Journal of Natural History. 12(2):195-215.

Ley JA, McIvor CC. 2002. Linkages between estuarine and reef fish assemblages: Enhancement by the presence of well-developed mangrove shorelines pp 539-562 In: Porter and Porter [eds] The Everglades, Florida Bay and Coral Reefs of the Florida Keys: An Ecosystem Sourcebook.

Lirman D, Cropper WP. 2003. The influence of salinity on seagrass growth, survivorship, and distribution within Biscayne Bay, Florida: field, experimental, and modeling studies. Estuaries. 26(1):131-141.

Lirman D, Deangelo D, Serafy J. 2008. Documenting Everglades restoration impacts on Biscayne Bay's shallowest benthic habitats. First Annual Report, University of Miami/Rosenstiel School of Marine and Atmospheric Science, submitted to South Florida Water Management District, West Palm Beach, FL.

Lirman D, Deangelo G, Serafy J, Hazra A, Hazra DS, Herlan J, Clausing R. 2008. Seasonal changes in the abundance and distribution of submerged aquatic vegetation in a highly managed coastal lagoon. Hydrobiologia. 596(1):105-120.

Logan M. 2011. Biostatistical design and analysis using R: a practical guide. John Wiley & Sons.

Lowry M, Folpp H, Gregson M, Suthers I. 2012. Comparison of baited remote underwater video (BRUV) and underwater visual census (UVC) for assessment of artificial reefs in estuaries. Journal of Experimental Marine Biology and Ecology. 416:243-253.

Luckhurst BE, Dean JM, Reichert M. 2000. Age, growth and reproduction of the lane snapper Lutjanus synagris (Pisces: Lutjanidae) at Bermuda. Mar Eco Prog Ser, 203:255-261.

Luo J, Serafy JE, Sponaugle S, Teare PB, Kieckbusch D. 2009. Movement of gray snapper Lutjanus griseus among subtropical seagrass, mangrove, and coral reef habitats. Marine Ecology Progress Series. 380:255-269.

Maciá S, Robinson MP. 2005. Effects of habitat heterogeneity in seagrass beds on grazing patterns of parrotfishes. Marine Ecology Progress Series. 303:113-121.

Madin EM, Dill LM, Ridlon AD, Heithaus MR, Warner RR. 2016. Human activities change marine ecosystems by altering predation risk. Global change biology. 22(1):44-60.

Manooch III CS, Mason DL. 1984. Age, growth, and mortality of lane snapper from southern Florida. Northeast Gulf Sci. 7(1):109-115.

Manooch III, C. S., & Matheson III, R. H. 1981. Age, growth and mortality of gray snapper collected from Florida waters. In Proc. Annu. Conf. Southeast. Assoc. Fish Wildl. Agencies 35:331-344.

Markley S, Milano G. 1985. Biscayne Bay today: a summary report on its physical and biological characteristics. Technical Report to Dade County Department of Environmental Resource Management. pp. 28

McFarland WN, Ogden JC, Lythgoe JN. 1979. The influence of light on the twilight migrations of grunts. Environmental Biology of Fishes. 4(1):9-22.

Mumby PJ, Edwards AJ, Arias-González JE, Lindeman KC, Blackwell PG, Gall, A, Gorczynska MI, Harborne AR, Pescod, CL, Renken H. Wabnitz CC. 2004. Mangroves enhance the biomass of coral reef fish communities in the Caribbean. Nature. 427(6974):533-536.

Mumby PJ. 2006. Connectivity of reef fish between mangroves and coral reefs: algorithms for the design of marine reserves at seascape scales. Biological conservation. 128(2):215-222.

Murphy MD, Murie DJ, Muller RG. 1999. Stock assessment of white grunt from the west coast of Florida. Report to the Florida Fish and Wildlife Conservation Commission.

Nagelkerken I, Blaber SJM, Bouillon S, Green P, Haywood M, Kirton LG, Meynecke JO, Pawlik J, Penrose HM, Sasekumar A, Somerfield PJ. 2008. The habitat function of mangroves for terrestrial and marine fauna: A review. Aquat Bot. 89:155-185

Nagelkerken I, Dorenbosch M., Verberk WCEP, de la Moriniere EC, Van Der Velde G. 2000. Day-night shifts of fishes between shallow-water biotopes of a Caribbean bay, with emphasis on the nocturnal feeding of Haemulidae and Lutjanidae. Marine Ecology Progress Series. 194:55-64.

Nagelkerken I, Kleijnen S, Klop T, Van Den Brand RA, de La Moriniere EC, Van der Velde G. 2001. Dependence of Caribbean reef fishes on mangroves and seagrass beds as nursery habitats: a comparison of fish faunas between bays with and without mangroves/seagrass beds. Marine Ecology Progress Series. (214):225-35.

Nagelkerken I, Roberts CM, Van Der Velde G, Dorenbosch M, Van Riel MC, De La Moriniere EC, Nienhuis PH. 2002. How important are mangroves and seagrass beds for coral-reef fish? The nursery hypothesis tested on an island scale. Marine ecology progress series. 244(2002):299-305.

Nagelkerken I, Van der Velde G, Gorissen MW, Meijer GJ, Van't Hof T, Den Hartog C. 2000. Importance of mangroves, seagrass beds and the shallow coral reef as a nursery for important coral reef fishes, using a visual census technique. Estuarine, coastal and shelf science. 51(1):31-44.

Ogden JC, Ehrlich PR. 1977. The behavior of heterotypic resting schools of juvenile grunts (Pomadasyidae). Marine Biology. 42(3):273-280.

Ogden JC, Quinn TP. 1984. Migration in coral reef fishes: ecological significance and orientation mechanisms. In: McCleave JD, Arnold GP, Dodson JJ, Neil WH, editors. Plenum, New York: Mechanisms of Migration in Fishes. pp. 293-308.

Ogden JC, Zieman JC. 1977. Ecological aspects of coral reef-seagrass bed contacts in the Caribbean. Proceedings of the Third International Coral Reef Symposium. 1:377-382

Parker GG, Ferguson GE, Love SK. 1955. Water resources of southeastern Florida. US Geological Survey water-supply paper. 1255:965.

Patterson AW, Whitfield AK. 2000. Do shallow-water habitats function as refugia for juvenile fishes?. Estuarine, Coastal and Shelf Science. 51(3):359-364.

Pauly D, Froese R. 1998. FishBase 96: concepts, design, and data sources. ICLARM. Manila. pp. 121-123

Robblee MB, Zieman JC. 1984. Diel variation in the fish fauna of a tropical seagrass feeding ground. Bulletin of Marine Science. 34(3):335-345.

Roessler MA, Beardsley GL. 1974. Biscayne Bay: Its environment and problems. Q. J. Fla. Acad. Sci. 37(4):186-204.

Rooker JR, Dennis GD. 1991. Diel, lunar and seasonal changes in a mangrove fish assemblage off southwestern Puerto Rico. Bulletin of Marine Science. 49(3):684-698.

Rutherford ES, Tilmant JT, Thue EB, Schmidt TW. 1989. Fishery harvest and population dynamics of gray snapper, Lutjanus griseus, in Florida Bay and adjacent waters. Bulletin of Marine Science, 44(1):139-154.

Saenger P, Gartside D, Funge-Smith S. 2013. A Review of Mangrove and Seagrass Ecosystems and their Linkage to Fisheries and Fisheries Management. Bangkok: RAP Publication, FAO.

Saloman CH, Naughton SP. 1984. Food of crevalle jack (Caranx hippos) from Florida, Louisiana, and Texas. NOAA Tech Memo. NMFS-SEFSC-134.

Sedberry GR, Carter J. 1993. The fish community of a shallow tropical lagoon in Belize, Central America. Estuaries. 16(2):198-215.

Serafy JE, Faunce CH, Lorenz JJ. 2003. Mangrove shoreline fishes of Biscayne Bay, Florida. Bull Mar Sci. 72(1):161-180.

Serafy JE, Lindeman KC, Hopkins TE, Ault JS. 1997. Effects of freshwater canal discharge on fish assemblages in a subtropical bay: field and laboratory observations. Marine Ecology Progress Series. 160:161-172.

Serafy JE, Valle M, Faunce CH, Luo J. 2007. Species-specific patterns of fish abundance and size along a subtropical mangrove shoreline: an application of the delta approach. Bull Mar Sci. 80(3):609-624.

Serrano X, Grosell M, Serafy JE. 2010. Salinity selection and preference of the grey snapper Lutjanus griseus: field and laboratory observations. Journal of Fish Biology. 76(7):1592-1608.

Short FT, Wyllie-Echeverria S. 1996. Natural and human-induced disturbance of seagrasses. Environmental conservation. 23(01):17-27.

Smith TM, Hindell JS, Jenkins GP, Connolly RM, Keough MJ. 2011. Edge effects in patchy seagrass landscapes: the role of predation in determining fish distribution. Journal of Experimental Marine Biology and Ecology. 399(1):8-16.

Snedaker SC, Biber PD. 1996. Restoration of mangroves in the United States of America: a case study in Florida. Pages 170–188 *in* C. D. Field, ed. 1996. Restoration of mangrove ecosys- tems. ISME, Okinawa, Japan. 304 p.

Snelson FF. 1992. Biological studies of crevalle jack (Caranx hippos) in Florida. St. Petesburg: Florida Marine Research Institute.

Starck WA, Davis WP. 1966. Night habits of fishes of Alligator Reef, Florida. Ichthyologica. 38(4):313-356.

Starck WA. 1971. Biology of the gray snapper, Lutjanus griseus (Linnaeus), in the Florida Keys. Studies in Tropical Oceanography. 10:12-150.

Stump K. 2013. The effects of nursery habitat loss on juvenile lemon sharks, Negaprion brevirostris. Doctoral dissertation. University of Miami.

Tanka SK. 1973. Suction feeding by the nurse shark. Copeia. 606-908

Taylor MD, Baker J, Suthers IM. 2013. Tidal currents, sampling effort and baited remote underwater video (BRUV) surveys: Are we drawing the right conclusions? Fisheries Research. 140:96-104.

Teas HJ. 1977. Ecology and restoration of mangrove shorelines in Florida. Environmental Conservation, 4(01):51-58.

Thompson R, Munro JL. 1974. The biology, ecology and exploitation and management of the Carribean reef fishes. Research Report. Zoology Department University of West Indies, 3:1-43.

Thompson R, Munro JL. 1983. The biology, ecology and bionomics of the snappers, Lutjanidae. Caribbean coral reef fishery resources. ICLARM Studies and Reviews. 7:94-109.

Torres LG, Heithaus, MR, Delius B. 2006. Influence of teleost abundance on the distribution and abundance of sharks in Florida Bay, USA. Hydrobiologia. 569(1):449-455.

Tulevech SM, Recksiek CW. 1994. Acoustic tracking of adult white grunt, Haemulon plumieri, in Puerto Rico and Florida. Fisheries research, 19(3):301-319.

Unsworth RK, Salinas De Leon P, Garrard SL, Jompa J, Smith DJ, Bell JJ. 2008. High connectivity of Indo-Pacific seagrass fish assemblages with mangrove and coral reef habitats. Mar Eco Prog Ser. 353:213.

Unsworth RK, Wylie E, Smith, DJ, Bell JJ. 2007. Diel trophic structuring of seagrass bed fish assemblages in the Wakatobi Marine National Park, Indonesia. Estuarine, Coastal and Shelf Science. 72(1):81-88.

Verweij MC, Nagelkerken I, Hol KEM, van den Beld AHJB, van der Velde G. 2007. Space use of Lutjanus apodus including movement between a putative nursery and a coral reef. Bull Mar Sci. 81:127–138

Watson DL, Harvey ES, Anderson MJ, Kendrick GA. 2005. A comparison of temperate reef fish assemblages recorded by three underwater stereo-video techniques. Marine Biology. 148(2):415-425.

Waycott M, Duarte CM, Carruthers TJ, Orth RJ, Dennison WC, Olyarnik S, Calladine A, Fourqurean JW, Heck KL, Hughes AR, Kendrick GA, Kensworthy WJ, Short FT, Williams SL. 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proceeding of the National Academy of Sciences. 106(30):12377-12381

Zhang JZ, Kelble CR, Fischer CJ, Moore L. 2009. Hurricane Katrina induced nutrient runoff from an agricultural area to coastal waters in Biscayne Bay, Florida. Estuarine, Coastal and Shelf Science. 84(2):209-218.