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A Quantitative Assessment of the January 2010 Cold Spell Effect on Mangrove Utilizing Coral Reef Fishes from Biscayne National Park, Florida

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NOVA SOUTHEASTERN UNIVERSITY OCEANOGRAPHIC CENTER

A QUANTITATIVE ASSESSMENT OF THE JANUARY 2010 COLD SPELL EFFECTS ON MANGROVE UTILIZING CORAL REEF FISHES FROM BISCAYNE NATIONAL PARK, FLORIDA

By

Jeffrey M. Ellis

Submitted to the Faculty of
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in partial fulfillment of the requirements for
the degree of Master of Science with a specialty in:

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Coastal Zone Management

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Jeffrey M. Ellis

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Abstract

This study examined the effects of the January 2010 cold spell on mangrove utilizing coral reef fishes off the southeast coast of Florida, USA, in the vicinity of Biscayne National Park (BNP). An ongoing, fishery-independent mangrove visual survey documenting fish assemblages in BNP provided data from the years 1998 to 2014 for examination. Of particular interest were the presence, abundance, and size structure for five mangrove utilizing coral reef fishes: sergeant major (*Abudefduf saxatilis*), yellowfin mojarra (*Gerres cinereus*), schoolmaster snapper (*Lutjanus apodus*), gray snapper (*Lutjanus griseus*), and great barracuda (*Sphyraena barracuda*). These species were selected for analysis due to their economic and ecologic importance, their potential as environmental indicators, their connectedness to multiple habitats, and their abundance within the available data set. Data were collected using a modified visual 'belt transect' method, consisting of 60 m^2 transects running parallel to the mangrove shorelines. Data for average length of fish were reconstructed to form standard normal distributions and the resulting lengths were assigned to various age-classes to create species-specific length-frequency distributions. Variations in presence and abundance were examined across three time periods (1998-2009; 2010-2011; 2012-2014), as well as comparisons of length-frequency distributions. Following the January 2010 cold spell, the presence and abundance values for the two years immediately following the event were significantly decreased compared to the years prior to the cold spell for most of the five species at either mainland (ML) or leeward key (LK) locations. Additionally, the presence and abundance estimates typically remained statistically decreased when compared against the remaining years in the available data set. The size structures for the majority of the five species at either location, however, were not consistently significantly different between the three time periods, as was hypothesized. Instead, the analyses showed mixed results, with the size structure typically shifting towards smaller individuals immediately following the event. These findings suggest that drops in water temperature resulting from cold spells are capable of directly impacting mangrove utilizing reef fish species, albeit to varying degrees depending on various factors, such as physiological tolerances, ecological life history strategies, and habitat requirements.

Keywords: cold spell, visual survey, Florida, mangroves, essential fish habitat, sergeant major, yellowfin mojarra, schoolmaster snapper, gray snapper, great barracuda

Introduction

This project focused on the January 2010 cold spell, a rare episodic stressor, through statistical analyses of fishery-independent mangrove survey and water temperature data in an attempt to determine alterations in the presence, abundance, and size structures of five mangrove utilizing coral reef fishes of economic and ecologic importance inhabiting Biscayne Bay, Florida. The primary analyzed data set focused on mangrove fish monitoring data from within BNP, consisting of counts of relative abundance and minimum, average, and maximum total length values. The data were sorted by habitat over alternating dry and wet seasons from 1998-2014 to allow for comparative analyses. We examined whether changes in presence, abundance, or size structure of selected reef fishes coincided with the January 2010 cold spell and if so, how long it took for levels to return to pre-stress values. Also examined were water temperature data from two YSI stations to assess the severity of the January 2010 cold spell and any other potential cold periods not addressed in the literature. This study provides insight into the impacts of low temperature extremes on coral reef fishes that utilize mangrove habitats as nursery and feeding grounds.

The current body of local literature pertaining to cold spells that impact fish has been mostly anecdotal and only recently included quantified effects. This is primarily due to the absence of data prior to these events and the inability to perform proper or standardized monitoring techniques (e.g., Willcox 1887; Finch 1917; Storey and Gudger 1936; Storey 1937; Miller 1940; Galloway 1941; Gilmore et al. 1978). Previous studies pertaining to oceanic temperature changes, specifically in relation to reef fish and the environments they inhabit, have focused primarily on the negative effects seen from increasing seawater temperatures (see Glynn 1985; Jokiel and Coles 1990; Goreau et al. 2000). This may be due to the gradual, and therefore persistent, alteration and effects of water temperatures during warming periods, whereas cold spells are brief in duration and rare in occurrence. Warm-water fluctuations have been documented as causing severe damage to ocean systems, such as coral bleaching, thereby leading to the demise of entire reef systems. The term “bleaching hotspot” has been used to describe increased sea surface temperatures (SST) of 1.0 °C or more above the expected temperatures (Goreau et al. 2000) and bleaching has been recorded in locations where water temperature has elevated 3-4 °C for short, 1-2 day durations (Jokiel and Coles 1990). However, no such definition of abrupt decreases in water temperatures exists. This study argues that these conditions associated with cold spells could be as great of a threat to marine ecosystems as their warm-water counterparts. Even though ocean warming is related to climate changes and cold spells are related to brief weather changes, alterations of increased and decreased water temperature should be studied in a similar fashion for the purpose of understanding physiological and/or behavioral responses in fish, especially in habitats that sustain important species and are more susceptible to environmental changes.

The most recent documented cold spell period in the South Florida region occurred during January 2010.[1] This study examined the effects of that cold spell on mangrove utilizing reef fishes from the southeast coast of Florida, United States, specifically from the BNP region. Statistical analyses determined whether the drop in temperature altered the presence, abundance, or size structure of selected coral reef fishes from the mangrove locations in which they reside. From the mangrove survey data, pre- and post-event analyses yielded species- and habitat-specific outcomes. Study results also provided insight into the connections between specific coral reef species and their use of adjacent mangrove ecosystems. The resulting effects from this environmental anomaly, such as declines in important fish populations, may contribute to the process of altering fishery policies to reduce stress on diminished stocks. Given the location of the available data set, the results only apply to the South Florida geographic area. However, the findings could be generalized to other tropical and subtropical linked mangrove and reef complexes within areas of like water temperature conditions inhabited with similar species.

Cold Spell Definition

Throughout the literature, there have been many references to cold spells, yet there is no single accepted definition to describe what actually constitutes a “cold spell” or “cold wave.” The most basic definition refers to a short period of time during which

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[1] Although there is no single accepted definition for the “South Florida” geographic region, this thesis used the term to encompass Monroe, Miami-Dade, Broward, and Palm Beach counties in southeastern Florida that contain a tropical climate, an area roughly corresponding to Marco Island/Naples on the western Florida coast diagonally northeastward to the Palm Beaches, including all of the Florida Keys and the Everglades National Park.

the air temperature of a certain area dropped significantly below the recorded average temperature. By this definition, the temperature at which a cold spell occurs would be different for varying regions and seasons based on climate. This definition also fails to take into account if the average used for comparison is a seasonal, monthly, or daily average. The other aspect of this definition is the length of time that the temperature is at or below this pre-determined level. This results in a lack of clarity as to whether a brief temperature dip below this level is considered a cold spell or if instead the temperature must remain below this point for an extended period of time.

The lack of an operational definition for a cold spell exists in other regions of the world: central Europe (Kysely et al. 2009), the Netherlands (Huynen et al. 2001), and China (Ma et al. 2012) to name a few. Without an agreed-upon definition for this phenomenon, various approaches have been utilized. One method is to define the cold spell as a set minimum number of consecutive days below a percentile of either minimum or average daily temperatures (e.g., Guo et al. 2012; Ma et al. 2012), while others require a minimum number of days below a set temperature for a specific region (e.g., Huynen et al. 2001), and still others use calculations of the mean and standard deviation of maximum daily temperatures (e.g., McCalla et al. 1978). Cold spells are typically determined by drops in air temperature, yet when assessing for effects on marine organisms, the more important variable is water temperature. During a cold spell, water temperatures will change at varying rates, to different minimums, and at lagged times compared to air temperatures. Water temperatures are also affected by multiple factors, such as the depth and temperature of surrounding water and the level of mixing that occurs due to wind speed and direction. In relation to the physiological tolerance of fish to decreased temperatures, a slower temperature decrease would allow more acclimation time, yielding lower cold-lethal limits (Moore 1976). With so many possible interpretations of what constitutes a cold spell, comparisons become difficult from both chronologic and geographic perspectives.

Cold Spell Etiology and Effects

Atmospheric cold spells can be attributed to meteorological occurrences. For example, according to the National Oceanic and Atmospheric Administration, the

January 2010 cold spell experienced in South Florida was the product of simultaneously occurring El Niño and negative Arctic Oscillation (AO) climate events (NOAA 2010). The El Niño-Southern Oscillation (ENSO) is both a fluctuation in sea surface temperature and air pressure of the overlying atmosphere in the equatorial Pacific Ocean. The AO alters climate in the northern hemisphere with the presentation of low pressures at mid-latitudes, known as a negative phase, which allows cold Arctic air to move south (NOAA 2010). In combination with both events, a high amplitude flow pattern prolonged the cold spell by keeping the cold air in place from December to February (NOAA 2010). North America, Europe, and Asia experienced record low temperatures in multiple cities from December 28, 2009 through January 13, 2010 (Wang et al. 2010).

When assessing cold spell effects on the marine environment, consideration of the heat exchange process between air and water is necessary. A shallower, smaller body of water such as a bay or lagoon is subject to greater water temperature changes during a shorter period of time as compared to deeper, larger bodies of water such as a benthic habitat in the open ocean. This is due to the former having limited heat storage capacity (Roberts et al. 1982). Therefore, a cold spell will have different effects on the varying habitats and their occupants. Some regions may suffer vegetation and animal loss during a cold spell, whereas other regions may experience relatively few to no changes during the same event. These unaltered regions may only be affected during more severe cold spells of greater temperature variations and/or longer durations. Alternatively, regions not directly affected by decreased air temperatures may be indirectly affected due to mixing of surrounding water. Circulatory flow patterns can mix the cooled shallow water into surrounding regions, reducing the temperature of nearby habitats. This surge of water can be initiated by strong northerly winds that usually complement cold spells with their rate of transport depending on tides and density gradients (Walker et al. 1982).

The winter cold spell of January 2010 is defined in the literature as a period of unusually low temperatures for the specified season in and around the study area and was observed to directly affect coral reef habitats and fishes on the southeast coast of Florida (e.g., Lirman et al. 2011; Adams et al. 2012; Colella et al. 2012). For the purposes of the following literature review on previous events, any designation of a cold spell noted by authors was accepted and comparatively examined.

Review of Assessments

Early studies from the 19th and 20th centuries consisted of anecdotal observations, typically from secondhand sources (Willcox 1887; Finch 1917; Storey and Gudger 1936; Storey 1937; Miller 1940; Galloway 1941; Gilmore et al. 1978). These works noted a range of dates during which air temperatures were colder than usual, a non-standardized estimate of the number of dead fish washed onto shores or floating on the surface in a given area, and a list of the most prominent of the affected species. No definitions were given for the term "cold spell" and documentation of affected fishes was also difficult due to the inaccuracies of species identification and the existence of multiple common names for a single species. Exact air and water temperatures were rarely recorded or accessible during the cold periods (Gilmore et al. 1978) making quantitative assessment difficult.

Previous Research on Cold Spells

Currently, the local literature pertaining to marine organisms and their habitats during cold spells encompasses 20 events in the South Florida and Texas Gulf Coast regions (Table 1). Although the current study is specific to the eastern coast of South Florida, the Gulf Coast areas of similar geographic range were included in the examination in addition to the Atlantic locations due to the small sample size and the presumed similarities pertaining to the cold effects.

Late 1800s

As early as the late 1800s, scientists began documenting the observed effects of such cold spells. Packard (1871) reported on two cold periods near Key West on December 24-25, 1856 and December 25, 1868. During the first cold spell, the air temperature was 6.7 °C and several reports were given of multiple fishes killed and washed ashore (though no specific species were notated). During the second event, the area suffered a frost and dead fish were again found ashore. Of importance, these studies represent some of the first examples of scientists seeking to understand and explore cold spells and the effects on the ocean ecosystem.

Table 1. Previous cold spells documented in the literature for the South Florida and Texas Gulf Coast regions. Note that “cold spell” for the purposes of this literature review was defined by the individual authors, not an empirical standard.

Cold Spell Date	**Source**	**Location**	**Min Water Temp (°C)**	**Min Air Temp (°C)**	**Duration (days)**
Jan. 12, 1886	Storey & Gudger, 1936	Sanibel Island, FL	N/A	N/A	N/A
Jan. 12, 1886	Finch, 1917	Key West, FL	N/A	5.0	1
1886-1887	Willcox, 1887	Cedar Keys to Punta Rassa, FL	N/A	N/A	N/A
Dec. 29-30, 1894	Storey & Gudger, 1936	Sanibel Island, FL	N/A	-2.2	2
Feb. 9, 1895	Storey & Gudger, 1936	Sanibel Island, FL	N/A	1.1	1
Jan. 3, 1898	Storey & Gudger, 1936	Sanibel Island, FL	N/A	-3.3	1
Feb. 14, 1899	Storey & Gudger, 1936	Sanibel Island, FL	N/A	0.0	1
Jan., 1905	Storey & Gudger, 1936	Sanibel Island, FL	N/A	-1.1	N/A
Feb. 2-4, 1917	Finch, 1917	Atlantic and Gulf coasts, FL	N/A	6.4	3
Feb. 3-4, 1917	Storey & Gudger, 1936	Sanibel Island, FL	N/A	-1.7	2
Jan. 15-16, 1928	Storey & Gudger, 1936	Sanibel Island, FL	N/A	0.3	2
Dec., 1934	Storey & Gudger, 1936	Sanibel Island, FL	N/A	-1.1	N/A
Jan. 27-29, 1940	Miller, 1940	Miami to Key West, FL	N/A	-0.6	3
Jan., 1940	Galloway, 1941	Key West, FL	13.9	6.1	N/A
Jan., 1940	Gunter, 1941	Aransas Bay, TX	4.7	1.5	N/A
Jan. 1, 1942	Gunter, 1947	Copano Bay, TX	N/A	N/A	1
Jan. 28-Feb. 3, 1951	Gunter, 1951	Aransas Pass, TX	3.3	-7.8	6
Jan. 11-12, 1973	Moore, 1976	Port Aransas, TX	0.5	N/A	2
Jan. 18-24, 1976	Gilmore et al., 1978	Indian River Lagoon, FL	13.0	0.0	7
Jan. 5-Feb., 1977	Bohnsack, 1983	Big Pine Key, FL	10.0	N/A	N/A
Jan. 15-27, 1977	Roberts et al., 1982	Florida Bay Reef Tract, FL	12.6	2.0	13
Jan. 19-24, 1977	Gilmore et al., 1978	Sanibel Island, Tampa Bay, and Indian River Lagoon, FL	6.0	-3.9	6
Jan.-Feb., 1977	Bullock & Smith, 1979	Tampa Bay, FL	12.0	N/A	N/A
Jan.-Feb., 1978	Bullock & Smith, 1979	Tampa Bay, FL	12.0	N/A	N/A
Jan. 8-21, 1981	Walker et al., 1982	Florida Bay and Reef Tract, FL	8.7	5.6	14
Jan. 10-14, 1982	Holt & Holt, 1983	Port Aransas, TX	5.5	-5.5	5
Jan. 2-14, 2010	Adams et al., 2012	Southwestern FL	<12.0	N/A	13
Jan., 2010	Lirman et al., 2011	Florida Reef Tract, FL	9.5	1.7	N/A
Jan., 2010	Colella et al., 2012	Florida Keys, FL	8.8	N/A	N/A

By the 1880s, scientists began reporting more specifically on the myriad of species affected by the cold spells.[2] The January 12, 1886 cold spell was documented along two different regions: the first by Willcox (1887) along the coast from Cedar Keys to Punta Rassa, and the second by Storey and Gudger (1936) and Finch (1917) in Key West. Willcox (1887) noted the great mortality of cavally (crevalle jack *Caranx hippos*), skipjacks (ladyfish *Elops saurus*), redfish (red drum *Sciaenops ocellatus*), sea-trout (spotted seatrout *Cynoscion nebulosus*), sheepshead (*Archosargus probatocephalus*), and large tarpon (*Megalops atlanticus*) on the shores of rivers and bays. “High numbers” of dead cowfish (scrawled cowfish *Acanthostracion quadricornis*) and balloon-fish (striped burrfish *Chilomycterus schoepfii*) on the shores of the Gulf of Mexico were also observed. Storey and Gudger (1936), as well as Finch (1917), noted an air temperature of 5.0 °C in Key West with reports of “thousands” of fish washing onto shore. Anecdotally, the authors interviewed the elders of that area who referred to the freeze as the worst on record that they had witnessed.

Sanibel Island Region

Storey and Gudger (1936) documented each of the following eight cold spells pertaining to marine environments, ranging from 1894 to 1934. The following reports are based on their publication. In December 29-30, 1894 scientists saw temperatures drop in the Sanibel Island region. Air temperatures were recorded at -2.2 °C near the water with an official minimum at Fort Myers of -4.4 °C. Shallow and deeper water species were both killed, including tarpon, gafftopsail catfish (*Bagre marinus*), silver mullet (white mullet *Mugil curema*), permit (*Trachinotus falcatus*), sharks, and Atlantic goliath grouper (*Epinephelus itajara*). The majority of the dead fishes came ashore the day after the cold period. Several weeks later, on February 9, 1895, temperatures again dropped in the Sanibel Island region to 1.1 °C near the water and -1.1 °C at Fort Myers. Although fewer fish were affected during this period compared to the cold spell several weeks prior, the fish that did suffer mortalities were washed ashore in a numbed state and could not return to the water.

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[2] The common names of the fish species documented from the previous cold spells were taken directly from the respective literature to avoid misinterpretation. However, American Fisheries Society (AFS) standard common and current scientific names are included in parentheses.

Three years later, witnesses reported temperatures as low as -3.3 °C with an official record at Fort Myers of -2.2 °C during the cold spell on January 3, 1898 over Sanibel Island. Mortality among shallow water fishes was “high”, especially among jack (*Caranx* sp.), snappers, gafftopsail catfish, cowfish, snook (*Centropomus undecimalis*), and eels. The following year, Sanibel Island was again hit with a cold spell on February 14, 1899. An official Fort Myers temperature record of -2.2 °C resulted in “many” fish being numbed, but not killed.

Another cold spell for the Sanibel Island region was not documented until January 26-27, 1905 with an official low of -2.8 °C at Fort Myers. Smaller fishes suffered the greatest casualties during this period with needlefish (*Strongylura* sp.) being most affected. The following cold spell in February 2-4, 1917 resulted in a low temperature of -1.7 °C with an official temperature reading at Fort Myers of -2.8 °C. “Countless” numbers of snappers and “many” grouper were killed, with the dead fish washing onto shore. Finch (1917) noted this cold period on both coasts of South Florida and reported an observed low of 6.4 °C at Key West. The next cold spell at Sanibel Island occurred January 15-16, 1928. Fort Myers recorded a low temperature of -1.1 °C on January 4 and Sanibel Island recorded a low of 0.3 °C during the cold period. “Many” species were killed outright and washed ashore, including catfish, snook, trunkfish (*Lactophrys* sp.), skipjacks, blue runners (*Caranx crysos*), and dwarf sand perch (*Diplectrum formosum*), while other species suffered “minor” casualties.

The final recorded cold spell at Sanibel Island occurred December 12, 1934. The temperature dropped suddenly to -1.1 °C near the water, leading to the “high” mortality levels of sheepshead and schoolmaster snapper (*Lutjanus apodus*). Fort Myers recorded an official reading of -1.7 °C on the same day. Reports show many fish were temporarily paralyzed, but a lack of wind prevented the fish from washing onto shore, resulting in no known mortalities.

1940

Miller (1940) was the first to include air temperatures and the observed effects on vegetation. He relied on first-hand observations and reports from “reliable” fishermen to assess the cold spell during January 27-29, 1940 from Miami down to Key West. Reports of air temperatures ranged from -0.6 °C to 13.9 °C for this particular cold spell. Minimum air temperatures were documented at -0.6 °C for Miami, 3.3 °C for Elliott Key, and 10.0 °C for Key West. Designations of "many," "some," and "a few" were given to the fishes observed to be stunned or killed. Those suffering "many" casualties included bonefish (*Albula vulpes*), moonfish (*Selene vomer*), mutton snappers (*L. analis*), lane snappers (*L. synagris*), gray snappers, grunts (*Haemulon* sp.), porgies (*Calamus* sp.), and mullet. Parrotfish (Family Scaridae), trunkfish, swellfish (*Spheroides* sp.), and needlefish suffered "some" losses. Grouper, filefish (*Stephanolepis hispidus*), brim (sea bream *Archosargus rhomboidalis*), barracuda (*Sphyraena* sp.), and jacks suffered "few" losses. It was noted, however, that the observations were possibly prejudiced due to reports of fishermen only collecting stunned specimens that were of commercial value. This event was later assessed by Galloway (1941) to include water temperatures, with a low of 6.1 °C on January 28, 1940 according to the Key West Weather Station and a minimum of 13.9 °C on the same date. The author also observed that "many" species were dead or washed onto shore.

1970s – 1980s

The next chronological cold spell documented in the literature was described by Bohnsack (1983) and took place 37 years after the 1940 event. Six model patch reefs had previously been constructed during the summer of 1975 in locations south of Big Pine Key. These reefs were visually monitored from June 1975 through September 1978, allowing for one of the first examinations of both pre- and post-cold spell event data for the South Florida area. A monthly fish census was conducted as often as possible and parameters such as species richness, total number of individuals, total estimated biomass, and mean fish length were recorded. Data taken prior to the cold spell were compared to the first census data post-event to determine whether individuals and species disappeared due to the cold.

The first cold period passed through the study location on January 5, 1977. A front then stalled over the area on January 15 and remained in the area for a few days. A minimum water temperature at the study site of 11.0 °C was recorded on January 20. The author noted that most fish deaths in the area occurred from January 20-23. The species that suffered casualties included bluestriped grunt (*Haemulon sciurus*), sailors choice (*Haemulon parrai*), schoolmaster snapper, and rainbow parrotfish (*Scarus guacamaia*). The results of a one-tailed t-test showed a statistically significant decline in mean number of species ($p<0.01$) and mean number of individuals per reef ($p<0.025$). The average fish length also increased significantly following the event. By 1978, the affected parameters had returned to previous levels (difference was non-significant). Based on the results, the author suggests that the cold disturbance reduced the number of residents on the reefs and the following summer allowed for an increase in juvenile recruitment success. This was due to reduced mortality, a decrease in predators, and a decrease in competition for both food and space.

Walker et al. (1982) examined the thermal response of Florida Bay and the Florida reef tract to a cold-air period from January 8-21, 1981 in the South Florida region. A water temperature minimum was reached of 8.7 °C on January 13. Of note were the "extensive" coral and fish mortalities during this period, including deaths of individual colonies of boulder star coral (*Montastraea annularis*), lettuce coral (*Agaricia agaricites*), and mustard hill coral (*Porites astreoides*).

Roberts et al. (1982) documented another cold spell from the same year on January 15-19, 1977. Stations at Florida Bay and the "Hen and Chickens" patch reef on the Atlantic side of the Keys provided *in situ* water temperatures during the study period. Although pre- and post-event water temperatures were not directly recorded, they were estimated using meteorological data and a computerized heat flux model. The minimum water temperatures recorded for the Florida Bay location and the shallow shelf station in the Keys were 12.9 °C on January 20 and 12.6 °C on January 24, respectively. Water temperatures remained below 16.0 °C for an eight day period for both locations. The authors determined that the boulder star corals of the area suffered cold induced stress, recorded as high-density stress bands. Personal observations by the authors near the stations indicated recently killed colonies of elkhorn coral (*Acropora palmate*) and staghorn coral (*A. cervicornis*) in May of the same year, which they believe to be caused by the 1977 cold spell event.

Gilmore et al. (1978) were the first to quantitatively estimate the number of killed fishes following a cold spell with the use of collection methods that incorporated accepted abundance categories. The authors combined detailed air and water temperature records in an attempt to reveal trends that led to hypothermal stress and mortality in marine fishes during cold periods in South Florida. The authors compared both coasts of Florida, including the Tampa Bay and Sanibel Island regions on the Gulf coast, and the Indian River Lagoon region on the Atlantic coast, during the cold spell from January 19-February 13, 1977. Minimum air temperatures recorded occurred on January 19 at St. Petersburg and on January 20 at Vero Beach, of 1.7 °C and -3.9 °C, respectively. A minimum water temperature of 6.0 °C was reached in the Indian River Lagoon on January 19. Minimum air and water temperatures were reached at Sanibel Island on January 22 at 8.3 °C and 10.6 °C, respectively. Following the cold period, the authors collected and estimated the number of killed fish by species. Fifty-six species suffered casualties in the Indian River Lagoon area, 36 species in the Tampa Bay area, and 19 species in the Sanibel Island region.

Bullock and Smith (1979) also documented a cold spell from January-February, 1977, as well as the following year's cold spell in the same region from January-February, 1978. Self Contained Underwater Breathing Apparatus (SCUBA) observations were conducted both years off the west coast near Tampa Bay with water temperatures remaining around 12.0-13.0 °C for several weeks. "Limited" kills were observed of the more sensitive tropical reef species, such as butterflyfishes (Family Chaetodontidae) and angelfishes (Family Pomacanthidae). However, no pre- or post-event data were collected for comparative purposes.

Contemporary Studies

Another prolonged period without a documented cold spell in the South Florida region then occurred, with the next cold spell occurring in January 2010. Three primary studies have focused on this time period, all with varying accounts of start and end periods, duration, minimum temperatures recorded, and other environmental parameters. Lirman et al. (2011) documented the cold period during January 2010 and the associated large-scale coral mortality along the Florida Reef Tract. A minimum air temperature was recorded of 1.7 °C at the Miami International Airport on January 10, 2010 with water temperatures on the Florida Reef Tract falling below 16.0 °C for up to six days.

Following the cold period, reef surveys were conducted from Martin County to the Lower Florida Keys and compared to prior surveys in the same areas taken since 2005. The greatest coral mortalities were suffered in inshore habitats where the temperatures fell below 11.0 °C for prolonged periods of time. This was the first account of coral mortality over large spatial and taxonomic scales due to a cold anomaly.

Adams et al. (2012) documented the cold spell to encompass a 13-day period beginning on January 2, 2010, which affected the southwestern coast of Florida. Water temperatures were averaged from four sensors near the Charlotte Harbor study site and recorded temperatures below 12.0 °C from January 6-16. Three years of pre-event and one year of post-event data were used to assess the effects on the common snook, revealing a statistically significant decrease in apparent survival of the species near the study site.

Colella et al. (2012) made use of data from the Coral Reef Evaluation and Monitoring Project (CREMP), which surveys benthic communities throughout the Florida Keys National Marine Sanctuary, to examine the prolonged cold period of January 2010. The lowest recorded water temperature during the 12-day cold spell was 8.7 °C at Long Key observing station in Florida Bay. The authors compared in situ temperature data to the recorded coral cover and demographics of the area before and after the event to show the significant decrease in percent live coral cover between the summer of 2009 and February 2010. The coral species showing greatest mortality were boulder star and mustard hill corals, while boulder brain coral (*Colpophyllia natans*) and massive starlet coral (*Siderastrea siderea*) were most tolerant of the cold exposure.

The current study also examined this most recent cold spell in the South Florida area. Presence, abundance, and size structure of specific coral reef fishes were analyzed from two separate mangrove shorelines within BNP. The prior studies focusing on the January 2010 cold spell were conducted using data from coral reef locations (Lirman et al., 2011; Colella et al., 2012) and from tagged fish collected during seine fishing. As such, this study is the first to examine the effects of the January 2010 cold spell from data collected within mangrove locations.

Mangrove Habitats

Difficulties pertaining to the sampling of fish within mangrove prop-root habitats have impeded studies focused around these locations (Faunce and Serafy 2006) and therefore have not been incorporated into cold spell analyses centering on affected fauna. This is primarily attributed to the fact that the entangled prop roots and low visibility from sedimentation and detritus make these locations difficult data collection sites.

Mangroves are limited to regions of specific climates and occur in areas of varying saltwater concentrations, tidal fluctuations, and substrates. Additionally, mangroves usually cannot tolerate temperature fluctuations greater than 10.0 °C or temperatures below freezing for any length of time (Odum et al. 1982). The mangrove prop root systems experience semi-diurnal tides (Thayer et al. 1987), which inundate the roots, allowing marine organisms to exist in and around the flora. Dennis (1992) defined a mangrove habitat as not only containing this prop-root system, but also exhibiting adjacent muddy bottom areas that develop from the associated deposition processes. Since these habitats occur under marine, estuarine, and freshwater conditions, they are utilized by a number of fishes at various life history stages (Faunce and Serafy 2006).

Some species may utilize prop root systems as temporary daytime refugia (Faunce and Serafy 2006) or for feeding purposes (Jaxion-Harm et al. 2013), yet there is a greater abundance and diversity among fishes within red mangrove prop roots at night compared to day (Thayer et al. 1987). Other reasons fish utilize mangrove locations include abundance of food (Laegdsgaard and Johnson 2001), increased shade and turbidity (Helfman 1981), and reduced pressure caused by predation (Laegdsgaard and Johnson 2001). This predation, specifically on early recruits of marine fish, reduces densities among mangrove and seagrass habitats and influences the size distributions of the inhabiting species and also affects competition (Hoey and McCormick 2004; McCormick and Hoey 2004; Hixon and Jones 2005; Almany and Webster 2006). These shallow back-reef habitats, however, have yielded varying results in terms of predation pressure.

Studies have suggested that these habitats have lower predation pressure compared to adjacent coral reef habitats, leading to higher post-settlement juvenile fish survival (Beets 1997; Dahlgren and Eggleston 2000), whereas other studies have shown relatively high densities of piscivores in shallow water estuaries (Nakamura and Sano

2004; Kulbicki et al. 2005; Baker and Sheaves 2007; Newman et al. 2007; Dorenbosch et al. 2009; Payne and Gillanders 2009; Unsworth et al. 2009). Recruitment of fish into mangrove habitats and subsequent ontogenetic shifts to other habitats during different life stages can be easier in open systems (Gratwicke and Speight 2005); however, higher predation pressure also exists compared to semi-enclosed systems, such as bays (Bullard and Hay, 2002; Belmaker et al. 2005). This is especially true if these habitats are adjacent to coral reef locations, facilitating the entry of piscivores (Baker and Sheaves 2007; Dorenbosch et al. 2007; Valentine et al. 2007). Aside from entry from reefs for feeding purposes, piscivores found in back-reef habitats may originally settle here as early juveniles before contributing to predation pressures later on in their life stages (Baker and Sheaves 2005).

Aside from periodic use in adulthood, coral reef fishes have been widely documented as utilizing mangroves and other inshore areas primarily as nursery habitats (Weinstein and Heck 1979; Odum et al. 1982; Stoner 1983; Sogard et al. 1987; Morton 1990; Dennis 1992; Eggleston 1995; Ault et al. 1998; Ley et al 1999; Bohnsack et al. 1999; Nagelkerken et al. 2000; Dahlgren and Eggleston 2001; Laegdsgaard and Johnson 2001; Mumby et al. 2004; Faunce and Serafy 2006; Dorenbosch et al. 2007). The definition of what makes a habitat a nursery area for demersal species is varied, but can include the presence of larvae or juveniles, the possibility for settlement, growth, and development, a low mortality rate for the occupants from predation or starvation, and shade to protect from high UV light levels (Dennis 1992; de la Morinière et al. 2004; Verweij et al. 2006). Beck et al. (2001) suggested that based on the previous research, an area is considered a nursery habitat if juvenile fishes or invertebrates occur at higher densities, the area allows for better success at avoiding predators, or the species grow faster compared to when they inhabit different locations. However, the authors developed their own hypothesis on the matter, stating that a habitat can be considered a nursery for a particular species if the production of individuals per unit area is on average greater than production from other habitats where juveniles also occur.

Dahlgren et al. (2006) also detailed various definitions of nursery habitats and build on the concept of Beck et al. (2001) by evaluating juvenile habitats based on their overall contribution to adult populations. Adams et al. (2006) even examined the “nursery function" of various habitats, which they defined as "the growth and survival of juvenile fishes and invertebrates followed by successful ontogenetic habitat shift into their adult habitat." Reef fishes in the western Atlantic, such as snappers (Family Lutjanidae), tend to follow four phases of habitat shifts: (1) spawning on reefs as adults; (2) migration and settlement into nearshore locations, such as seagrass beds and hardbottoms; (3) movement to nearby mangrove shorelines; (4) migration back to reefs upon sexual maturity (Ogden 1997, Lindeman et al. 1998). This movement between habitats is known as an ontogenetic shift or migration (Adams et al 2006; Jaxion-Harm et al 2012).

Mangroves are also thought to provide valuable nutrients to growing juveniles through a detritus-based food web (Odum et al. 1982; Laegdsgaard and Johnson 2001) with larger juveniles feeding on an assemblage of zooplankton (Dennis 1992). Although their use may not be a necessity for young reef fish, juveniles seem to show a preference for mangroves and seagrass beds when available (Nagelkerken et al. 2002). A study by Laegdsgaard and Johnson (2001) found that small juvenile fish preferred mangrove locations to seagrass habitats due to their increased foraging success at the former location. Upon growth, juveniles showed a preference for mud flats, presumably because of the increased restriction experienced by the larger fish in the dense and intricate prop root systems of mangroves. This habitat separation allows otherwise vulnerable organisms to develop to a mature size at each stage, increasing their chance of survival upon their return to the high predation of coral reef environments (Dennis 1992; Mumby et al. 2004).

In addition to the presence of mangroves, the location of the mangrove stands has also been shown to regulate species richness and total number of juvenile reef fish. Faunce and Serafy (2008b) examined five different shoreline types based on various cross-shelf positions in the fringing mangroves of southeastern Florida. Fishes were sampled by means of a visual survey, in which each species was enumerated and estimates of minimum, average, and maximum total length were recorded. The study focused on the effects of the proximity of the mangroves to creek mouths and oceanic inlets and on seasonal differences on the species of snapper, grunt, and great barracuda. The study found that all species examined exhibited either a seasonal or spatial preference to varying mangrove locations, revealing that not all mangrove shorelines are equivalent in terms of the value they hold as fish habitats.

Although these findings depict mangroves as a top priority to juvenile reef fish in their shallow water sites, a study by Nagelkerken and Faunce (2008) used artificial "mangrove units" – constructed in various depths and locations and with root structure both present and not present – to determine what specifically makes mangroves attractive to this grouping of fish. Based on their findings, the authors rejected the ideas that fish are attracted to mangroves solely for their shallow water and confined embayment locations, but instead accepted the idea that the structure of the prop roots attracted the juvenile fish.

Cold Effects on Fish

The most documented aspect of the South Florida cold spells in the literature is the effect on fish, most likely due to the importance of commercial and recreational fishing in the area. Water temperature fluctuations alter the behavior of fish species and it is widely accepted that high levels of mortality can occur in extreme cases. As previously discussed, the literature on cold spells contains mostly anecdotal reports of affected fish species. Observations often included estimates of numbed or dead fish floating on the surface or washing onto shore, or declines in fish catches immediately following a cold spell. Storey and Gudger (1936) noted that fishing was usually "poor" during the first few weeks following a freeze, while Bullock and Smith (1979) specifically examined cold periods during 1977 and 1978 and found that during these times, both commercial and recreational catch figures of groupers and snappers were ”virtually non-existent” compared to usual numbers.

Certain fishes in these regions can show site fidelity following a settlement period and, unlike pelagic species, do not relocate to other locations (Mora and Ospina 2002). Finch (1917) noticed that gregarious species did not suffer as many casualties compared to species of solitary habitats during a cold spell. This difference can be attributed to the ability of the former to leave colder inland waters and move farther out to sea where temperatures are less variable. Species such as pufferfish (Family Tetraodontidae), boxfish (Family Ostraciidae), filefish (Family Monacanthidae), and triggerfish (Family

Balistidae) are noted for having high mortality rates during hypothermal anomalies due to their limited tropical habitat mobility (Gilmore et al. 1978).

When a cold spell occurs, decreased air temperatures will lower water temperatures and/or induce upwelling. One important aspect linked to the severity of the effects is how quickly the temperature drops. Sudden freezes could potentially cause more widespread damage (Storey and Gudger 1936), whereas gradual cooling may present a greater opportunity for fishes to leave shallow areas that are susceptible to larger temperature drops for deeper, more stable temperature locations. A number of factors determine how quickly the water temperature drops and the resulting minimum temperature. For example, a strong wind working against a strong tide will mix the surface water with deeper water, causing an increased rate of cooling (Storey and Gudger 1936). Moore (1976) reported that mortalities were related to the rate at which the water temperature dropped as compared to the length of time fishes were exposed to cold temperatures. Fries (1952) observed a relationship between recovery time following what he termed an induced "chill coma" and the severity of the cold.

Fish suffer both primary and secondary effects when exposed to extreme cold-water anomalies. Primary effects are almost immediate and are directly caused by the decreased water temperature. Fish exposed to decreased temperatures undergo a range of initial shock symptoms known as a "primary chill-coma," including mild distress, convulsions, cessation of respiratory actions, and paralysis (Doudoroff 1942). Doudoroff (1945) later determined that fish suffer from osmoregulative failure causing a slow death during decreased water temperature periods. More recently, Adams et al. (2012) found the cessation of cellular functions to be the cause of the mortality of common snook during excessive exposure to cold-water temperatures.

These primary effects can then cause secondary effects. Doudoroff (1942) explained how fish recovered from the initial symptoms during a cold period not extreme enough to kill the fish outright, but after a length of time the fish would again show signs of increasing distress and fell into a "secondary chill-coma." This could sometimes lead to ceased respiration, lack of response to stimuli, and eventually death. Even fish that are temporarily numbed during a cold spell can suffer casualties from a combination of winds and tides that can wash them ashore or onto exposed grass flats (Storey and

Gudger 1936). Other secondary effects include loss of food resources (Bohnsack 1983), altered larval growth (Mora and Ospina 2002), and reduced fitness and immune response, along with increased susceptibility to predation (Adams et al. 2012).

As previously stated, surface water cools faster and is subject to a greater overall change in temperature compared to deeper water, making fish near the surface more susceptible to paralysis before having a chance to respond to the danger (Storey and Gudger 1936). Reef fish inhabiting mangrove shorelines are therefore at a very high risk during cold spells due to the shallow locale of the environment. Warmer summer and fall seasons may also entice marine species to remain closer to shore during the beginning of the winter season, making the fish more vulnerable to sudden-onset temperature anomalies (Holt and Holt 1983).

Historically, cold spells have caused noticeable alterations to marine flora and fauna, including mangroves and reef fish. Recognizing the extent to which these events alter the parameters of fish populations is a crucial next step in implementing policies to mitigate added stress due to anthropogenic factors, such as habitat degradation and overfishing. However, all fishes are not equally affected by stressors, which make our understanding of their life histories and ecologies of paramount importance. One of the differentiating factors between various species of fish in terms of cold spell susceptibility is thermal tolerance. The occupied range of a species of fish results from the thermal tolerance with ontogeny and the dispersal, with the overwinter survival of juveniles as a limiting aspect (Wuenschel et al. 2012). There is, however, a difference between chronic and acute thermal tolerance limits. The former relates to long periods of time with temperatures slightly above or below determined limits, while the latter describes brief, extreme temperature alterations above or below set limits (Wuenschel et al 2012). Exceeding either chronic or acute thermal tolerance limits results in negative effects on the exposed fish. The cold spell of interest for this study is an example of an acute thermal stress.

Other factors that could be associated with variations in expected cold spell effects between fish species are diet, foraging techniques, age, mobility, and trophic level; the latter described here in terms of the number of predators that prey on the species of interest as part of the secondary effects following the event. In summation, cold spells can disrupt the routine behavioral and physiological patterns of tropical species and push them below their tolerance levels, possibly resulting in high numbers of hypothermal fish mortalities.

Species of Interest

This study focused on five species of mangrove utilizing reef fishes. They are: (1) sergeant major (*Abudefduf saxatilis* Linnaeus, 1758); (2) yellowfin mojarra (*Gerres cinereus* Walbaum, 1792); (3) schoolmaster snapper (*Lutjanus griseus* Walbaum, 1792); (4) gray snapper (*Lutjanus apodus* Linnaeus, 1758); and (5) great barracuda (*Sphyraena barracuda* Edwards, 1771). These species were selected for study based on their economic and/or ecologic importance, as well as their frequency of occurrence within the available data set. Reef fishes of South Florida support important commercial and recreational fisheries and are also indicators of environmental conditions (Bohnsack et al. 1999). Ontogenetic habitat shifters whose adult populations are regularly targeted by fisheries and whose juveniles occupy mangrove shorelines as nursery habitats include species of grouper, snapper, and barracuda (Faunce and Serafy 2008b). Gray snapper and great barracuda have previously been documented as being among the most abundant fish species in Biscayne Bay (Serafy et al. 2003; Faunce and Serafy 2008b) and are also important economically to both commercial and recreational fisheries and the dive tourism industry of the area (Serafy et al. 2007; Hammerschlag et al. 2010). Ecologically, these species span multiple trophic levels (de Sylva 1963; Starck and Schroeder 1970; Emery 1978; Randall and Vergara 1978; Harrigan et al. 1989; Hettler 1989; Schmidt 1989; Rooker et al. 2006; Hammerschlag-Peyer and Layman 2012), while from a practical perspective, all five of the selected species were comparatively easily documented in terms of accurate identification to the species level, count, and size structure measurements given the visual survey methodology.

Sergeant Major

Sergeant major belongs to the family Pomacentridae and is a tropical and subtropical species occurring primarily in the western Atlantic from Rhode Island south to Uruguay, including the northern Gulf of Mexico (Alshuth et al. 1998). They typically inhabit shallow water areas such as coral reefs and are opportunistic planktivores, feeding mostly on plankton, benthic invertebrates, and plants (Gilmore and Greenfield 2002; Feitoza et al. 2003). Sergeant majors lay adhesive eggs on hard substrates (Shaw, 1955). Following hatching, sergeant major larvae most likely spend time in the open water during a planktonic larval phase (Sale 1980) before settling on reefs as juveniles; however, recent research has instead suggested the settling of juveniles on nursery habitat sites, such as mangroves (Serafy et al. 2003). Adults are commonly found in large feeding aggregations of several hundred individuals (Allen 1991).

Yellowfin Mojarra

Yellowfin mojarra is a tropical species found in the western Atlantic around Bermuda and from Florida south to southeastern Brazil, including the Bahamas and the Gulf of Mexico, and also in the eastern Pacific from Bahia Santa Maria, Baja, California to Chimbote, Peru, including the Galapagos Islands (Gilmore and Greenfield 2002). It is thought to be the most widely distributed of the 28 species in the genus *Gerres* (Rodríguez-Romero et al. 2009). They inhabit shallow coastal waters including coral reefs, bays, bights, and mangroves (Gilmore and Greenfield 2002). It was found that yellowfin mojarra appear to be dependent on mangroves as nursery habitats in the U.S. Virgin Islands and adult populations on nearby reefs were limited by the size of mangrove stands (Halpern 2004). Their diet consists of crabs, bivalves, gastropods, polychaetes, and other benthic invertebrates (Gilmore and Greenfield 2002) as well as plant detritus (Chávez Comparán and Hammann 1989).

Schoolmaster Snapper

Schoolmaster snapper is a tropical species found in the western Atlantic Ocean usually from Florida south to Trinidad and northern Brazil. They typically inhabit shallow coastal areas including coral reefs, sand beds, and mangrove forests (Allen 1985). Schoolmaster snapper juveniles inhabit mangroves and other shallow inshore nursery areas before moving to coral reefs (Wormald et al. 2013). Nagelkerken et al. (2000) determined that schoolmaster snapper are dependent on mangroves as nursery habitats. Unlike yellowfin mojarra, mangrove stand size was not found to be a limiting factor for schoolmaster snapper adult populations on reefs (Halpern 2004). A study by MacDonald et al. (2009) indicated a limited range of behavior for schoolmaster snapper when inhabiting mangrove prop roots. Behavior consisted almost exclusively of swimming, resting, and feeding, with feeding occurring less than 3% of the time. Schoolmaster snapper spent a majority of the documented time within the dense bottom section of the mangrove prop-root system, which provides abundant shelter. This study also noted that the percentage of time spent near the bottom among the dense roots decreased as the size of the individual increased.

Faunce and Serafy (2008b) found that schoolmaster snapper occupied mangrove shorelines in greater densities during the wet summer months. The species also exhibited consistent seasonal density patterns. In a study by Rooker (1995), schoolmaster snapper were sampled from mangrove and coral reef habitats off southwestern Puerto Rico. Habitat type was found to be an influential component of sampled fish size, with all specimens less than 90 mm taken from mangrove habitats and a gradual transition from mangrove locations to reef habitats occurring over a restricted size range from 100-190 mm. A study by Verweij et al. (2007) tagged sub-adult schoolmaster snapper in the Caribbean to study their movement patterns between mangrove and seagrass bed nursery habitats and the presumed adult habitat of nearby coral reefs. The study found that these fish moved small distances during a single day, but overall movements between daytime resting sites during the duration of the study varied between 6-325 m. The tagged schoolmaster snapper also showed high fidelity to daytime shelter sites and only four out of 59 examined fishes moved between the bay and the coral reef.

Schoolmaster snapper is a large generalist carnivore that is very abundant on Caribbean and subtropical Atlantic reefs (Wormald et al. 2013). Their diet changes with fish size, with small schoolmaster snapper (≤70 mm) consuming mostly crabs and amphipods, while fish larger than 100 mm feed heavily on fish, crabs, shrimp, and stomatopods (Rooker 1995). The author suggests that these ontogenetic diet shifts are correlated with changes in jaw morphology, with a larger fish possessing a larger jaw and therefore being capable of taking in larger prey. A study by de la Morinière et al. (2003) found that schoolmaster snapper go through a major diet change at a size-class that usually occurs during the ontogenetic habitat shift from nursery habitats to coral reefs.

Sexual maturity occurred at sizes much larger than this diet change/habitat shift point. The study also found that tanaids and crabs make up large volumes of the diet of juvenile schoolmaster snapper, while a negative relationship was found for smaller crustaceans as part of the diet with increasing fish body size. The authors also reported that a positive relationship was found for decapods and prey fish in the diet with increasing fish size.

Gray Snapper

Gray snapper is a tropical and subtropical species found primarily in the western Atlantic ranging from North Carolina south to Brazil, including the Gulf of Mexico, the West Indies, Bermuda, and the Bahamas Islands (Rutherford et al. 1983; Allman and Grimes 2002). Gray snapper are found inshore in seagrass beds and near mangrove thickets during multiple life stages from larvae to early adulthood (Burton 2001). Snappers exhibit ontogenetic habitat shifts from shallow embayments (such as mangroves) to offshore spawning sites (such as coral reefs) (Faunce and Serafy 2008b). Adult gray snapper are most commonly found around coral reefs and other hard bottom substrates (Allman and Grimes 2002). Nagelkerken et al. (2000) also found the same dependence on mangroves as a nursery habitat for gray snapper.

Faunce and Serafy (2007) found that gray snapper utilize mangrove habitats as a secondary habitat after spending time in seagrass beds. The shift was shown to occur after approximately 8-10 months and when they had reached around a size of 10.5-12.0 cm total length. Gray snapper were also shown to demonstrate a positive relationship between fish size and distance from an inlet when inhabiting mangrove shorelines. A relationship also exists between water depth and fish size with adults residing in the deepest waters and the juveniles in the shallowest waters. Chester and Thayer (1990) found that juvenile gray snapper occurred most often in locations where seagrass density and species diversity were high. Young snapper have been observed in sea grass beds until approximately 80 mm standard length (SL), after which they tend to congregate near debris and channel edges (Allman and Grimes 2002).

Spawning of gray snapper occurs offshore, with eggs and larvae transported via currents into estuarine, shallow seagrass, and mangrove nursery areas (Burton 2001). Gray snapper reportedly spawn outside of the waters near Everglades National Park during the summer and fall months and enter the area as young juveniles (Rutherford et al. 1989). Data collected by Faunce and Serafy (2008b) supported the previous notion that gray snapper make seasonal movements offshore to reef areas during the summer months and inshore to mangrove areas during the winter months (Starck and Schroeder 1970). Luo et al. (2009) found that gray snapper exhibited diel movements between nearshore habitats, such as mangroves and seagrass beds, for the primary purpose of foraging. This suggests that sampling in and around mangrove prop roots would yield different results depending on the time of day the sampling was conducted. During their reproductive season, gray snapper have also been found to occasionally move from inshore habitats to offshore reef habitats. Of important note, all of the documented migrations to reefs occurred at night. Known locally as "mangrove snapper," the gray snapper is usually present within mangrove prop root systems during daylight hours (Fauncy and Serafy 2008a). This makes gray snapper an excellent candidate to study around the clear South Florida waters.

Gray snapper are one of the top predators in seagrass beds and on coral reefs and are an important ecological part in the communities of marine ecosystems (Denit and Sponaugle 2004). Gray snapper are opportunistic carnivores feeding almost entirely on crustaceans and fish, such as mysids, amphipods, carideans, penaeids, and fishes (Hettler 1989); however, their diet does change with size and habitat (Rutherford et al. 1983). This was observed along the west coast of Florida, in which the diets of all size classes of gray snapper examined consisted of 59% penaeid shrimp, and 24% carideans and fish (Hettler 1989). Conversely, this same study found that while the snappers inhabited mangroves, no penaeid shrimp were in the specimens and fish made up 45% of the stomach contents.

Florida gray snapper juveniles have a mean growth rate of 0.62-0.88 mm/d (Denit and Sponaugle 2004). Water temperature differences have been implicated as accounting for at least half of the variability in juvenile gray snapper growth rates (Denit and Sponaugle 2004). Rutherford et al. (1983) provided another look into the effects of temperature on gray snapper growth rates by examining them from the mainland shoreline of Everglades National Park in South Florida during the years 1975-1977. The 1975 year-class grew faster in comparison to the other two year-classes. Unusually cold weather occurred in January 1976 and January 1977, which could have altered regular growth patterns. Temperatures were recorded below 4.4 °C for three days in 1976 and for five days in 1977, well below the long-term average temperatures for the month. Allman and Grimes (2002) found juvenile gray snapper growth rates to vary between 0.6 mm/d and 1.0 mm/d, with the difference explained by possible temperature variations. Previous studies have found juvenile gray snapper to have an acute thermal tolerance minimum limit of roughly 7 °C, whereas the chronic minimum limit was 17 °C for a time period of more than 210 cumulative degree-days. It was also noted that juvenile growth was highest at roughly 33 °C, growth ceased below 17 °C, and feeding ceased at approximately 11 °C (Wuenschel et al. 2004; Wuenschel et al. 2012).

Gray snapper are commonly harvested in recreational and commercial fisheries in the U.S. South Atlantic and Cuba and have been the focus of numerous studies due to their abundance and economic importance (see Faunce and Serafy 2008a). Higman (1966) determined that the most productive recreational fishing season in Everglades National Park for gray snapper was late summer, which usually coincided with the highest water temperatures and lowering levels of salinity. The oxygen deficient waters in the park area caused by local heavy runoff carrying decomposed organic material from mangrove swamps may drive the gray snapper out of the mangrove habitats into more fishing accessible areas. The South Atlantic Fishery Management Council (SAFMC) is responsible for the management of gray snapper in Florida waters (Burton 2001).

Great Barracuda

Great barracuda is the most widespread and largest of the barracuda species in the family Sphyraenidae, ranging from Massachusetts south to southeastern Brazil in Atlantic waters and reaching 2 m in length and up to 45 kg in weight (D'Alessandro et al. 2011). Great barracuda have shown movement patterns that are generally characteristic of both coastal reef fishes and pelagic billfishes (Daly-Engel et al. 2012). Great barracuda exhibit consistent seasonal density patterns, with greater densities being observed along mangrove shorelines during the wet seasons and a random distribution pattern during the dry seasons (Faunce and Serafy 2008b). The less aggregated distribution of great barracuda during the dry seasons may be linked to environmental pressures and stronger territoriality among the space-limited, social-hierarchical forming species (de Sylva 1963). Great barracuda larvae are found to occur from June through November in the Straits of Florida during times of maximum temperatures and are almost entirely constrained to the upper 50 m of the water column (D'Alessandro et al. 2011). Schmidt (1989) reported the occurrence of great barracuda in water at temperatures as low as 14.3 °C. Great barracuda were found to be dependent on mangroves as a nursery habitat (Nagelkerken et al. 2000).

The diet of larval great barracuda less than 10 mm SL consists mostly of copepods and copepod nauplii, while fish larvae begin to appear in the gut contents of great barracuda at 8 mm SL (D'Alessandro et al. 2011). Previous studies have reported on the solidarity of adult great barracuda (Gudger 1918; de Sylva 1963), however Paterson (1998) reported that the species commonly aggregates in pairs and groups. The author attributes this behavior to possible anti-predatory functioning, pre-spawning activities, foraging advantage, and mutual attraction. Great barracuda are a very popular recreationally fished species in the United States and are an important part of their ecosystem due to their range, diversity of occupied habitats, and role as apex predators (D'Alessandro et al. 2011).

Present Study

This study investigated the effects of a cold spell on mangrove utilizing reef fishes in southeastern Florida. Due to the examination of quantitative pre- and post-event data, the results offer the opportunity to test the hypothesis that multiple indices, including presence, abundance, and size structure will be directly affected for the five species of interest. More specifically, it is believed that the size structure will shift towards larger, older fish in the survey seasons immediately following the cold spell due to the primary and secondary effects reducing the numbers of smaller fish in a greater proportion compared to larger fish. Conversely, it is believed that the values for presence and abundance will decrease under the same circumstances due to the same primary and secondary effects reducing the number of fish in the area. These hypotheses were tested by examining variations in seasonal cohort data, statistically comparing three time periods for variations in presence, abundance, and size structure, and inspecting descriptive variables. Due to the predicted effects on the five species and the previously determined propagation of year-class strength for some species during a time period of one or two years from the mangroves to the adjacent reefs (Jones et al. 2010), it is predicted that the examined parameters will require the recruitment of a new year-class to fully return to pre-stress levels.

Data specific to important reef and mangrove species could potentially be incorporated into stock assessments and ecosystem-based management efforts. This would contribute to fishery policy decisions, specifically in setting catch limits for commercial and recreational fisheries. Another important aspect of this study is that it may have implications for how the scientific community understands the connections within the reef and mangrove ecosystems and how temperature drops may affect the mangrove utilizing reef fishes. It is unknown how episodic cold snaps alter or diminish the importance of mangroves as nursery habitats for juvenile reef fishes. Also, the examination of recorded water temperatures during, and immediately following, the cold spell may provide information on any direct or lagged effects based on the duration and severity of the event at both mainland and leeward key shorelines in the study area.

Materials and Methods

A single data set was analyzed for the majority of this study. The subset used of the mangrove fish survey data set, described in full below, encompasses surveys from August 1998 to October 2014 in the southern Biscayne Bay region of BNP. Water temperature data from January 2005 to May 2014 collected by two YSI stations located along both mainland and leeward key mangrove shorelines of BNP were also examined.

Study Location

The coastal ecosystem of South Florida includes a mixture of freshwater and saltwater regions, such as marshes, lagoons, estuaries, seagrass beds, mangroves, and coral reefs (Ault et al. 1998). The current mangrove fish survey data set has been continuously collected since 1998 within the BNP region. BNP includes most of Biscayne Bay and extends roughly 10 km offshore, encompassing the northern section of the Florida Keys reef tract that can be found roughly 6 km offshore (Burns 1985; Jones et al. 2010). Biscayne Bay is one of three shallow subtropical bays located in or around BNP (see Figures 1 and 2) (Serafy et al. 2007). Only three species of mangrove are found in this area: red mangrove (*Rhizophora mangle*); black mangrove (*Avicennia germinans*); and white mangrove (*Laguncularia racemosa*) (Odum et al. 1982).

Biscayne Bay can be divided into different regions due to the effects of human alterations. The northern portion of the Bay is home to the city of Miami and is surrounded mostly by concrete seawalls and limestone boulders (Teas et al. 1976). As such, this area contains very few mangrove stands. Northern Biscayne Bay has also undergone bottom dredging and filling, resulting in a decreased natural filtering capacity and sediment instability (Browder et al. 2005). Now, the benthic communities of this area are mostly seagrasses with some calcareous green algae, which in part is due to the elimination of direct sewage discharge into the Bay (Browder et al. 2005). However, the southern half of Biscayne Bay has experienced fewer human alterations and is predominately bordered by red mangroves with marine seagrasses, such as turtle grass (*Thalassia testudinum*), comprising the majority of the substrate vegetation (Serafy et al. 2007). The benthic communities here also consist of hard and soft corals, as well as macroalgae and sponges (Browder et al. 2005). This southernmost region exemplifies the natural habitat of mangroves and the resulting connection with certain reef fishes. Southern Biscayne Bay is mostly very shallow, averaging <3 m in depth, with small passes opening to the ocean on the eastern mangrove-lined shorelines leading to limited tidal flushing (Machemer et al. 2012).

The shorelines of southern Biscayne Bay can be separated into two main locations: the western, mainland shorelines, and the eastern, leeward key shorelines. Both areas are predominantly covered by mangrove habitat, although mangrove coverage is higher at the leeward key locations compared to other areas within BNP (Machemer et al. 2012). Comparatively, the leeward key mangroves are in deeper water and are less susceptible to environmental conditions due to the buffering capacity of the adjacent ocean water, specifically that of the Gulf Stream. Water depth for all surveys conducted at the locations of interest from 1998-2014 ranged from 9-157 cm at mainland locations and from 17-175 cm at leeward key locations. The depth of mainland surveys averaged 59 cm, while leeward key surveys averaged 70 cm. Also, the shallower waters of the

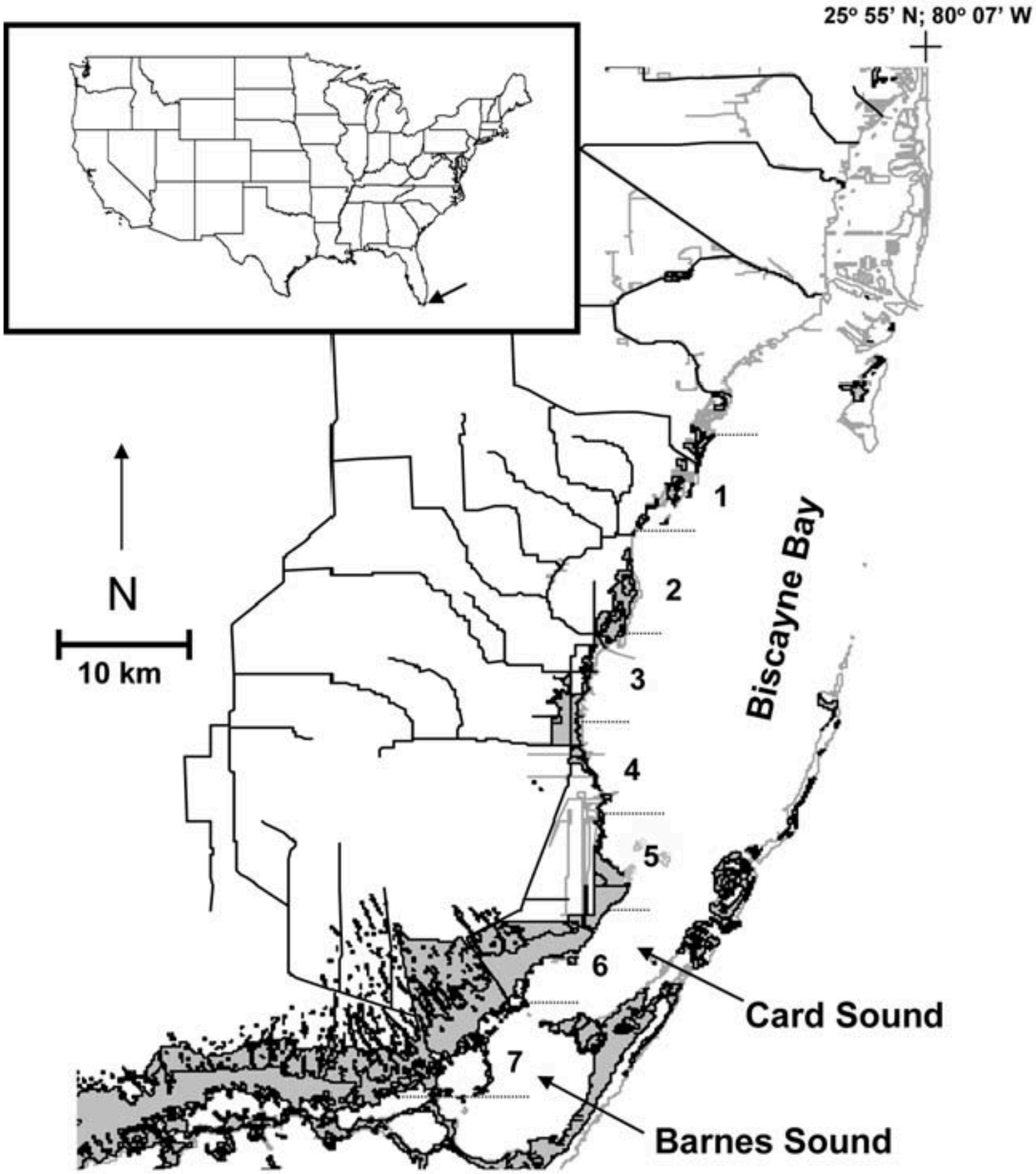


Figure 1. Map showing the location of Biscayne Bay on the southeastern coast of Florida. The shaded area signifies mangrove-dominated wetland areas (map from Serafy et al. 2007).

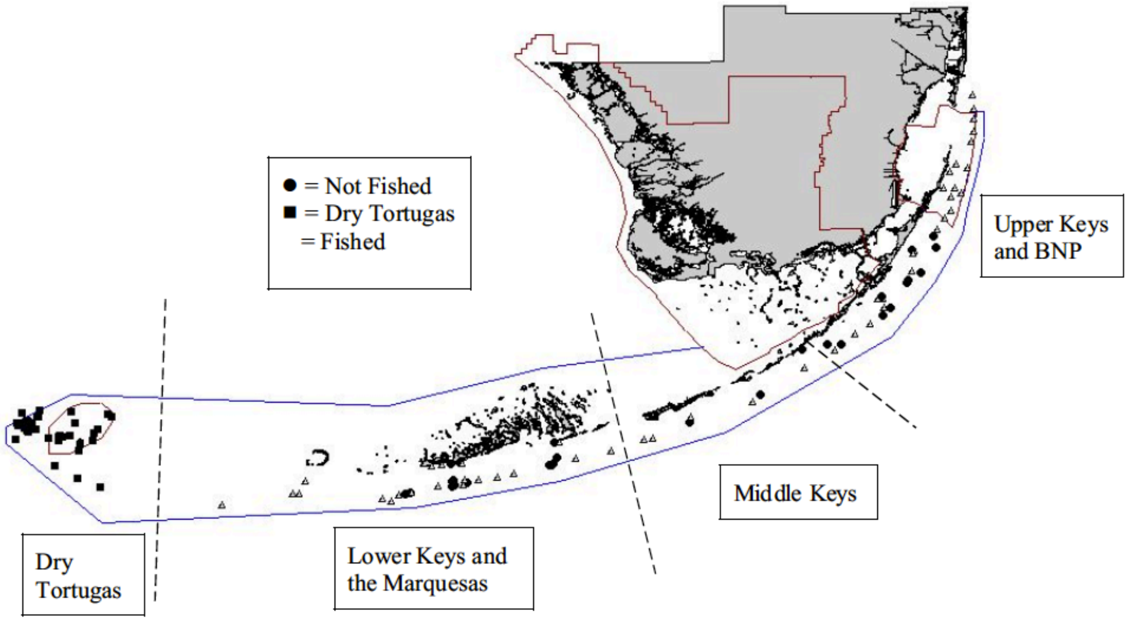


Figure 2. Map showing the location of BNP in relation to other sites in the Florida Keys (map from Bohnsack et al. 1999).

mainland locations may not be as capable of supporting high of levels of food availability, whereas the deeper waters of the leeward key locations may provide more space for foraging and shelter (Machemer et al. 2012).

The western section of the Bay receives fresh water from interspersed natural creeks, artificial channels, and canals (Serafy et al. 2007). This area experiences rapid salinity fluctuations due to the water control network that has nearly eliminated the natural flow of fresh water from Lake Okeechobee to the Everglades and coastal systems, periodically flushing backed-up freshwater from locks through the canals, lowering the salinity in the area by as much as 27 ppt in as little as a 60 minute period (Serafy et al. 1997; Harrington and Serafy 1998). This pulsed discharge of fresh water also degrades the ecotones and estuarine habitats of the Bay due to bottom scouring and alterations to the natural salinity gradients from the Everglades (Browder et al. 2005). These fluctuations lead to adverse effects on inhabiting fishes, depending on their osmoregulatory ability and behavioral response (Serafy et al. 1997; Machemer et al. 2012), and can affect growth, survival, and reproduction (Browder et al. 2005). Overall, the alterations to the natural flow of freshwater into Biscayne Bay have diminished the ability of the mangrove habitats to support a healthy and diverse fish community (Browder et al. 2005). The Bay's wet season experiences high salinity variation, while its dry season has lower salinity variation (Serafy et al. 2003).

There are only a few protected areas in the entire region, with fishing regulations complying with the state of Florida statutes (Jones et al. 2010). Concerns of overfishing and destruction of habitat have resulted in the establishment of the Florida Keys National Marine Sanctuary (FKNMS) in 1990. Although the protected area encompasses 9500 km^2, commercial and recreational fishing are mostly permitted within the boundaries aside from very small sections of reef systems set aside for scientific research (Serafy et al. 2003).

Jones et al. (2010) suggested the mangroves of BNP were nursery habitats for the fishes that inhabit the nearby reefs. The authors further suggested inter-habitat connectivity for four of the ten examined species: sergeant major, schoolmaster snapper, gray snapper, and great barracuda, supporting the idea that fishes can undergo ontogenetic migrations between the juvenile and adult life stages from the mangroves of

Biscayne Bay to the offshore reefs. The study also determined that some species experience a propagation of year-class strength during a time period of one or two years from the mangroves to the reefs, also supporting the idea of the mangrove nursery habitat.

YSI Temperature Data

Water temperature records from two YSI stations in BNP, one located in the mainland mangrove locations and the other in the leeward key mangrove locations, were examined to determine the immediate and lagged effects of the January 2010 cold spell and any differences between the two sites. The YSI stations continuously recorded water temperatures in 15-minute intervals; examined here is the record from January 2005 to May 2014. For each of the stations, twelve means were created using the data points for each individual month (for example, January 2005, January 2006, January 2007, and so on, were averaged together to create one monthly mean) and standard deviations were then calculated (Tables 2 and 3). Each monthly mean was then compared to the individual 15-minute intervals of those respective months. The intervals that fell three standard deviations below the computed monthly mean for longer than a period of a few hours were determined as statistically significant decreases in water temperature. Since the literature documented a cold spell in the South Florida region from January 2-14, 2010 (Adams et al. 2012), this time period was specifically examined. This data was also used to assess other periods in which the water temperature of the area was significantly lower than average. The two stations are designated as BNP40 and BNP10 and are geographically located at positions 25.505 deg N by 80.336 deg W and 25.397 deg N by 80.234 deg W, respectively.

Mangrove Fish Survey Data

Data for the mangrove fish survey data set were obtained as part of an ongoing, fishery-independent visual survey study. The sampling method documenting fish assemblages in southern Biscayne Bay along mangrove prop root locations has previously been described in full (see Serafy et al. 2003). A modified version of the Rooker and Dennis (1991) visual ‘belt transect’ census method was employed, consisting

Table 2. Means and standard deviations for monthly water temperatures recorded by YSI station BNP40 along the mainland mangrove locations in Biscayne Bay. Also listed is the value three standard deviations below the mean, which was used to determine significant decreases in water temperature at the study location.

Month	**Mean**	**SD**	**3 SD Below Mean**
January	21.08	3.23	11.40
February	22.44	3.09	13.17
March	23.37	2.77	15.07
April	26.04	2.33	19.04
May	28.00	2.06	21.81
June	30.14	2.13	23.75
July	30.83	2.01	24.79
August	31.08	2.01	25.04
September	29.85	1.71	24.73
October	27.33	2.37	20.23
November	23.90	2.39	16.75
December	22.43	2.94	13.63

Table 3. Means and standard deviations for monthly water temperatures recorded by YSI station BNP10 along the leeward key mangrove location in Biscayne Bay. Also listed is the value three standard deviations below the mean, which was used to determine significant decreases in water temperature at the study location.

Month	**Mean**	**SD**	**3 SD Below Mean**
January	20.57	3.05	11.41
February	21.64	2.71	13.51
March	22.81	2.38	15.68
April	25.36	1.82	19.90
May	27.52	1.56	22.84
June	29.56	1.50	25.05
July	30.48	1.21	26.84
August	30.71	1.32	26.75
September	29.69	1.10	26.39
October	27.19	2.18	20.66
November	23.30	2.21	16.68
December	21.85	2.91	13.11

of an observer snorkeling 30 m long transects running parallel to the mangrove shorelines. During each survey, the observer documented the species of each fish seen and if in a grouping, the number of fish per group. The size structure of the fish or grouping would also be recorded. Size structure included estimated values for minimum, mean, and maximum total length in the grouping of like fish in inches. Belt transects were 2 m wide starting at the edge of the mangrove prop roots and measured landward amid the prop roots themselves, resulting in each transect being 60 m^2 in size. Surveys were conducted between the hours of 09:00 and 17:00 to avoid low-light level issues. A modified version of the training procedure described by Bell et al. (1985) was employed to address potential issues with observer bias in fish length estimation. A similar protocol was also developed to maximize fish counting accuracy and precision. Surveys were assigned a season depending on when they were conducted, with dry seasons being primarily January-March and wet seasons being primarily July-September (see Figure 3). A few surveys in the first years of data collection were also conducted during the spring season in May and the fall season in late November.

Original transects were chosen at random every season following the method explained by Diaz (2001). However, recent selections of exact transect locations were chosen based on previous results of fish density and diversity, as well as variations in salinity. Visual assessments were not conducted when visibility conditions were poor, danger due to large sharks or reptiles, or when access to the survey location was limited due to very shallow water depths. Various sources were used to identify fish to the species level, but this was not always possible given the problems of visually identifying fish with uniform coloration and similar morphology, especially when a part of large, mobile, mixed-species schools (Rooker and Dennis 1991). All problematic taxa were therefore identified to the genus or family level (Rooker and Dennis 1991) and all small, silvery, fork-tailed fishes that inhabit the water column and are often found in large schools were placed into a single group (Humann 1994). During each survey, water temperature, salinity, and dissolved oxygen measurements were obtained using a Hydrolab® multi-probe instrument. Depth recordings were also taken along each transect using a 2 m long polyvinyl chloride pole with markings every 2 cm.

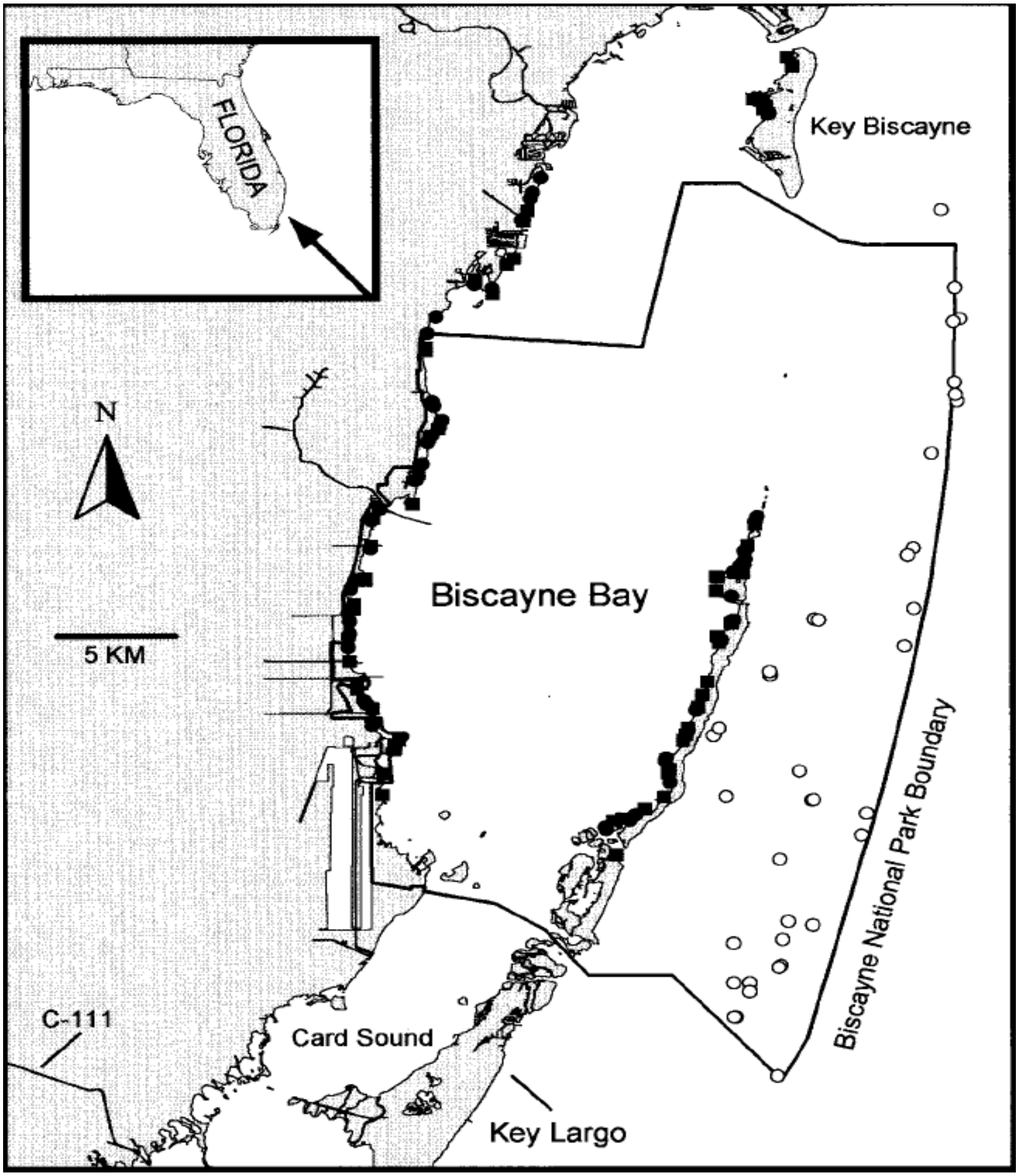


Figure 3. Map showing a sample of visual fish surveys between 1998 and 2000 in Biscayne Bay (map from Serafy et al. 2003). Shaded circles and squares represent dry and wet season surveys, respectively, along the mainland and leeward key shorelines.

Data Reconstruction

For the purposes of this study, a subset of the original data set was used and several methods were employed to create a more cohesive summary. Data have previously been grouped into four seasons, but due to the limited number of surveys during the fall and spring seasons, only dry and wet seasons (dry: primarily January-March; wet: primarily July-September) were examined. Mangrove surveys were conducted along seven different locations; however, due to a lack of consistent sampling at several of these sites, this study only examined surveys from the mainland and leeward side of the northern Florida Keys locations. These two locations were also selected due to previous works finding significant differences in reef fish abundance (Serafy et al. 2003; Faunce and Serafy 2007, 2008b; Jones et al. 2010).

Original data for minimum, average, and maximum total length of fish were recorded for the assemblages in inches, but the values were converted to centimeters by multiplying each value by 2.54. If there was one fish of a particular species during a survey, the size structure values for the minimum, average, and maximum total length should be the same value. In the case of a survey containing two fish, the average total length should be the average between the minimum- and maximum-recorded lengths. In instances where survey records did not create a feasible size structure based on the number of fish seen (e.g., $n = 2$, min = 1.0 cm, ave = 1.0 cm, max = 4.0 cm), new values were computed in a consistent manner dependent on the number of fish and the original three values (e.g., $n = 2$, min = 1.0 cm, ave = 2.5 cm, max = 4.0 cm).

The corrected minimum, average, and maximum total length values and corresponding abundance values for each survey were then reconstructed to fit a standard normal distribution following the methods of Meester et al. (1999). This was done to more accurately assess the size distribution of a species during a season given the varying abundances that corresponded to the length values. Standard normal distributions were applied to the recorded abundance estimates of each survey and then those estimates were correlated to the minimum, average, and maximum total length values, creating five new samples for each original survey. An example of this process can be seen in Table 4. The abundance values for each species in a particular sample were assumed to be normally distributed so that they could be split into five sub-abundances if the abundance was

Table 4. Example of the mangrove survey data reconstruction following the methods of Meester et al. (1999) to more accurately assess the size distributions of the target species.

Before Reconstruction					**After Reconstruction**		
Survey #	**Abundance**	**Min.**	**Ave.**	**Max.**	**Survey #**	**Abundance**	**Length**
1	22	7.62	12.70	15.24	1	1.00	7.62
					1	4.50	10.16
					1	11.00	12.70
					1	4.48	13.97
					1	1.00	15.24
2	5	5.08	7.62	10.16	2	1.00	5.08
					2	0.25	6.35
					2	2.50	7.62
					2	0.25	8.89
					2	1.00	10.16

greater than three fish. For a more complete understanding of the methods used to calculate a species' average length in a given survey, refer to Meester et al. (1999).

Descriptive Analyses

Following the selection of the data subset and the reconstruction of the survey data, descriptive analyses were computed for comparative purposes. Variables were examined by location, season, and species and included the number of surveys conducted (Table 5), the overall presence (Table 6), the overall abundance (Table 7), and the size range, including average total length (Table 8).

Length-frequency distributions of each sample were also calculated for all species at both mainland and leeward key mangrove locations. This was done to assign abundance values to various age-classes to better assess the inhabiting populations for both shorelines and to help determine the role of the habitat in terms of the life histories and ontogenetic shifts for the target species. For each species, von Bertalanffy growth functions were used to assign lengths at yearly intervals (see Jones et al. 2010). The von Bertalanffy growth equation is:

$$L_t = L_a\,(1 - e^{-K(t - t_0)})$$

where L_t is the length at time t, L_a is the theoretical asymptotic total length, K is the growth rate parameter, and t_0 is the age of the fish at zero length if the growth of the fish followed the equation. The reconstructed length-frequency data were then differentiated into one of five age classes (i.e. Age 0 to 4^+) based on these lengths for each survey. Individuals with an average total length less than the predicted size at Age 1 were designated as juveniles. Combined percentages of length-frequency distributions over all seasons were then computed and plotted for each species at both locations. Corresponding mean abundances (assessed here as the number of fish per 60 m^2 – the area of each mangrove survey) for each age-class by habitat location were also determined. Details of the von Bertalanffy growth functions used are given in Table 9 and lengths defining each age class are provided in Table 10.

Due to their comparatively high presence and abundance values from the data subset, gray snapper were selected out of the five target species for further examination. Seasonal length-frequency distributions from 2009-2011 were plotted and the peaks were

Table 5. Number of visual surveys by year and season used in the present analysis. Mangrove transects in BNP are designated by habitat location: mainland (ML, n = 2634), leeward key (LK, n = 735).

Year	**Season**	**ML**	**LK**
1998	Wet	15	17
1999	Dry	17	15
	Wet	15	18
2000	Dry	28	38
	Wet	31	30
2001	Dry	27	51
	Wet	22	51
2002	Dry	29	20
	Wet	30	20
2003	Dry	28	19
	Wet	38	20
2004	Dry	38	20
	Wet	38	20
2005	Dry	96	23
	Wet	108	20
2006	Dry	97	20
	Wet	117	19
2007	Dry	119	20
	Wet	121	21
2008	Dry	120	21
	Wet	121	21
2009	Dry	118	21
	Wet	121	21
2010	Dry	121	21
	Wet	121	21
2011	Dry	121	21
	Wet	119	23
2012	Dry	121	21
	Wet	129	13
2013	Dry	121	21
	Wet	78	14
2014	Dry	111	21
	Wet	98	13

Table 6. Overall presence data derived from the subset of the mangrove fish survey data set for each of the five target species. Data are organized by mangrove survey location (ML and LK). Also shown is the overall breakdown for the entire data subset.

Species	**ML**		**LK**		**Total**	
	#	**%**	**#**	**%**	**#**	**%**
Mangrove Data	2634	78.2	735	21.8	3369	100.0
Sergeant major	100	3.8	192	26.1	292	8.7
Yellowfin mojarra	625	23.7	291	39.6	916	27.2
Schoolmaster snapper	137	5.2	191	26.0	328	9.7
Gray snapper	805	30.6	550	74.8	1355	40.2
Great barracuda	1008	38.3	401	54.6	1409	41.8

Table 7. Overall abundance totals for each of the five target species for the subset of the mangrove fish survey data set. Data are organized by mangrove survey location (ML and LK). Also shown are mean abundance values (number of fish per 60 m^2 – the area of each mangrove transect during surveys).

Species	ML	#/Survey	LK	#/Survey	Total	#/Survey
Sergeant major	286	0.109	511	0.695	797	0.237
Yellowfin mojarra	3977	1.510	1333	1.813	5310	1.576
Schoolmaster snapper	478	0.181	841	1.144	1319	0.391
Gray snapper	12452	4.713	15296	20.626	27748	8.184
Great barracuda	2302	0.874	1158	1.575	3460	1.027

Table 8. Size range, reported as minimum, mean, and maximum total length (TL, cm) for each of the five species of interest observed during visual surveys of mangrove transects in BNP during 1998-2014 designated by habitat location: mainland (ML, n = 2634) and leeward key (LK, n = 735).

Species	**Location**	**Min.**	**Mean**	**Max.**
Sergeant major	ML	1.3	4.6	12.7
	LK	1.3	6.3	20.3
Yellowfin mojarra	ML	2.5	12.0	40.6
	LK	3.5	14.2	40.6
Schoolmaster snapper	ML	2.5	12.2	35.6
	LK	2.5	12.1	38.1
Gray snapper	ML	1.3	20.6	70.0
	LK	1.3	18.3	48.9
Great barracuda	ML	1.9	26.5	116.8
	LK	2.5	22.9	99.1

Table 9. von Bertalanffy growth function parameters for the five species of interest. L_a = theoretical asymptotic total length (cm), K = annual growth coefficient, t_0 = theoretical age when length = 0 (for original sources, see Jones et al. 2010).

Species	**L_a (cm)**	**K**	**t_0**
Sergeant major	24.1	0.200	-2.300
Yellowfin mojarra	34.1	0.650	0.000
Schoolmaster snapper	54.7	0.223	-0.497
Gray snapper	67.0	0.168	-0.695
Great barracuda	149.9	0.216	1.690

Table 10. Predicted lengths (TL, cm) for the five species of interest based on von Bertalanffy growth functions (Jones et al. 2010).

Species	**Age 1**	**Age 2**	**Age 3**	**Age 4$^+$**
Sergeant major	11.6	13.9	15.8	17.3
Yellowfin mojarra	16.3	24.8	29.2	31.5
Schoolmaster snapper	15.5	23.4	29.6	34.6
Gray snapper	16.6	24.4	31.0	36.6
Great barracuda	66.0	82.3	95.5	106.0

labeled with numbers to depict the existence of varying cohorts and the seasonal movement of said cohorts to better examine the resulting effects from the January 2010 cold spell. Data availability for the presence and abundance of the four remaining target species was minimal during this time period, which hindered the inspection of cohort movement on a season-by-season basis.

Statistical Analyses

The presence for each of the five target species by location and season were compared using a Chi-square analysis across three time spans: the years prior to the cold spell (1998-2009; hereafter referred to as time period 1); the two years immediately following the cold spell (2010-2011; hereafter referred to as time period 2); the remaining years following the cold spell (2012-2014; hereafter referred to as time period 3). The analysis examined the number of surveys with the fish of interest present compared to the number of surveys with the fish of interest absent. Due to the comparative analysis of three time periods for each target species and location, a Bonferroni-adjusted significance level of 0.0167 was calculated to account for the increased possibility of type-I error.

The relative abundance for each target species by location and season was compared over the same three time spans. A non-parametric Kruskal-Wallis test was conducted to determine if a significant difference existed within the relative abundance data at the $p < 0.05$ level. For those instances where a significant difference was determined, a post-hoc Mann-Whitney U test was employed to examine where the significant differences existed between the three time periods. Statistical significance was determined at the Bonferroni-adjusted significance level of $p < 0.0167$ (i.e., 0.05/3) due to the three iterations of comparatively testing the three time periods.

The length-frequency distributions of the five target species were examined by location across the varying time periods (1, 2, and 3) by conducting non-parametric two-sample Kolmogorov-Smirnov tests. This compared the cumulative length-frequency distributions with the null hypothesis that both groups compared were sampled from populations with identical distributions. Again, statistical significance was determined at the Bonferroni-adjusted significance level of $p < 0.0167$. Due to the occurrence of zero values when the distributions were separated into the three time periods by season, and since the Kolmogorov-Smirnov test relies on the comparison of existent distributions, the values were grouped by time period alone. Plots of the length-frequency distributions by time period were constructed for each target species by location to assess the resulting *D*-statistic from the two-sample K-S tests.

Results

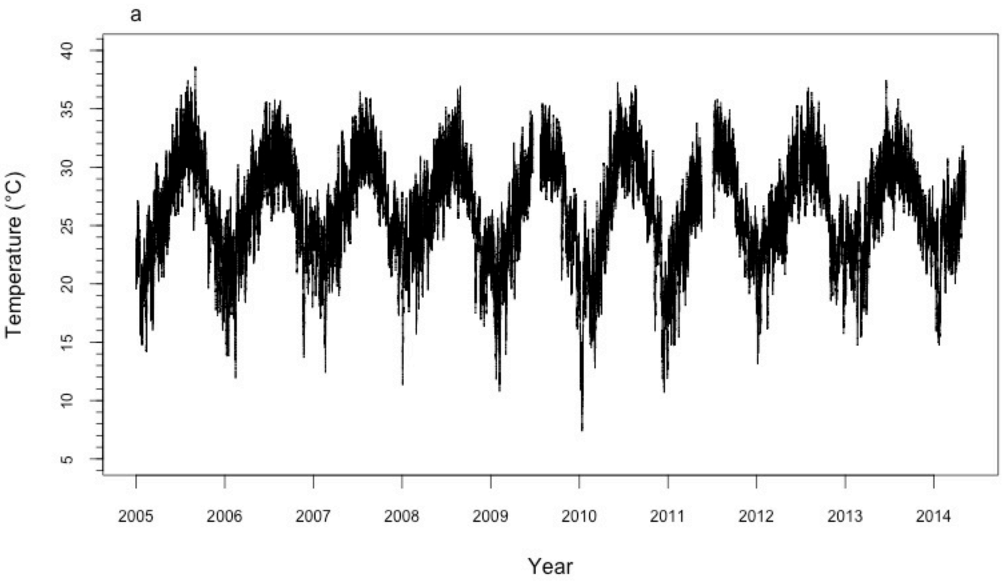
Reduced Water Temperature Periods

The individual water temperatures recorded by the two YSI station were plotted to depict variations in seasonal temperatures and deviations from winter monthly means (Figures 4a-f). Several periods of time during the winter months (November-March) experienced consistent water temperatures below the three standard deviation threshold for longer than a few hours. Chronologically, the significant cold periods for YSI station BNP40 located at the mainland mangrove shoreline were November 21-24, 2006, February 5-6, 2009, January 6-8, 2010, January 10-13, 2010, March 4-6, 2010, December 14-16, 2010, and December 28-29, 2010. The significant cold periods detected by YSI station BNP10 located at the leeward key mangrove shoreline were February 5-6, 2009, January 6-7, 2010, January 9-14, 2010, March 4-7, 2010, December 14-16, 2010, and December 28-29, 2010. When considering the records from both stations, the cold period of interest, from January 2-14, 2010, resulted in the majority of 15-minute intervals from January 6-14, 2010 falling below three standard deviations of the January average water temperature. It appears that the water temperature decrease lagged a few days behind the air temperature decrease, and recovered to temperatures within three standard deviations around the same time the documented cold spell ended.

General

The average total length for each species, combined over all examined seasons and both mangrove locations, calculated from the reconstructed averages are 5.7 cm for sergeant major, 12.6 cm for yellowfin mojarra, 12.2 cm for schoolmaster snapper, 19.3 cm for gray snapper, and 25.2 cm for great barracuda.

For each of the five species, 33 seasons were examined (17 wet seasons and 16 dry seasons). Only great barracuda were observed during every season at both locations.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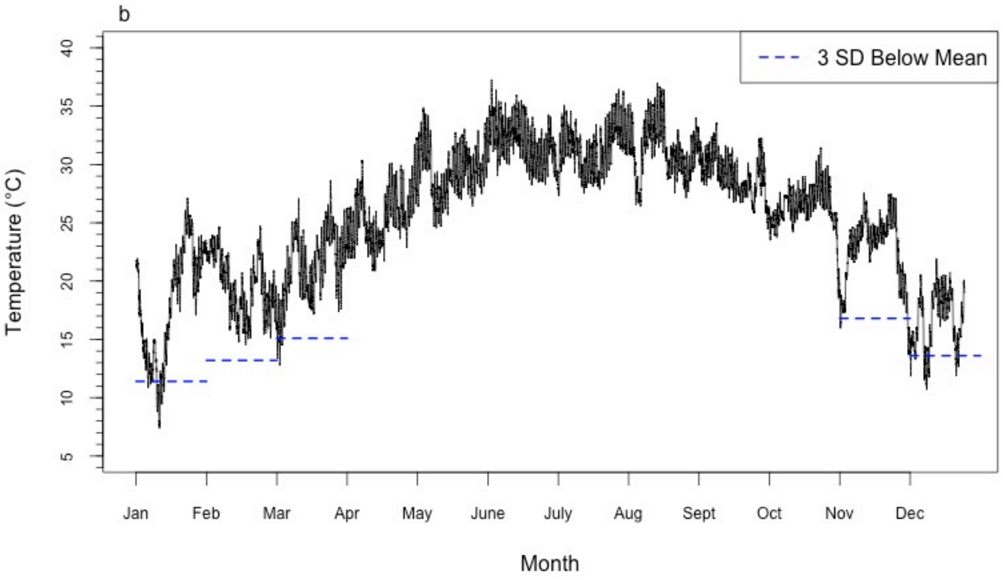
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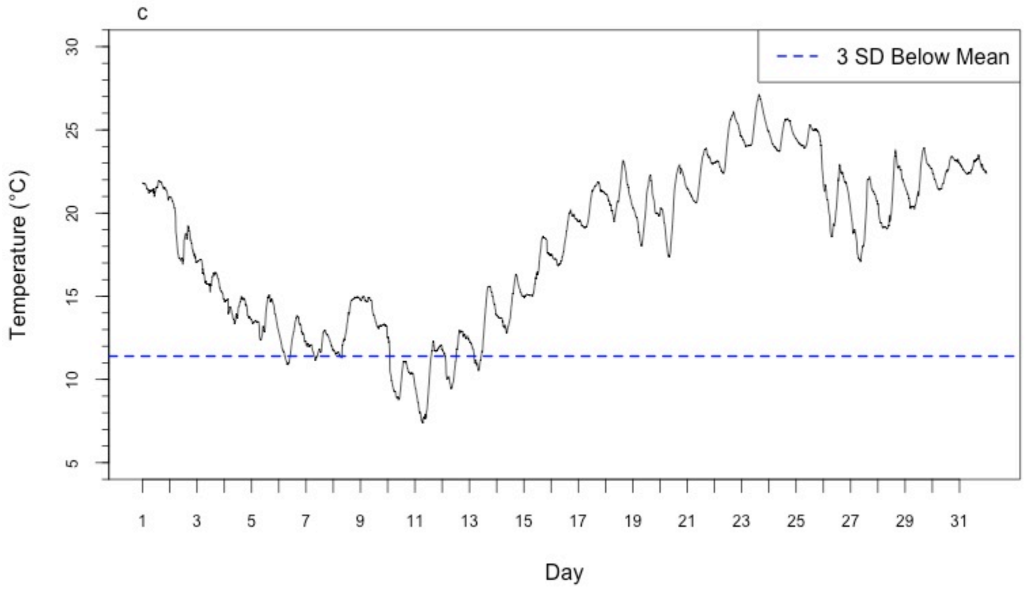
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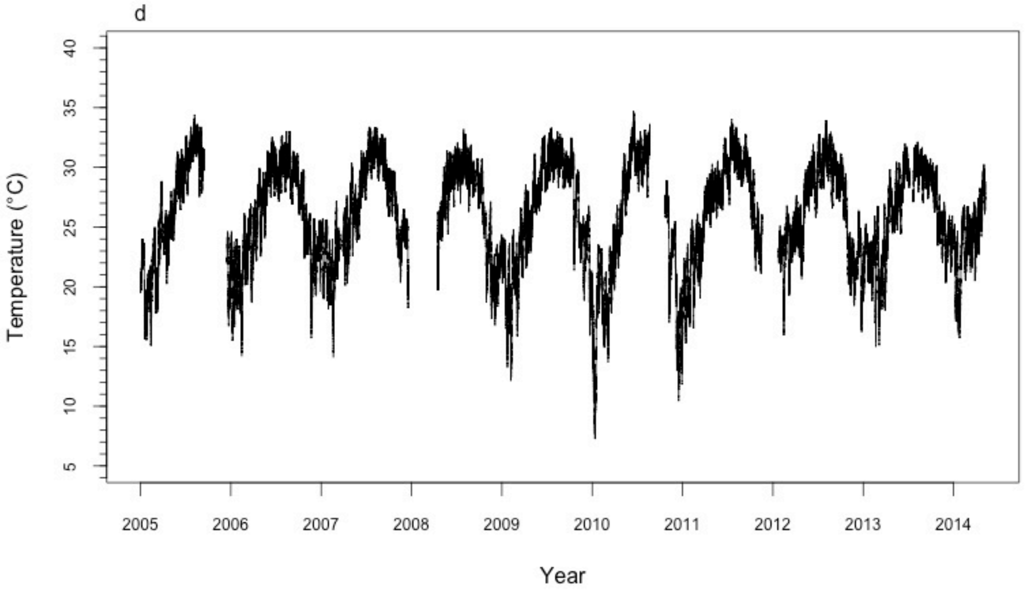
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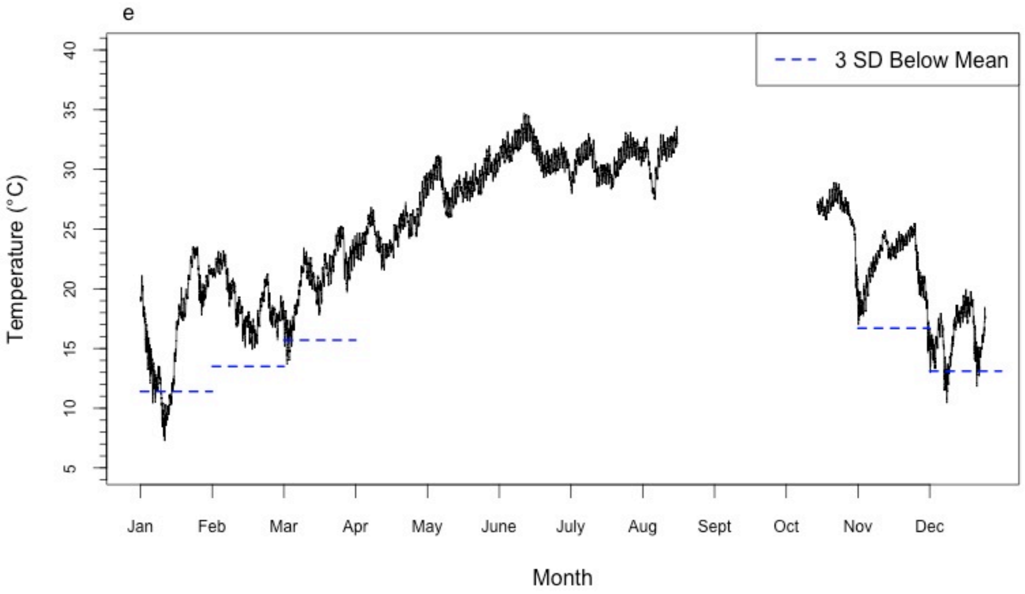
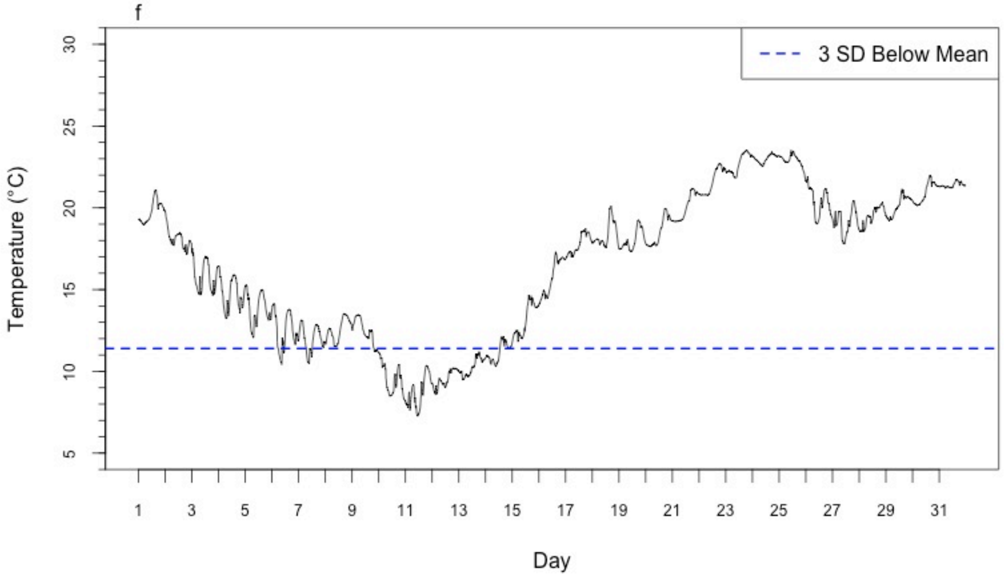




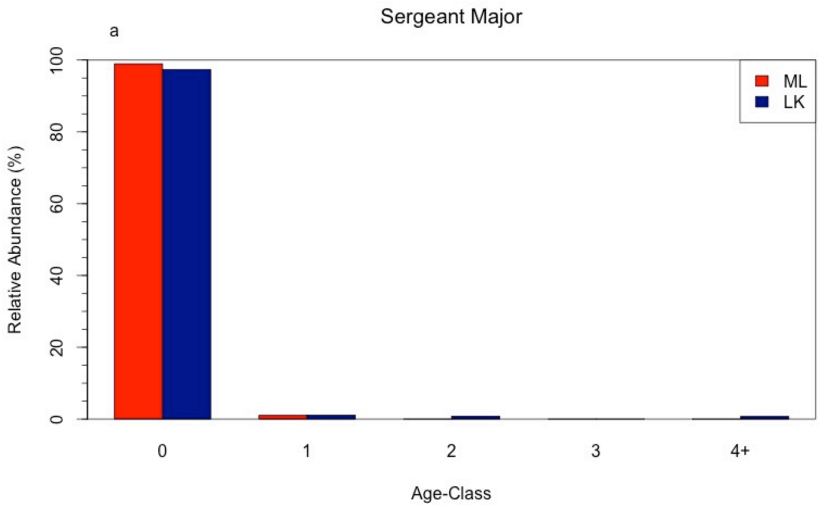
Figures 4a-f. Plots for the YSI water temperature data from station BNP40 (a-c) located along the mainland mangroves and station BNP10 (d-f) located along the leeward key mangroves in Biscayne Bay. Data were logged in 15-minute intervals. The cold spell of interest (January 2010) is noticeable for the overall observation period (a and d, January 2005 – May 2014) and can be seen during the 2010 data period (b and e) and the January 2010 data period (c and f). Three standard deviations below the monthly means are depicted for the winter seasons (b and e, January-March and November-December) and for the month of January (c and f).

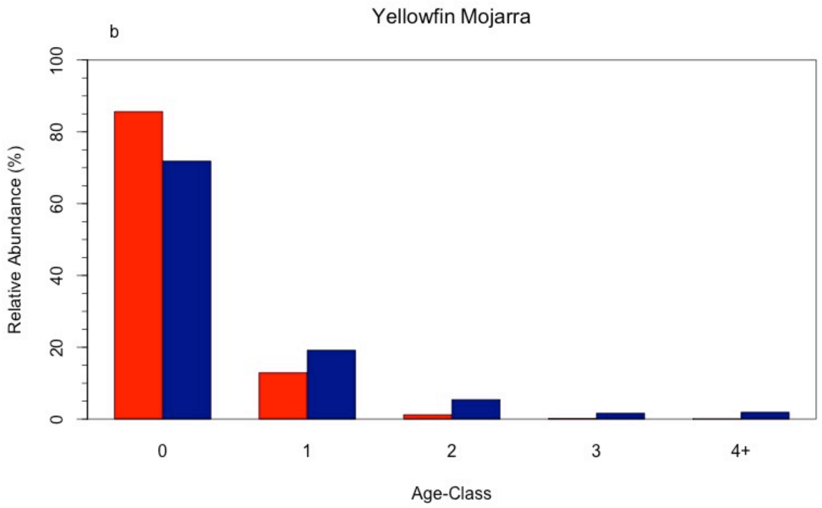
Gray snapper were present in all but one survey season, while yellowfin mojarra were present in all but three survey seasons. There were multiple survey seasons throughout the dataset that did not have a single fish of interest present, resulting in 0% presence, zero fish per survey, and no data for size structure. Schoolmaster snapper and sergeant major were completely absent from 12 and 13 seasons, respectively.

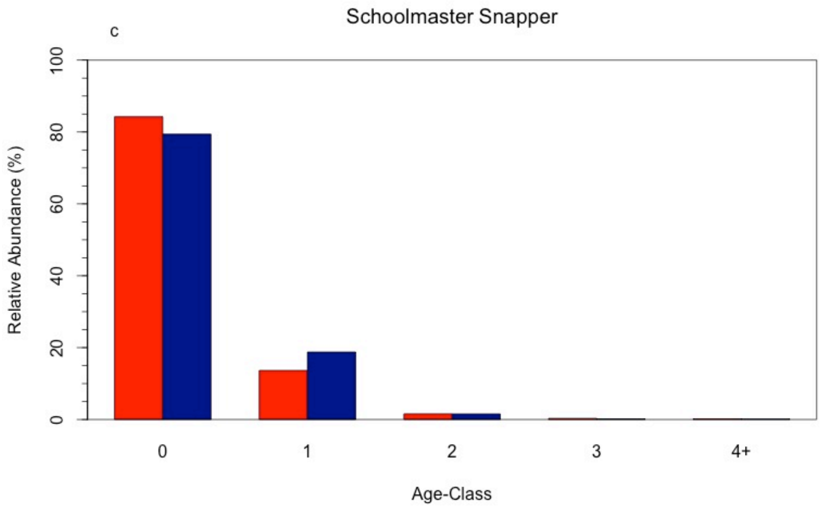
The 2010 dry season had the most recorded null values (seven out of ten occurrences, when assessing both mainland and leeward key locations for each of the five species of interest). The only other seasons with multiple null values across species and locations were the 2011 and 2012 dry seasons, with four each. The wet seasons typically saw higher percentages of presence and relative abundance compared to the dry seasons. Aside from yellowfin mojarra, the presence and relative abundance for the examined species were usually higher for the leeward key surveys compared to the mainland surveys. Out of the five species at both locations, only yellowfin mojarra at the leeward key locations recorded the highest number of fish per survey during a dry season (dry 2004).

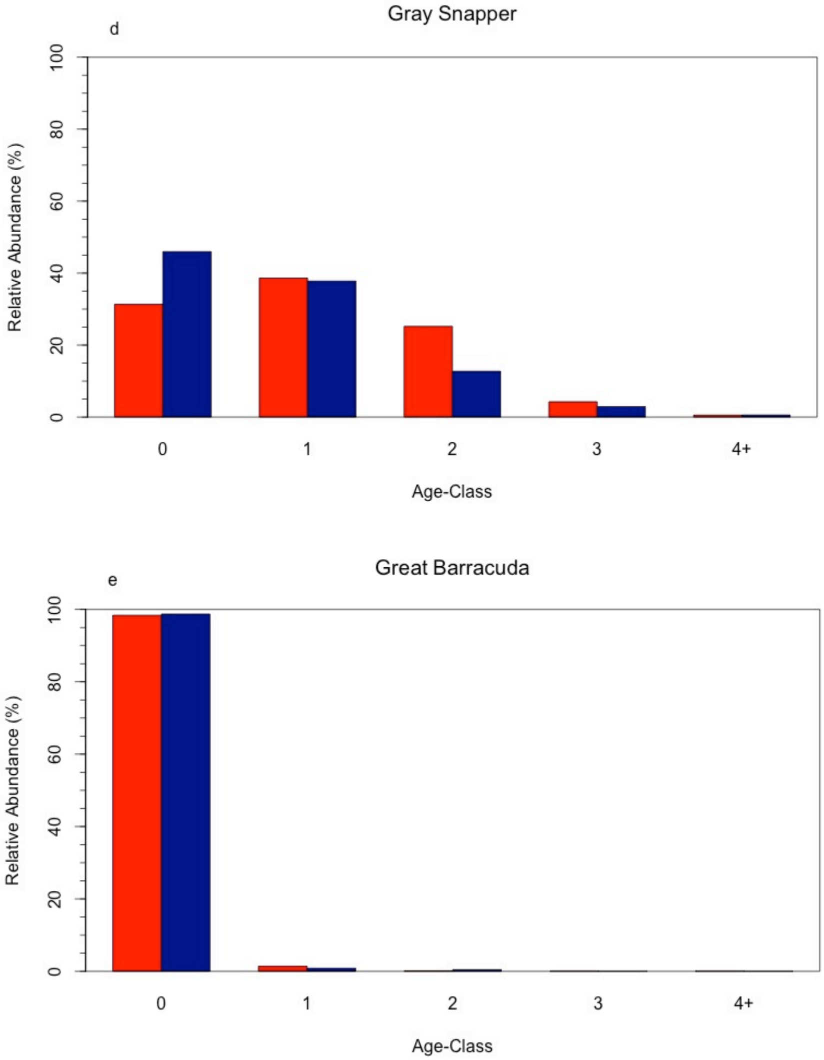
Relative Abundances by Age-Class

Plots of the species-specific length-frequency distributions of percent relative abundance by age-class can be seen in Figures 5a-e. Sergeant major and great barracuda inhabit both mainland and leeward key mangrove shorelines almost exclusively as Age 0 fish (97.9% and 98.5%, respectively). Yellowfin mojarra and schoolmaster snapper also inhabit these shorelines primarily as Age 0 fish (82.2% and 81.2%, respectively), but Age 1 fish were also prevalent (14.5% and 16.9%, respectively). Gray snapper is the only species of interest with high occurrences among Ages 0, 1, and 2 fish (39.4%, 38.2%, and 18.4%, respectively). The plots of seasonal length-frequency distributions from 2009-2011 for gray snapper can be seen in Figures 6a-l. For mainland locations, a total of seven cohorts were identified over the six-season period, with the season immediately following the January 2010 cold spell showing a dominance of a young cohort followed by slow growth of that cohort over the following season. For leeward key locations, a total of seven cohorts were identified over the six-season period, with the season

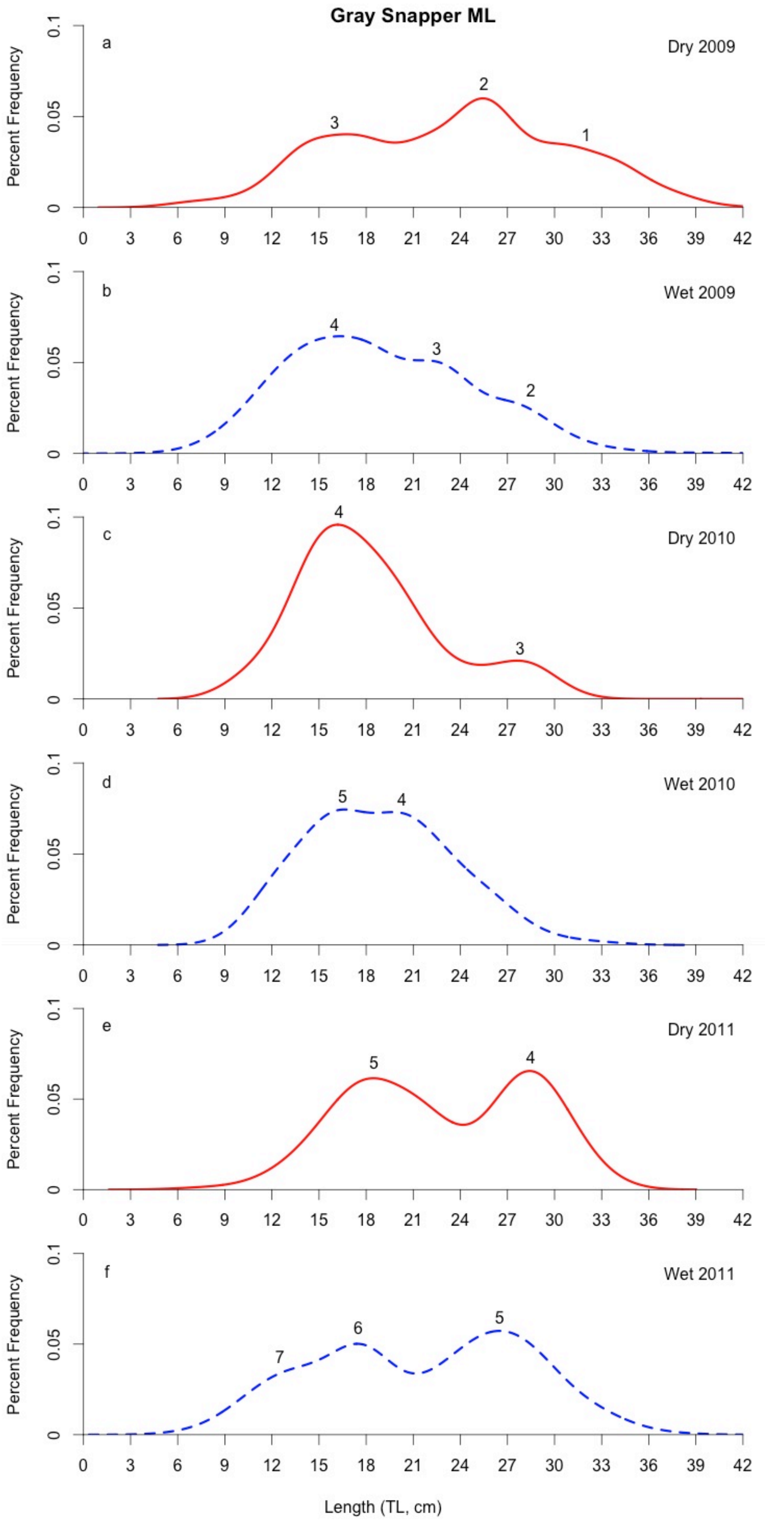

Sergeant Major
a
ML
LK
Relative Abundance (%)
0
20
40
60
80
100
0
1
2
3
4+
Age-Class

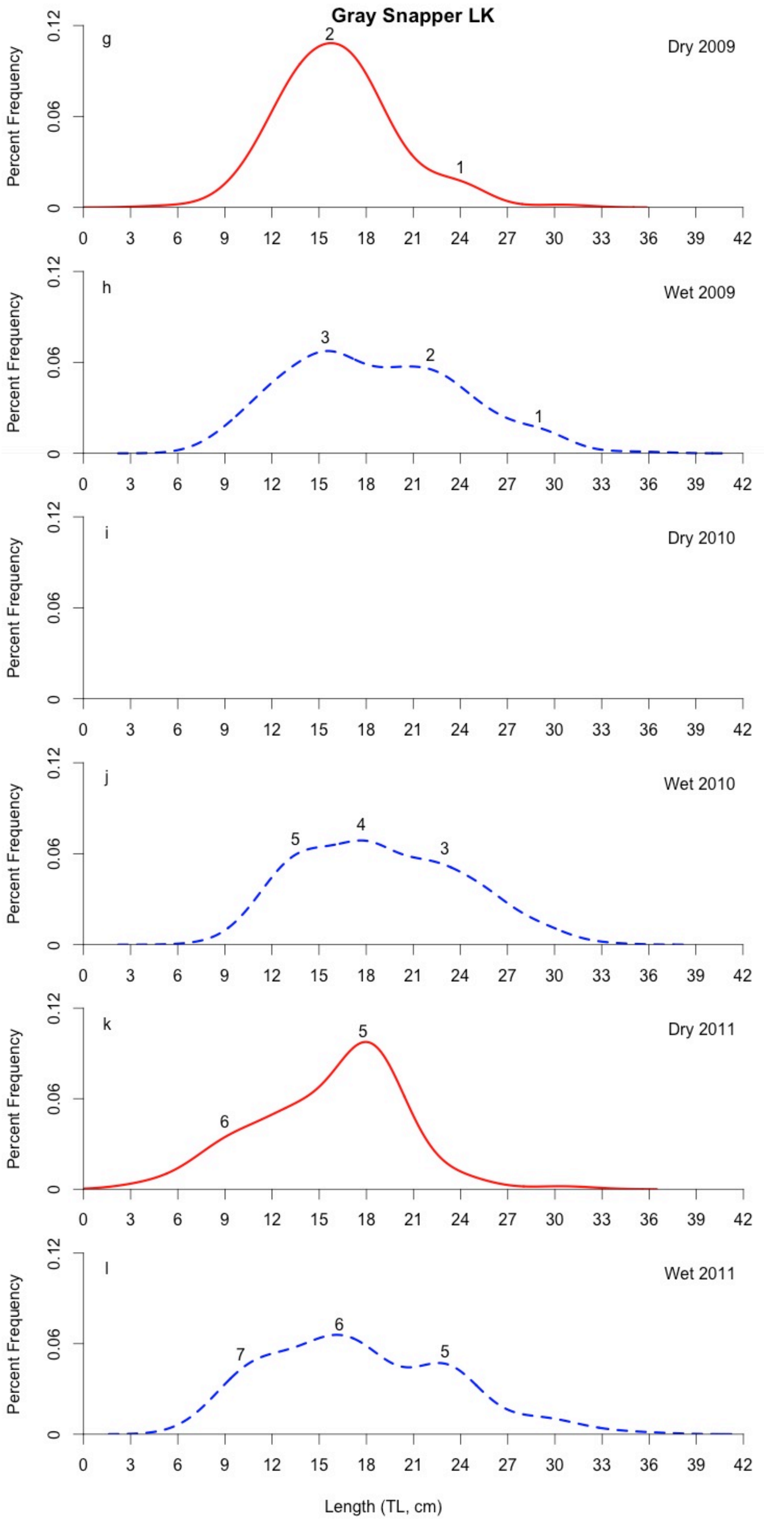

Yellowfin Mojarra
b
Relative Abundance (%)
0
20
40
60
80
100
0
1
2
3
4+
Age-Class


Schoolmaster Snapper
c
Relative Abundance (%)
0
20
40
60
80
100
0
1
2
3
4+
Age-Class



Figures 5a-e. Plots for each of the five species of interest showing the percent relative abundance distributed into age-classes. Both mainland and leeward key mangrove locations are shown. Percents were pooled across all examined survey seasons.


Gray Snapper ML
a
Dry 2009
3
2
1
b
Wet 2009
4
3
2
c
Dry 2010
4
3
d
Wet 2010
5
4
e
Dry 2011
5
4
f
Wet 2011
7
6
5
Percent Frequency
0
0.05
0.1
0 3 6 9 12 15 18 21 24 27 30 33 36 39 42
Length (TL, cm)



Figures 6a-l. Plots for gray snapper showing the length-frequency distributions by survey season from the 2009 dry season to the 2011 wet season as percent frequencies. Both mainland (a-f) and leeward key (g-l) locations are shown. Also shown are labels of the peaks to depict the existence and movement of varying cohorts.

immediately following the January 2010 cold spell completely absent of fish, yielding no length-frequency distribution.

Effects Post-Cold Spell

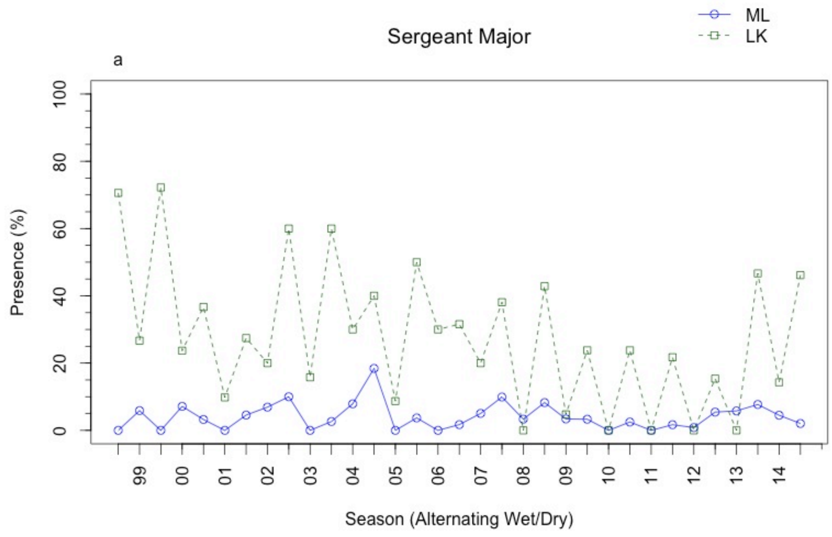
The immediate effects of the cold spell can be seen when examining for presence, relative abundance, and average total length (see Figures 7a-e, 8a-e, and 9a-e). The species of interest were absent from seven of the ten 2010 dry seasons. The three instances where fish were present during this season were gray snapper at mainland locations, and great barracuda at both locations. Data for the species- and location-specific length-frequency distributions of relative abundance by age-class and survey season depict a similar response for all five species examined. Immediately following the January 2010 cold spell, nearly all parameters for each species across all age-classes were equally affected.

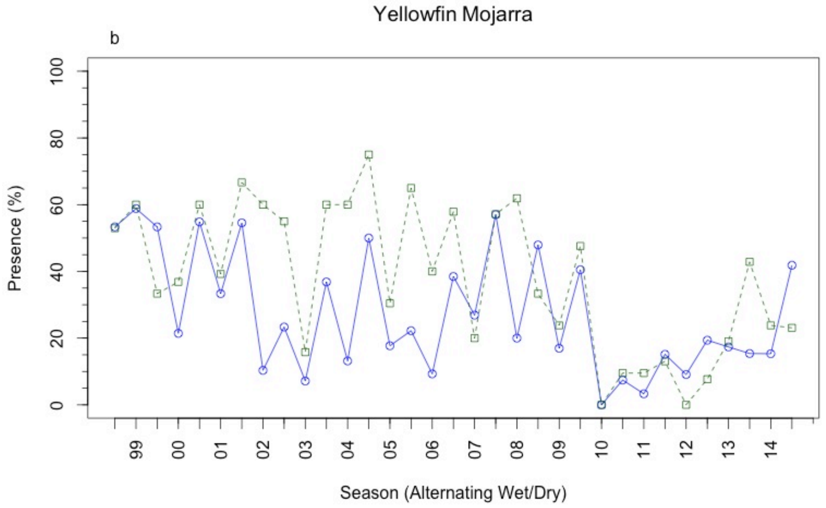
Variations in Presence

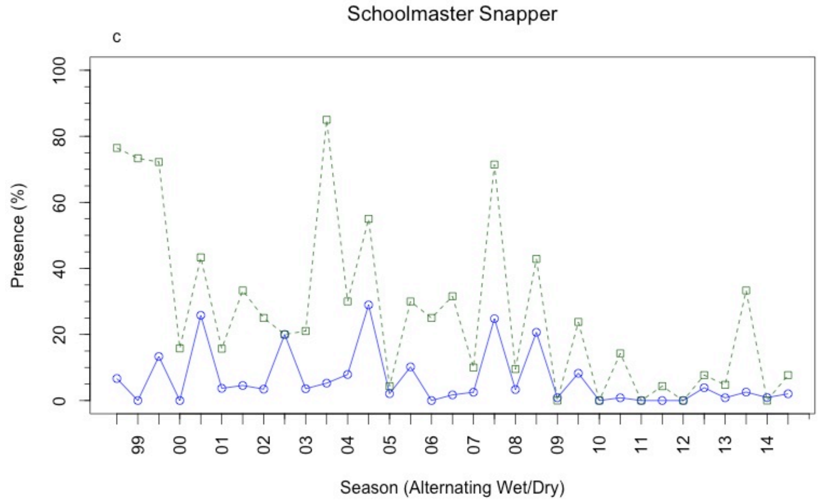
The results from the Chi-square analyses for presence can be found in Tables 11-15. There were statistically significant decreases in presence for a majority of the comparisons between time periods 1 and 2, including every instance for sergeant major and yellowfin mojarra. There were no instances of significant increases in presence between these two time periods for any of the five species. The results also suggest an increase in presence between times periods 2 and 3, however few of the results were significant. The only instances of significant increases in presence between time periods 1 and 3 were sergeant major at mainland locations during dry seasons, gray snapper at mainland locations during wet seasons, great barracuda at mainland locations during dry seasons, and great barracuda at leeward key locations during dry seasons.

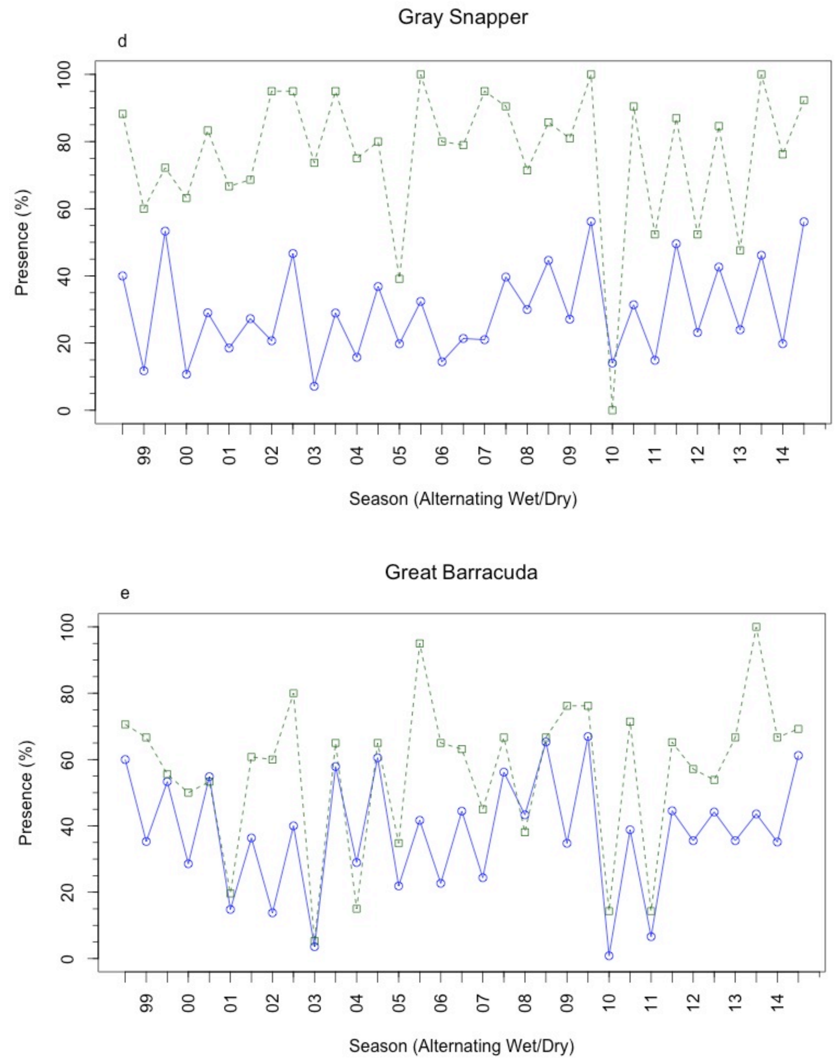
Variations in Relative Abundance

The results from the Kruskal-Wallis and Mann-Whitney U tests for relative abundance can be found in Table 16. Statistically significant differences existed for most of the species, except sergeant major at mainland locations during wet seasons, and gray snapper and great barracuda at leeward key locations during wet seasons. For those

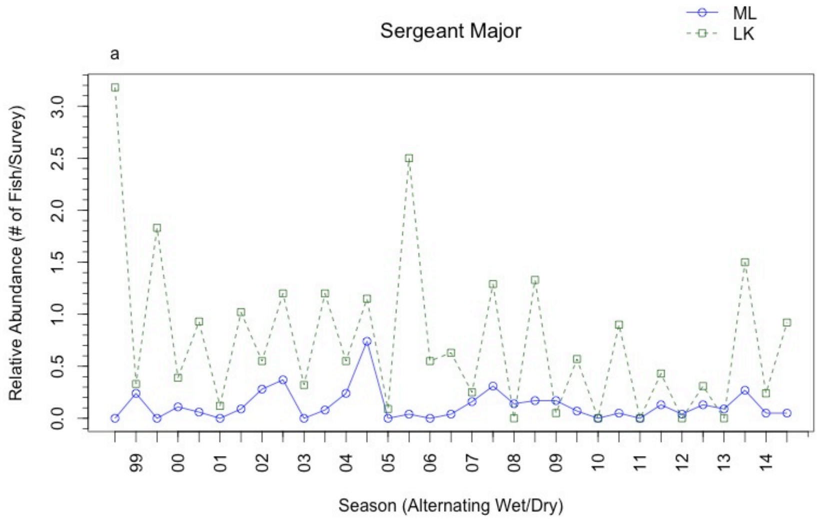

ML
LK
Sergeant Major
a
Presence (%)
0
20
40
60
80
100
99
00
01
02
03
04
05
06
07
08
09
10
11
12
13
14
Season (Alternating Wet/Dry)

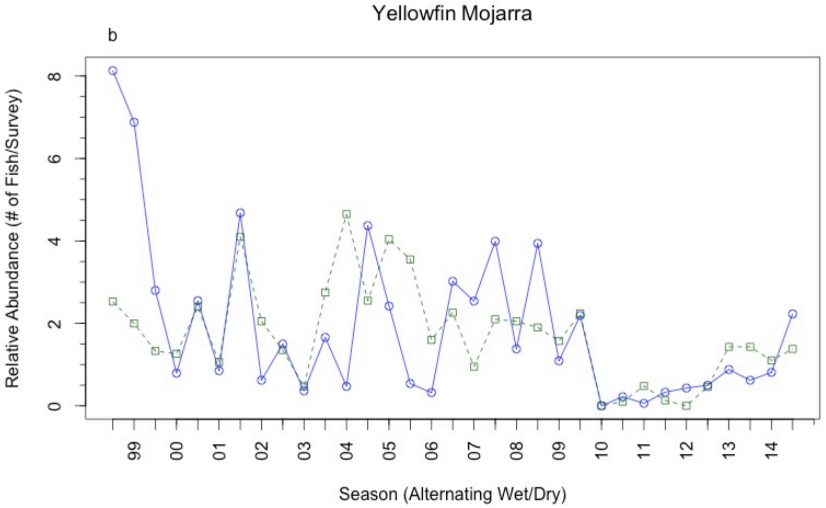

Yellowfin Mojarra
b
Presence (%)
0
20
40
60
80
100
99
00
01
02
03
04
05
06
07
08
09
10
11
12
13
14
Season (Alternating Wet/Dry)

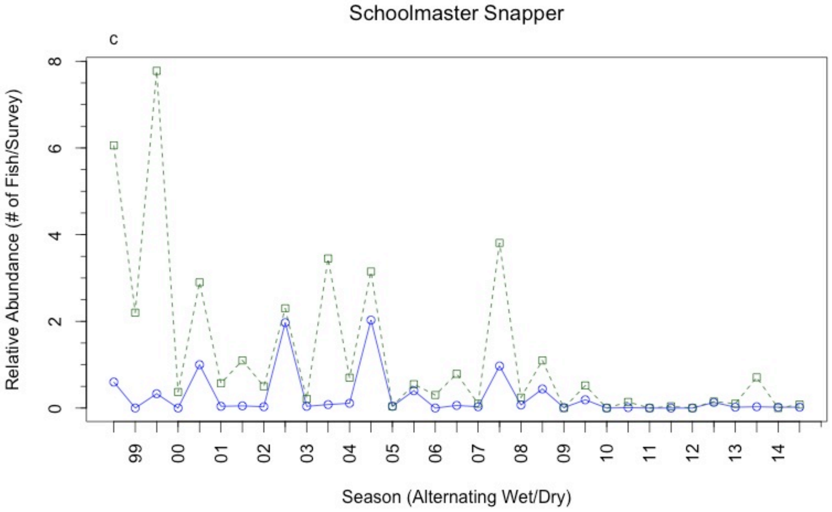

Schoolmaster Snapper
c
Presence (%)
0
20
40
60
80
100
99
00
01
02
03
04
05
06
07
08
09
10
11
12
13
14
Season (Alternating Wet/Dry)

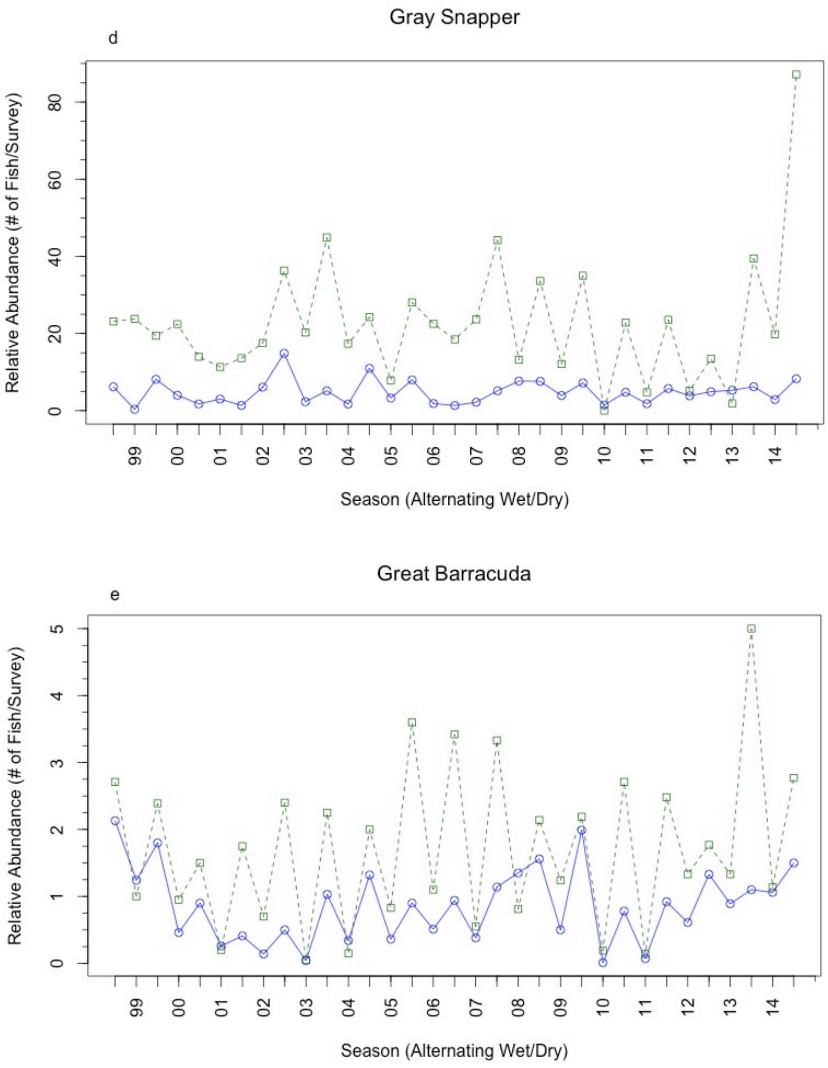


Figures 7a-e. Plots for each of the five species of interest showing presence, depicted here as the percentage of surveys each fish was present for during the survey seasons. Alternating wet and dry seasons from the 1998 wet to the 2014 wet season are shown. Both mainland (green dashed) and leeward key (blue solid) mangrove locations are shown for each species.

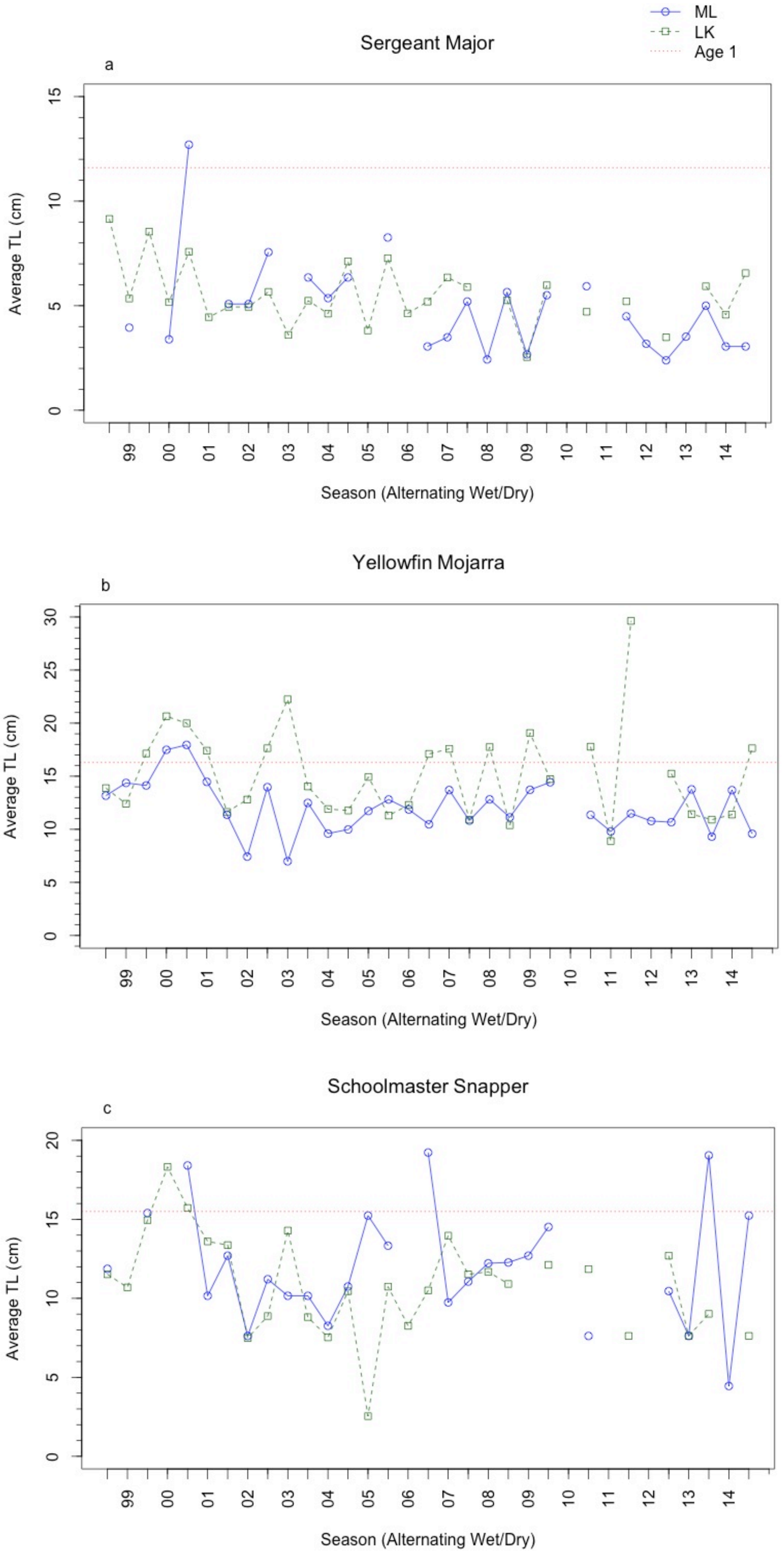

Sergeant Major
a
ML
LK
Relative Abundance (# of Fish/Survey)
0.0
0.5
1.0
1.5
2.0
2.5
3.0
99
00
01
02
03
04
05
06
07
08
09
10
11
12
13
14
Season (Alternating Wet/Dry)

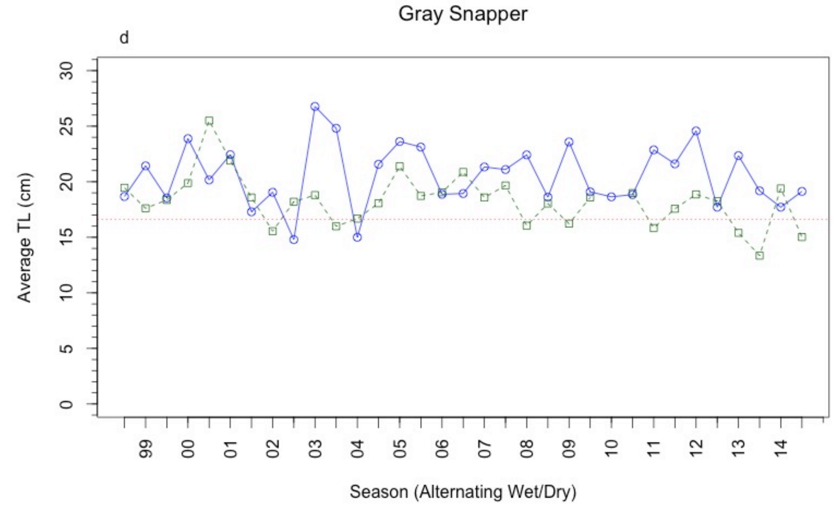

Yellowfin Mojarra
b
Relative Abundance (# of Fish/Survey)
0
2
4
6
8
99
00
01
02
03
04
05
06
07
08
09
10
11
12
13
14
Season (Alternating Wet/Dry)

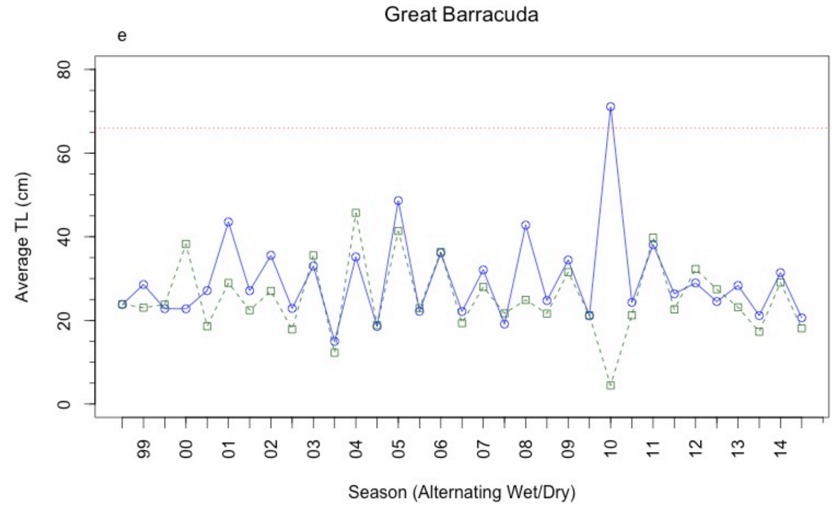

Schoolmaster Snapper
c
Relative Abundance (# of Fish/Survey)
0
2
4
6
8
99
00
01
02
03
04
05
06
07
08
09
10
11
12
13
14
Season (Alternating Wet/Dry)



Figures 8a-e. Plots for each of the five species of interest showing relative abundance, depicted here as the average number of fish recorded per survey. Alternating wet and dry seasons from the 1998 wet to the 2014 wet season are shown. Both mainland (green dashed) and leeward key (blue solid) mangrove locations are shown for each species. Note variations in the y-axis scales.


ML
LK
Age 1
Sergeant Major
a
Average TL (cm)
0
5
10
15
99
00
01
02
03
04
05
06
07
08
09
10
11
12
13
14
Season (Alternating Wet/Dry)
Yellowfin Mojarra
b
Average TL (cm)
0
5
10
15
20
25
30
99
00
01
02
03
04
05
06
07
08
09
10
11
12
13
14
Season (Alternating Wet/Dry)
Schoolmaster Snapper
c
Average TL (cm)
0
5
10
15
20
99
00
01
02
03
04
05
06
07
08
09
10
11
12
13
14
Season (Alternating Wet/Dry)





Figures 9a-e. Plots for each of the five species of interest showing the average total length of fish recorded during each survey season. Surveys included are from the 1998 wet season to the 2014 wet season by alternating wet and dry season. Both mainland (green dashed) and leeward key (blue solid) mangrove locations are shown for each species. The length at Age 1 for each species is also shown to depict whether or not the average size of fish was that of a juvenile at Age 0. Note variations in the y-axis scale.

Table 11. Chi-square analysis results for comparisons of the presence for sergeant major at both mangrove locations. Comparisons were made between pre-cold spell years (1: 1998-2009), the two years immediately following the cold spell (2: 2010-2011), and the remaining post-cold spell years (3: 2012-2014). Significant results are listed in **bold.** Also shown is whether the difference among the Chi-square analysis corresponded to an increase or decrease in presence.

				Chi-Square				
Species	**Site**	**Season**	**vs**	χ^2	**df**	***p***	**Φ**	**↑ or ↓**
Sergeant major	ML	Wet	1:2	779.981	1	**< .001**	-.876	↓
			1:3	816.945	1	**< .001**	-.869	↓
			2:3	3.053	1	.081	.075	↑
		Dry	1:2	852.110	1	**< .001**	-.943	↓
			1:3	918.806	1	**< .001**	-.927	↑
			2:3	9.111	1	**.003**	.124	↑
	LK	Wet	1:2	6.592	1	**.010**	-.143	↓
			1:3	0.459	1	.498	-.038	↓
			2:3	2.187	1	.139	.161	↑
		Dry	1:2	8.036	1	**.005**	-.161	↓
			1:3	5.688	1	.017	-.131	↓
			2:3	2.059	1	.151	.140	↑

Table 12. Chi-square analysis results for comparisons of the presence for yellowfin mojarra at both mangrove locations. Comparisons were made between pre-cold spell years (1: 1998-2009), the two years immediately following the cold spell (2: 2010-2011), and the remaining post-cold spell years (3: 2012-2014). Significant results are listed in **bold.** Also shown is whether the difference among the Chi-square analysis corresponded to an increase or decrease in presence.

				Chi-Square				
Species	**Site**	**Season**	**vs**	χ^2	**df**	***p***	**Φ**	**↑ or ↓**
Yellowfin mojarra	ML	Wet	1:2	78.458	1	< **.001**	-.278	↓
			1:3	26.623	1	< **.001**	-.157	↓
			2:3	17.717	1	< **.001**	.180	↑
		Dry	1:2	43.954	1	< **.001**	-.214	↓
			1:3	4.499	1	.034	-.065	↓
			2:3	26.458	1	< **.001**	.211	↑
	LK	Wet	1:2	31.422	1	< **.001**	-.312	↓
			1:3	14.221	1	< **.001**	-.211	↓
			2:3	2.656	1	.103	.178	↑
		Dry	1:2	19.693	1	< **.001**	-.252	↓
			1:3	14.731	1	< **.001**	-.211	↓
			2:3	2.437	1	.118	.152	↑

Table 13. Chi-square analysis results for comparisons of the presence for schoolmaster snapper at both mangrove locations. Comparisons were made between pre-cold spell years (1: 1998-2009), the two years immediately following the cold spell (2: 2010-2011), and the remaining post-cold spell years (3: 2012-2014). Significant results are listed in **bold.** Also shown is whether the difference among the Chi-square analysis corresponded to an increase or decrease in presence.

				Chi-Square				
Species	**Site**	**Season**	**vs**	χ^2	**df**	***p***	**Φ**	**↑ or ↓**
Schoolmaster snapper	ML	Wet	1:2	35.219	1	< **.001**	-.186	↓
			1:3	27.661	1	< **.001**	-.160	↓
			2:3	4.789	1	.029	.094	↑
		Dry	1:2	5.492	1	.019	-.076	↓
			1:3	3.965	1	.046	-.061	↓
			2:3	1.376	1	.241	.048	↑
	LK	Wet	1:2	21.814	1	< **.001**	-.260	↓
			1:3	11.934	1	**.001**	-.194	↓
			2:3	1.302	1	.254	.124	↑
		Dry	1:2	9.343	1	**.002**	-.174	↓
			1:3	11.403	1	**.001**	-.186	↓
			2:3	0.673	1	.412	.080	↑

Table 14. Chi-square analysis results for comparisons of the presence for gray snapper at both mangrove locations. Comparisons were made between pre-cold spell years (1: 1998-2009), the two years immediately following the cold spell (2: 2010-2011), and the remaining post-cold spell years (3: 2012-2014). Significant results are listed in **bold.** Also shown is whether the difference among the Chi-square analysis corresponded to an increase or decrease in presence.

				Chi-Square				
Species	**Site**	**Season**	**vs**	χ^2	**df**	***p***	**Φ**	**↑ or ↓**
Gray snapper	ML	Wet	1:2	0.329	1	.566	.018	↑
			1:3	8.197	1	**.004**	.087	↑
			2:3	3.019	1	.082	.074	↑
		Dry	1:2	4.846	1	.028	-.071	↓
			1:3	0.299	1	.584	.017	↑
			2:3	5.810	1	**.016**	.099	↑
	LK	Wet	1:2	0.504	1	.478	.040	↑
			1:3	1.794	1	.180	.075	↑
			2:3	0.363	1	.547	.066	↑
		Dry	1:2	32.502	1	< **.001**	-.324	↓
			1:3	3.743	1	.053	-.106	↓
			2:3	10.752	1	**.001**	.320	↑

Table 15. Chi-square analysis results for comparisons of the presence for great barracuda at both mangrove locations. Comparisons were made between pre-cold spell years (1: 1998-2009), the two years immediately following the cold spell (2: 2010-2011), and the remaining post-cold spell years (3: 2012-2014). Significant results are listed in **bold.** Also shown is whether the difference among the Chi-square analysis corresponded to an increase or decrease in presence.

				Chi-Square				
Species	**Site**	**Season**	**vs**	χ^2	**df**	***p***	**Φ**	**↑ or ↓**
Great barracuda	ML	Wet	1:2	12.221	1	< **.001**	-.110	↓
			1:3	2.253	1	.133	-.046	↓
			2:3	3.324	1	.068	.078	↑
		Dry	1:2	61.540	1	< **.001**	-.253	↓
			1:3	6.568	1	**.010**	.078	↑
			2:3	82.641	1	< **.001**	.373	↑
	LK	Wet	1:2	0.028	1	.867	.009	↑
			1:3	1.051	1	.305	.057	↑
			2:3	0.477	1	.490	.075	↑
		Dry	1:2	10.833	1	**.001**	-.187	↓
			1:3	10.732	1	**.001**	.180	↑
			2:3	24.786	1	< **.001**	.486	↑

Table 16. Kruskal-Wallis and Mann-Whitney U results for comparisons of relative abundance for the five target species at both locations and during both seasons. Comparisons were made between pre-cold spell years (1: 1998-2009), the two years immediately following the cold spell (2: 2010-2011), and the remaining post-cold spell years (3: 2012-2014). Significant results are listed in **bold.** Also shown is whether the difference among the Mann-Whitney U analysis corresponded to an increase or decrease in relative abundance.

			Kruskal-Wallis			**Mann-Whitney U**					
Species	**Site**	**Season**	χ^2	**df**	p	**1 - 2**	**↑↓**	**1 - 3**	**↑↓**	**2 - 3**	**↑↓**
Sergeant major	ML	Wet	5.350	2	.069	---	---	---	---	---	---
		Dry	8.424	2	**.015**	**.006**	↓	.626	↓	**.003**	↑
	LK	Wet	7.000	2	**.03**	**.010**	↓	.368	↓	.157	↑
		Dry	13.297	2	**.001**	**.005**	↓	**.015**	↓	.163	↑
Yellowfin mojarra	ML	Wet	101.297	2	**<.001**	**<.001**	↓	**<.001**	↓	**<.001**	↑
		Dry	45.450	2	**<.001**	**<.001**	↓	.023	↓	**<.001**	↑
	LK	Wet	41.571	2	**<.001**	**<.001**	↓	**<.001**	↓	.070	↑
		Dry	25.110	2	**<.001**	**<.001**	↓	**.002**	↓	.070	↑
Schoolmaster snapper	ML	Wet	60.042	2	**<.001**	**<.001**	↓	**<.001**	↓	.029	↑
		Dry	8.923	2	**.012**	.019	↓	.048	↓	.241	↑
	LK	Wet	34.646	2	**<.001**	**<.001**	↓	**<.001**	↓	.199	↑
		Dry	20.374	2	**<.001**	**.002**	↓	**.001**	↓	.425	↑
Gray snapper	ML	Wet	10.315	2	**.006**	.656	↓	**.001**	↑	.038	↑
		Dry	6.740	2	**.034**	.022	↓	.567	↑	**.012**	↑
	LK	Wet	2.002	2	.368	---	---	---	---	---	---
		Dry	31.768	2	**<.001**	**<.001**	↓	**.008**	↓	**.003**	↑
Great barracuda	ML	Wet	16.777	2	**<.001**	**<.001**	↓	.053	↑	.048	↑
		Dry	82.928	2	**<.001**	**<.001**	↓	**.006**	↑	**<.001**	↑
	LK	Wet	1.877	2	.391	---	---	---	---	---	---
		Dry	27.782	2	**<.001**	**.001**	↓	**<.001**	↑	**<.001**	↑

instances where the Kruskal-Wallis test determined significant differences, the post-hoc Mann-Whitney U tests determined significant decreases in almost every comparison between the years prior to the cold spell (time period 1) and the two years immediately following the cold spell (time period 2). The only results that were not significant were schoolmaster snapper at mainland locations during dry seasons, and gray snapper at mainland locations during both wet and dry seasons. Eleven of the 17 comparisons between time periods 1 and 3 reported statistically significant differences, eight of which were significant decreases, while gray snapper at mainland locations during wet seasons and great barracuda at both locations during dry seasons showed significant increases. Only seven of the 17 comparisons between time periods 2 and 3 reported statistically significant differences, all of which were increases in relative abundance.

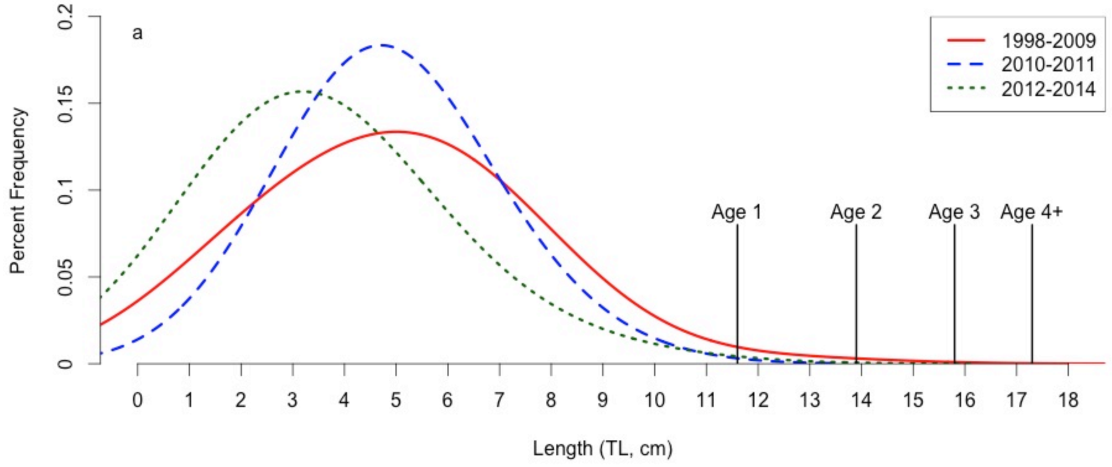
Variations in Size Structure

The results from the Kolmogorov-Smirnov tests for length-frequency distributions can be found in Table 17. Plots for percent frequency of lengths comparing the three time periods for each species at both locations can be found in Figures 10a-j. Statistically significant differences existed between three of the six time period comparisons for sergeant major. Yellowfin mojarra exhibited statistically significant differences between all three time periods at mainland locations, while statistically significant differences existed only when comparing time periods 1 and 3 at leeward key locations for schoolmaster snapper. Gray snapper results, however, indicated statistical significant differences between all time periods at both locations. Great barracuda only recorded statistically significant differences between time periods 1 and 2 at mainland locations.

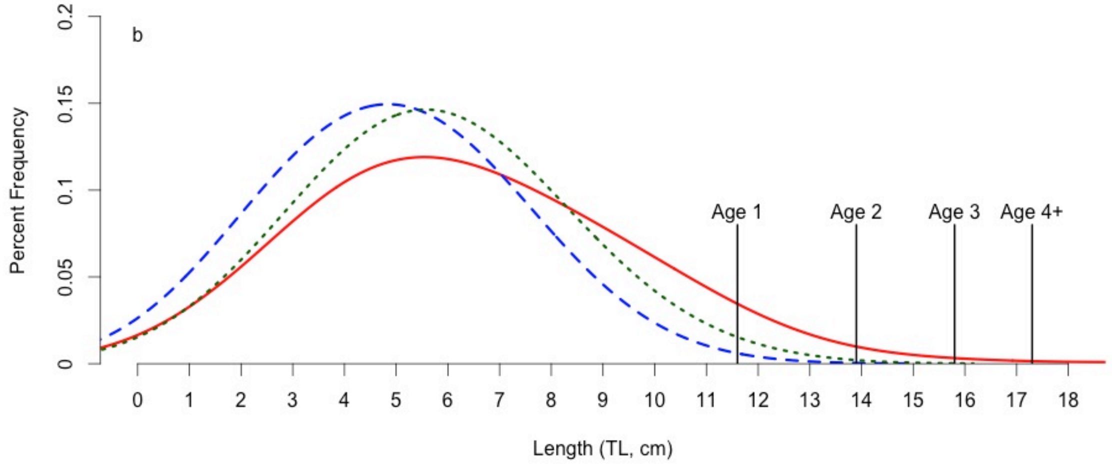
Table 17. Kolmogorov-Smirnov results for comparisons of the length-frequency distributions for the five target species at both mangrove locations. Comparisons were made between pre-cold spell years (1: 1998-2009), the two years immediately following the cold spell (2: 2010-2011), and the remaining post-cold spell years (3: 2012-2014). Significant results are listed in **bold.**

		Kolmogorov-Smirnov					
		1 vs 2		**1 vs 3**		**2 vs 3**	
Species	**Site**	**D**	***p***	**D**	***p***	**D**	***p***
Sergeant major	ML	0.3255	.036	0.3489	**<.001**	0.6645	**<.001**
	LK	0.3283	**.004**	0.1849	.146	0.2143	.398
Yellowfin mojarra	ML	0.1942	**.008**	0.1898	**<.001**	0.2303	**.002**
	LK	0.3725	.025	0.2002	**.001**	0.2532	.341
Schoolmaster snapper	ML	0.8323	.494	0.2553	.081	0.6538	.805
	LK	0.2947	.880	0.4478	**.005**	0.3667	.790
Gray snapper	ML	0.0774	**<.001**	0.0993	**<.001**	0.1038	**<.001**
	LK	0.0927	**<.001**	0.2593	**<.001**	0.2321	**<.001**
Great barracuda	ML	0.1190	**.010**	0.0680	.025	0.0912	.126
	LK	0.0907	.335	0.0467	.854	0.0857	.616

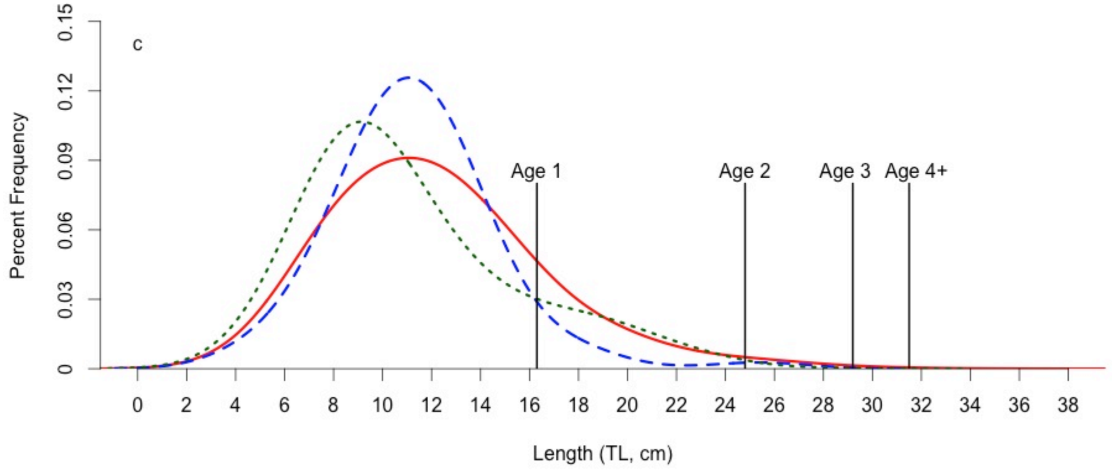
Sergeant Major ML

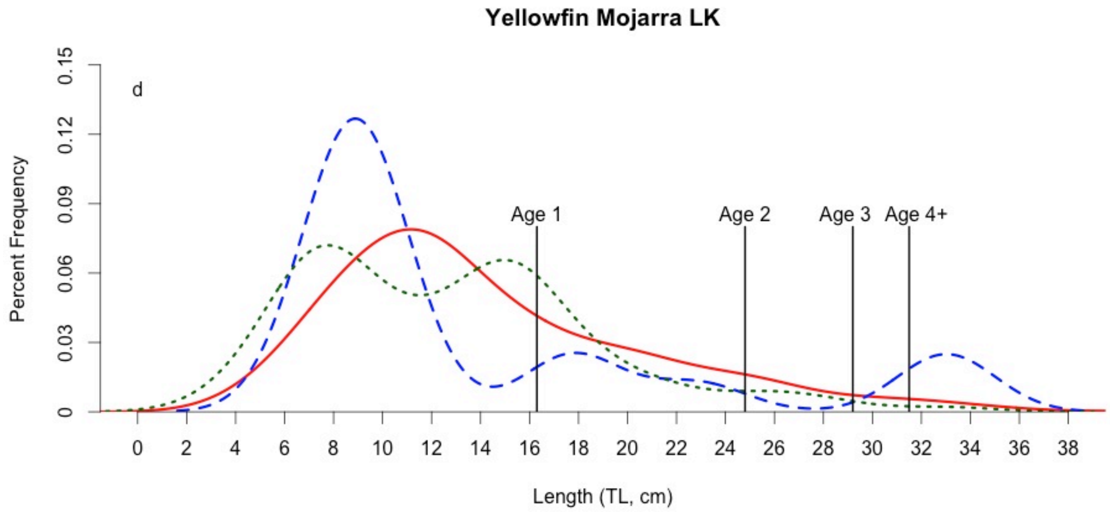


Sergeant Major LK

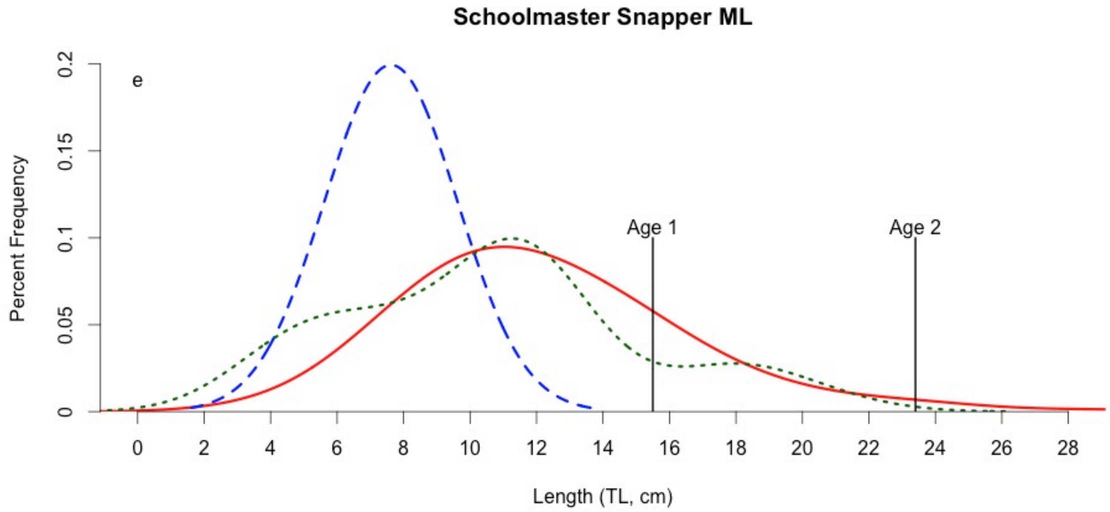


Yellowfin Mojarra ML

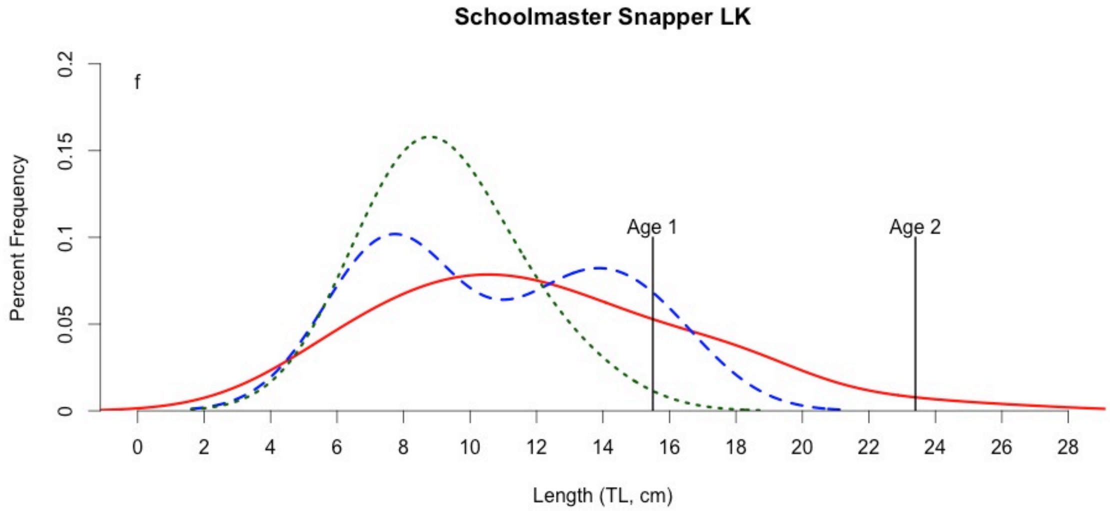




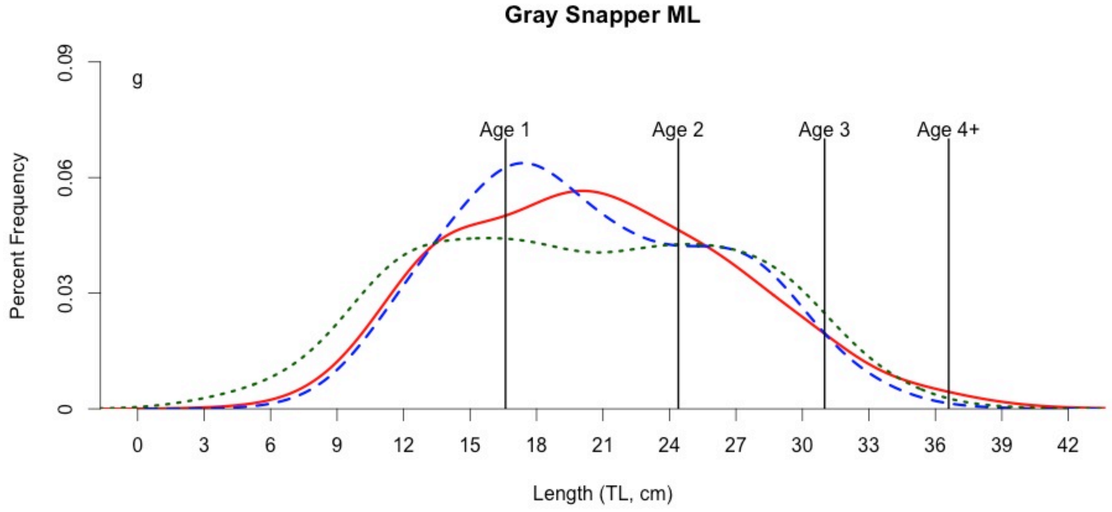
Yellowfin Mojarra LK

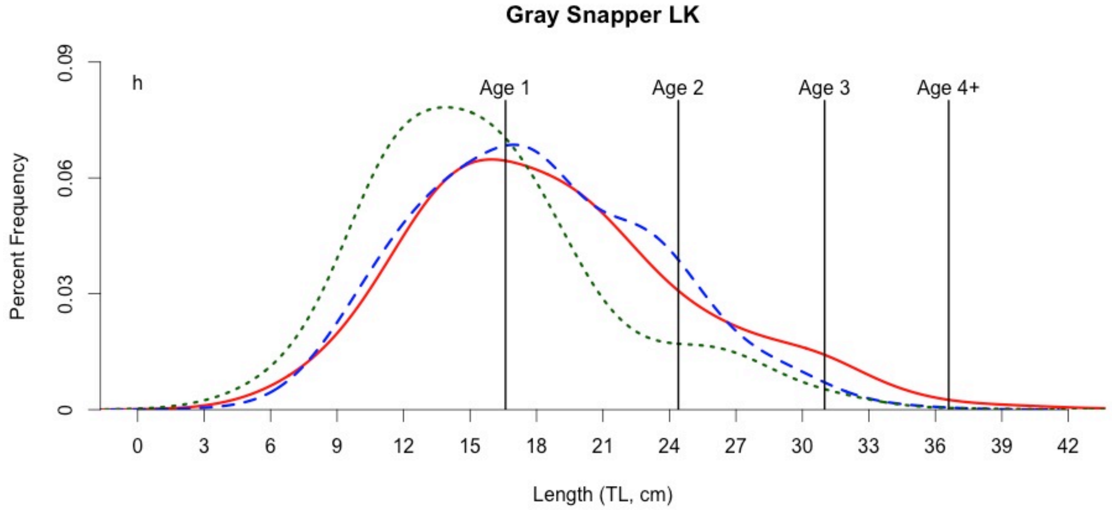


Schoolmaster Snapper ML



Schoolmaster Snapper LK


Gray Snapper ML
g
Percent Frequency
0
0.03
0.06
0.09
Age 1
Age 2
Age 3
Age 4+
0
3
6
9
12
15
18
21
24
27
30
33
36
39
42
Length (TL, cm)


Gray Snapper LK
h
Percent Frequency
0
0.03
0.06
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Age 1
Age 2
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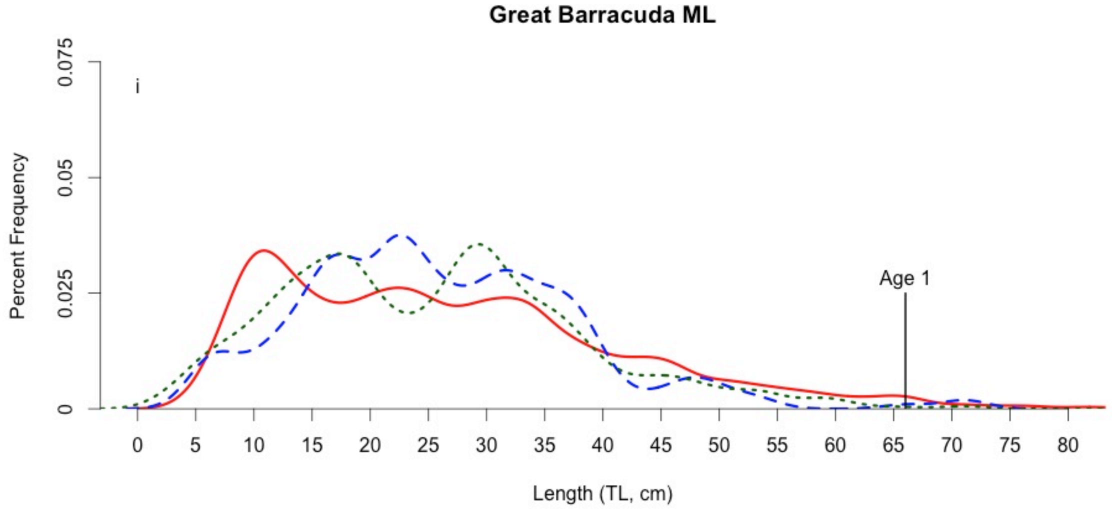
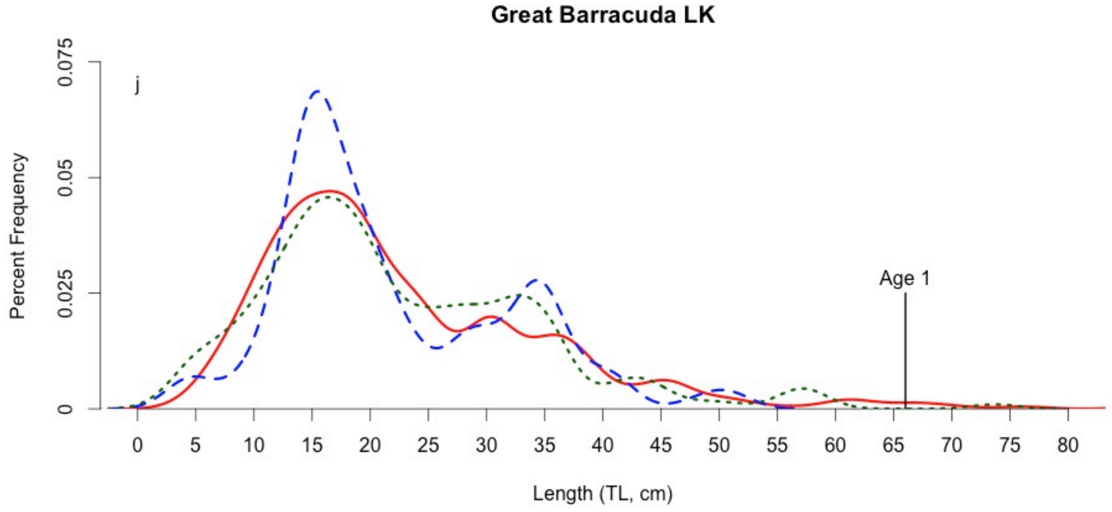
Length (TL, cm)


Great Barracuda ML
i
Percent Frequency
0
0.025
0.05
0.075
Age 1
0
5
10
15
20
25
30
35
40
45
50
55
60
65
70
75
80
Length (TL, cm)



Figures 10a-j. Plots for each of the five target species showing the combined length-frequency distributions for each of the three time periods (1998-2009, 2010-2011, 2012-2014) as percent frequencies. Both mainland and leeward key locations are shown. Also shown are the corresponding predicted lengths for age-classes. Note variations in the x- and y-axis scales.

Discussion

The findings from this study suggest that five economically and ecologically important coral reef species that utilize mangrove locations as nursery habitats in BNP are affected by the environmental conditions associated with cold spells. The impact of cold spells on fish assemblages has been documented in the literature (e.g., Storey and Gudger 1936; Miller 1940; Galloway 1941; Bohnsack 1983; Gilmore et al. 1978; Adams et al. 2012), although few studies have gone beyond anecdotal reports with quantitative assessments. The ability to determine mortality and/or emigration of varying age-classes for these species following a cold spell allows for more informed policy decisions, such as altered catch-limits or the implementation of no-take zones, to prevent further deterioration of stocks suffering declines. This study is unique in that it gathered scientific data previously unavailable and examined trends following a cold spell event. In sampling via transects around mangrove stands, the data set is more comprehensive and inclusive of difficult survey locations, across the span of multiple years, and in both wet and dry seasons. As such, this study was able to assess alterations in presence, abundance, and size structure of sergeant major, yellowfin mojarra, schoolmaster snapper, gray snapper, and great barracuda in a mangrove nursery habitat following a cold spell event.

The results from the examination of the YSI water temperature data revealed that the majority of 15-minute intervals from January 6-14, 2010 fell below three standard deviations of the January average water temperatures. The documented January 2010 cold spell, therefore had a direct impact on the water temperature of the study locations. When comparing the water temperature records of the two mangrove locations, the average monthly water temperatures were colder at the leeward key location, but were less variable, given the smaller standard deviations. This can be explained by the differences in the average water depths at the locations, with the mainland locations shallower and therefore more easily warmed, but also with limited heat storage capacity (Roberts et al. 1982) that leads to more variable temperatures. The buffering capacity of the open ocean water at the deeper leeward key locations, however, makes for smaller temperature fluctuations and therefore provides a more stable and suitable nursery habitat for juvenile fishes in terms of water temperature alterations.

Additional time intervals that experienced statistically significant water temperature decreases have not been previously addressed in the current body of literature as documented cold spells. One such reason is that it is likely that these anomalies took place during cold periods of shorter duration and less extreme cold air temperatures. Another reason for these intervals being understudied is that the temperatures may have dropped slowly. Slower temperature decreases do not garner as much attention as more severe cold spells, such as the one that occurred in January 2010. Third, slow temperature decreases in combination with the mixing of shallower, cooler surrounding water due to winds associated with cooler periods (Walker et al. 1982) would lead to less noticeable effects despite the water temperature still significantly decreasing. Lastly, if the cooling of the water occurred slowly over a longer period of time, species would be more capable of acclimation or relocation prior to the water temperatures reaching their lowest point (Moore 1976). These potential cold spells and their resulting effects should be examined in future studies. The population variables of fishes inhabiting this study area may have been affected to some extent and the results could add to the current body of literature on the effects of less severe cold periods.

The presence, abundance, and average total length figures for the five species reveal differences between species, seasons, and among the mangrove locations. Gray snapper was by far the most abundant of the five species and was the second most present during surveys behind great barracuda. Sergeant major was the least present and abundant, followed in both categories by schoolmaster snapper. These results may suggest one of two possibilities: (1) gray snapper and great barracuda are generally much more prevalent among the mangrove shorelines of the study area as compared to the other three species; 2) gray snapper and great barracuda are more easily noticed and identified during visual surveys compared to sergeant major and schoolmaster snapper.

Previous research has shown that yellowfin mojarra are commonly found in high densities as juveniles on mudflats in addition to their primary juvenile habitat among mangroves (Nagelkerken et al. 2000), leading to the conclusion that they have a low dependence on mangrove habitats (Nagelkerken et al. 2001). This would account for the lower presence of yellowfin mojarra at the study sites. Previous research has shown that sergeant major and schoolmaster snapper show significant correlations of abundance between mangrove and reef habitats, but with a temporal lag time of two years, suggesting the potential utilization of intermediate habitats (Jones et al. 2010). This also suggests that these two species may not reside among the mangrove shorelines for as long as the other target species, leading to a lower presence compared to gray snapper and great barracuda. A high abundance of Age 0 juveniles for sergeant major was found in both mangrove and reef habitats (Jones et al. 2010), which could split the recruitment among different locations, reducing the overall presence at the mangrove study sites.

In regards to the second potential explanation, for the discrepancy in visual observations for presences, the mangrove locations of study are known nursery habitats for small juveniles (Odum et al. 1982; Dennis 1992; Ault et al. 1998; Bohnsack et al. 1999; Nagelkerken et al. 2000; Mumby et al. 2004; Faunce and Serafy 2006; Dorenbosch et al. 2007). It is likely that the majority of the observed fishes were smaller than a respective Age 1 size. For the five target species, there are also noticeable size differences in Age 1 fish. The theoretical size of an Age 1 sergeant major is the smallest of the five target species (11.6 cm) and the second smallest Age 1 fish is the schoolmaster snapper (15.5 cm). For the two species with highest presence, gray snapper have a slightly larger theoretical size at Age 1 (16.6 cm) and great barracuda have the largest theoretical size at Age 1 (66.0 cm). In addition, the mean size of fish observed during all surveys for sergeant major and schoolmaster snapper were the two smallest of the five target species, at 5.7 cm and 12.2 cm, respectively, while gray snapper and great barracuda recorded the two largest mean sizes at 19.3 cm and 25.2 cm, respectively.

The results for the age-class distributions suggested that sergeant major and great barracuda utilized the survey locations almost exclusively as Age 0 fish. Age 0 yellowfin mojarra and schoolmaster snapper primarily inhabited the mangrove shorelines, but a smaller percentage of Age 1 fish were also present, suggesting that these two fishes utilize the mangroves for a longer duration in their lifespan. Lastly, approximately 40% of Age 0 and 40% of Age 1 gray snapper were found to utilize the mangroves while a smaller, though still substantial, percentage of Age 2 fish (almost 18%) were also present. These age-class distributions support the previous research suggesting that the mangroves are used predominantly for young juveniles before starting their ontogenetic habitat shift to the adjacent coral reefs (Faunce and Serafy 2007). However, it also suggests that larger, and therefore older, gray snapper were also regularly identified. This is supported by previous research that noted gray snapper typically migrate to mangrove locations from seagrass beds at approximately 8-10 months of age and use the mangrove shorelines of southeastern Florida for approximately two years (Faunce and Serafy 2007, 2008). Therefore, the larger average size of gray snapper due to the prevalence of Age 0-2 fish at the study site and the comparative larger size of great barracuda even as Age 0 fish may make these two species more readily identified during visual surveys.

Size of fish, however, is not always the most effective predictor of positive visual survey identifications. According to Brian Teare, a NOAA affiliate and the observer for the mangrove visual surveys over the past five years, accurate identification during visual surveys in mangrove habitats depends more on the species-specific behavioral traits of the fishes than the size of the individual (pers. comm.; June 9, 2015). The great barracuda, for instance, tends to hover motionless to aid in the ambush predator foraging technique, while gray snapper can actually be attracted to a surveyor if they are not moving at a consistent pace during the data collection along the transect (Teare 2014). However, based on the large difference in overall numbers of fishes observed and previous research on gray snapper and great barracuda (Serafy et al. 2003; Faunce and Serafy 2008), it is most likely that gray snapper and great barracuda are the most prevalent of the five target species at both of the mangrove study locations.

Differences among presence and abundance parameters were clear when comparing dry and wet seasons. Both variables were typically greater during wet seasons for all five species. Although little information is available on the recruitment times for the species of interest (see Robertson et al. 1993; Kadison et al. 2010), gray snapper were reported to spawn during the summer and fall months outside of Everglades National Park (Rutherford et al. 1989). There have also been studies pertaining to the variations between dry and wet seasons (de Sylva 1963; Barimo and Serafy 2003; Serafy et al. 2007) that agree with the results of this study. Great barracuda, for example, was less aggregated during dry seasons, potentially due to environmental pressures and territoriality issues (de Sylva 1963). Recruitment to the inshore mangrove habitats during the wet season would cause an influx in both presence and abundance, allowing for young juveniles to grow at a quicker pace due to the increased water temperatures.

In terms of variations between mangrove locations, the overall presence and mean abundance for surveys conducted at leeward key locations were greater for all five target species compared to the surveys conducted at mainland locations (Tables 6 and 7). The average total length for all surveys, however, was greater at leeward key locations for sergeant major and yellowfin mojarra, but greater at mainland locations for the three remaining target species (Table 8). A greater proportion of fishes by survey area inhabit the leeward key locations and for schoolmaster snapper, gray snapper, and great barracuda, they are smaller in size, suggesting a greater population of younger fishes. One potential reason for these discrepancies is that the leeward key shorelines in BNP are smaller in area as compared to the mainland shorelines, resulting in a denser nursery habitat. Further, the leeward key mangroves may be preferred over the mainland mangroves due to the previously mentioned buffering capacity provided by the adjacent Gulf Stream ocean conditions. These results support the notion that the leeward key locations are better suited as nursery habitats compared to the mainland locations.

The primary hypotheses for this study centered on the effects of the January 2010 cold spell. During the 2010 dry season, which was the first survey season following the event, not a single sergeant major, yellowfin mojarra, or schoolmaster snapper was recorded in any of the 142 surveys (121 at mainland locations and 21 at leeward key locations). There were also no gray snapper recorded at leeward key locations and there were only five great barracuda present during four of the 142 surveys (one fish seen during one survey at mainland locations and four fish seen during three surveys at leeward key locations). Although gray snapper were still present at mainland locations, there numbers were still drastically reduced compared to previous seasons. These results alone suggest clear and recognizable effects from the cold spell on the overall presence and abundance of each target species.

The Chi-square analysis results for comparisons of presence and the Kruskal-Wallis and post-hoc Mann-Whitney U analysis results for comparisons of abundance between the three time periods suggest sergeant major, yellowfin mojarra, and schoolmaster snapper all experienced nearly the same outcome following the cold spell. That is, both parameters indicated statistically significant decreases from time period 1 to time period 2 and although a majority of the results show increases for both parameters from time period 2 to 3, few of the results were significant and in a majority of cases, significant decreases still existed between time period 1 and 3. Results for presence were expected to rebound quicker than abundance, considering a species was deemed present during a survey as long as a single individual was documented. However, in most cases, the recorded presence in the subsequent seasons post-cold spell were as depressed as their abundance counterparts. These results suggest that even a brief cold spell, such as the one of specific interest during January 2010, can have significant effects on mangrove utilizing reef fishes for many years following the event.

The two remaining species, gray snapper and great barracuda, did not exhibit as severe of a response during the two years immediately following the cold spell, suggesting either fewer mortalities, a quick return of fish that emigrated away from the study area during the event, or recruitment of new fish to the mangrove locations. Thermal tolerance studies for the five target species are lacking. However, a close examination of the adult ranges should assist in assessing which species are more susceptible to cold-water temperatures, since temperature is a determining factor of the occupied locations of a species when combined with ontogeny and life history (Wuenschel et al. 2012).

Since yellowfin mojarra and schoolmaster snapper have the most limited range of the five target species (Allen 1985; Gilmore and Greenfield 2002), they are most likely to have thermal tolerance minimums that are higher than the other species, which is a potential explanation for their drastic reductions in presence and abundance following the cold spell. Sergeant major, gray snapper, and great barracuda all have larger north-south ranges in the western Atlantic (Rutherford et al. 1983; Alshuth et al. 1998; Allman and Grimes 2002; D'Alessandro et al. 2011), suggesting that they might be less susceptible to cold water temperatures. The analysis results point towards this being a potential explanation for gray snapper and great barracuda, although sergeant major did not fair well following the cold spell. This may be due to a combination of factors, such as their original low levels of presence and abundance in the study location, their small size compared to the other target species, a high level of predation on cold-affected individuals, or their inability to emigrate outside of the affected shallow water habitats.

The size structure of the target species was predicted to shift towards larger individuals following the cold spell, as the primary cold effects and secondary physiological and predatory effects would result in the mortalities of the inhabiting juvenile fishes (Adams et al. 2012; Bohnsack 1983; Doudoroff 1942; Doudoroff 1945; Mora and Ospina 2002). This would leave the larger individuals, whether due to reduced susceptibility to the cold effects or the return to the study location following a brief emigration by the more gregarious older fish. However, the study results did not support this hypothesis. When compared across time periods, the length-frequency distributions plotted in Figures 10a-j suggest an immediate shift to smaller fish in the two years immediately following the cold spell. The results of the Kolmogorov-Smirnov analysis show statistically significant differences for a half of the comparisons among the target species for the length-frequency distributions between time periods 1 and 2. Six out of the ten comparisons between time periods 1 and 3 also show statistically significant results, suggesting that the size structures for some of the target species did not return to pre-stress levels by the end of the study period.

Although the connection between size of fish and susceptibility to reduced water temperatures has not been extensively investigated, there have been several studies describing larger white perch (*Morone americana*) and summer flounder (*Paralichthys dentatus*) of the Age 0 class surviving for longer periods of time when exposed to near-lethal levels of water temperature (Johnson and Evens 1996; Malloy and Targett 1991). Colder temperatures decrease the metabolic rates of fish, leading to lower activity levels and reduced degrees of responsiveness (Sogard and Olla 2000). Possible reasons for reduced survivability of smaller fish include reduced energy storage and lower lipid content as compared to larger fish (Henderson et al. 1988; Thompson et al. 1991; Umino et al. 1991; Griffiths and Kirkwood 1995; Schultz and Conover 1997). The lower lipid content may result in the faster exhaustion of reserves due to varying metabolic rates of smaller fish (Miranda and Hubbard 1994; Paloheimo and Dickie 1966). However, regardless of fish size, their tolerance of extreme cold temperatures can be exceeded, resulting in osmoregulatory failure (Sogard and Olla 2000). As previously noted, most of the target species utilized the study site mangroves as nursery habitats, with a majority of the abundances comprised of Age 0 fish. However, given the larger comparative size of great barracuda at Age 1 and the composition of gray snapper at both mangrove locations consisting of Age 0-2 fish, both of these species may have experienced lesser effects from their larger sizes in terms of reduced susceptibility to the cold event.

The additional examination of seasonal length-frequency distributions for gray snapper from 2009-2011 revealed an interesting trend in cohort movement following the January 2010 cold spell (Figures 6a-l). At both mainland and leeward key locations, new cohorts of juvenile fish appear to show up during the wet seasons, whereas the dry seasons typically did not see new cohorts. Rather, the largest sized cohort would no longer show up during the dry season, suggesting an ontogenetic shift to the adjacent coral reefs at this time. At mainland locations, the 2010 dry season was dominated by what would have been cohort #4. However, given the small incremental increase in lengths from cohort #4 during the 2009 wet season, this suggests either a period of reduced growth rates among the surviving individuals or the immigration of displaced fish from other locations. Also, cohort #2 was no longer present during the 2010 dry season, most likely suggesting the early migration from the shallow mangrove waters to the deeper waters of the adjacent reefs. The following 2010 wet season again saw the recruitment of a new cohort (#5), while cohort #4 seemed to experience reduced growth rates.

At leeward key locations, no fish were present during the 2010 dry season. However, the following 2010 wet season saw what would have been cohort #3 return to the area along with two new cohorts of smaller juveniles. It is unknown whether these individuals were the same fish that made up cohort #3 during the 2009 wet season. It is most likely that the cohorts seen during the 2010 wet season immigrated to the area due to the space availability following the mortalities of the cold spell and for foraging purposes on new juvenile recruits of other species, considering the partial piscivorous diet of gray snapper (Hettler 1989). At this location, early migration may have taken place during the 2011 dry season for cohort #4. At both mangrove locations, similarly distributed length-frequency cohorts were present during the 2011 wet season in comparison to the 2009 wet season, which was the last season prior to the cold spell.

The alterations on the predicted cohorts in terms of size and migration time may adjust the influx of adult recruits onto the nearby coral reefs. If extensive mortalities occurred and/or cohort growth was slowed due to the cold-water temperatures, the number of adults on the reefs would be diminished or smaller in size. Based on the conducted analyses and the examination of size structure plots, it is likely that the reefs did indeed see decreased populations of the five target species in the seasons following the January 2010 cold spell. If allowed to recover uninhibited, the populations at the reefs would most likely rebound after several years of new juvenile recruitment to the mangrove nursery habitats. However, if commercial and recreational fishing continued at the same rates as before the event, the species targeted by these industries may have a more difficult time rebounding.

The current data set was part of a larger database gathering longitudinal observational data of multiple species from multiple locations. Observations were gathered with underwater visual fish surveys, a quick and effective method for gathering distribution, abundance, and size structure data for quantitative analyses (Faunce and Serafy 2006). Limitations to such surveys do exist due to various threats to internal validity such as varying levels of visibility, potential fish responses to those performing the data collection, subjectivity in observer experience, recording errors, and difficulties in observing cryptic species (Chael and Thompson 1997; Thompson and Mapstone 1997; Ley et al. 1999; Eggleston et al. 2004). Although there are ways to reduce such problems (e.g., training observers on standardized methods for observer recordings), other methods such as tagging, genetic studies, and otolith chemistry could, in conjunction with visual surveys, result in strengthening non-spurious findings. Also, when collecting data at the species-specific level, records of abundance tend to be positively skewed and dominated by zero values (Lo et al. 1992), making conventional parametric statistical analyses inappropriate (Serafy et al. 2007).

An additional challenge is operationalizing cold spell terminology. The review of the current literature on cold spells found no agreed upon definition for what constitutes a cold spell. For the purposes of the review, any designation of a cold spell by an author was accepted. It is suggested that moving forward, a functional definition for cold spells be adopted from a managerial perspective. The definition used in this study to examine the YSI water temperature data could be used as a starting point on which to build a complete description. This definition would be based on the resulting drop in water temperatures at various locations inhabited by target species of interest and/or economical or ecological importance on a monthly average basis. Such a designation, combined with life history and thermal tolerance information for the identified species, would aid in computing projected mortalities among various age-classes at specific locations and subsequently, the populations overall. This would lead to varying levels of alterations to fishery limits following a cold spell event, preventing additional stress on an already damaged stock.

Visual surveys for the mangrove shoreline fish monitoring data set have recently focused primarily on mainland locations, accounting for roughly 85% of survey sites (Teare 2014). Given the results from this and other studies, namely the high presence and abundance values at the leeward key locations (Serafy et al. 2003; Faunce and Serafy 2008b) and the determination that these mangroves on the eastern shorelines of Biscayne Bay are most likely more suitable as nursery habitats for economically and ecologically important coral reef fish species, one suggestion is a more equal distribution of future survey conduction. This will provide further information on the critical leeward key mangrove habitats without neglecting the larger area of the mainland shorelines, since these coastal locations are comparatively more exposed to the effects of human alterations and the resulting environmental degradation.

Although this study is restricted to the BNP mangrove complex of South Florida, the results may be applied to other areas of similar geographic or temperature ranges and with comparable species. Any locations with species of commercial and recreational importance that suffer cold spells would also benefit from implementing survey techniques similar to those described for the mangrove transects as a means of comparing pre- and post-event data.

Summary and Conclusions

Overall, this study attempts to quantitatively assess the effects of a severe cold spell on mangrove utilizing reef fishes in the BNP study area. Specifically, the January 2010 cold spell was examined with respect to the presence, abundance, and size structure of sergeant major, yellowfin mojarra, schoolmaster snapper, gray snapper, and great barracuda. Additionally, the data were assessed to determine the necessary time needed for occurrence and size variables to return to pre-stress levels.

Several conclusions can be drawn from this work for consideration in future studies: (1) both the presence and abundance of the five species decreased at both mangrove locations and remained at reduced levels for several seasons before beginning to return to pre-stress values; (2) the data suggest the species suffered mortalities rather than emigrating to different locations considering the time taken to return to pre-stress levels; and (3) the various target species examined in this study appear to respond differently to the effects of the January 2010 cold period, meaning episodic cold snaps can be influential in restructuring fish communities.

Although these results provide insight on the expected effects of various fish stocks following a cold spell, they are not adequate on their own to alter fishery limits. A more in-depth analysis of pre- and post-event data, coupled with differentiating potential emigration out of the study area compared to mortality, and examining the adjacent coral reef locations to assess the movement and sizes of various age-classes, would present quantitative variations among older age-classes among stocks of commonly fished species, such as snapper and barracuda. It has previously been suggested that the presence of habitats such as mangroves act as a ‘waiting room’ location for juvenile fishes inhabiting coral reefs at later life stages, buffering the reef populations during years of poor recruitment (Bardach 1959; Parrish 1989). If it is determined that cold spells, such as the January 2010 event, do have a negative overall effect on juvenile mangrove utilizing reef fishes, the detriment of a poor recruitment situation would be magnified, leading to even further stock declines.

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