Fish Community Changes in Marine Reserves

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Abstract

Marine protected areas are a multi-use management tool that is widely being adapted to protect biodiversity. Within them there are strict "no-take" zones where all extraction is prohibited. "No-take" zones, also known as marine reserves, provide an opportunity to apply a more ecosystem and habitat-based approach which further supports biodiversity and ecosystem recovery. This proposal will study coral reef sites in the Florida Keys National Marine Sanctuary that have over 20 years of protection as "no-take" zones to compare changes in the fish community.

Introduction

The state of marine ecosystems is globally in decline due to persistent anthropogenic stressors such as overfishing, pollution, climate change, ocean acidification, habitat destruction, etc. (Donner et al., 2005; Harley et al., 2006; Halpern et al., 2008, IPCC 2011). With the decrease of marine biodiversity also comes the loss of ecosystem functions and services which human communities heavily depend on (Moberg & Folke 1999, Foley et al. 2010, Midgley 2012, Harborne et al. 2017). Marine protected areas (MPA) are used as a management tool to conserve and protect coastal resources, especially vulnerable tropical marine and coastal ecosystems (Wabnitz et al. 2010). They are put in place along with traditional fisheries management strategies, such as catch and size quotas, as a way to reduce fishing pressures and improve or maintain the ecosystem state (Halpern & Warner 2002, Lester et al. 2009, Cheng et al. 2019). MPA's are often multi-use with distinct zonings to allow different activities such as fishing, boating, recreational activity, research, etc. to satisfy all stakeholder needs. This project will analyze reef fish data to graph and project fish recovery curves within "no-take" areas in the Florida Keys National Marine Sanctuary (FKNMS).

The Florida Keys National Marine Sanctuary and Protection Act was signed into law in 1990 to protect 2,900 square nautical miles (9,947 square kilometers) of coastal waters off the Florida Keys (Dept of Comm., NOAA 1996). There are five distinct marine zones: ecological reserves, existing management areas, sanctuary preservation areas, special-use areas, and wildlife management areas (Fig. 1). In 1997, 18 Sanctuary Preservation Areas (SPAs) were established to protect shallow, heavily used reefs from users by prohibiting all consumptive activities such as "Possessing, moving, harvesting, removing, taking, damaging, disturbing, breaking, cutting, spearing, or otherwise injuring any coral, marine invertebrate, fish, bottom formation, algae, seagrass or other living or dead organism, including shells, or attempting any of these activities." (Dept of Comm., NOAA 1997). Ecological Reserves and Special-use Areas, although different zones, have similar level of protection to SPAs, where no fishing or extraction is allowed. This project is focusing on patch reefs with "no-take" designation, from here on called marine reserves.

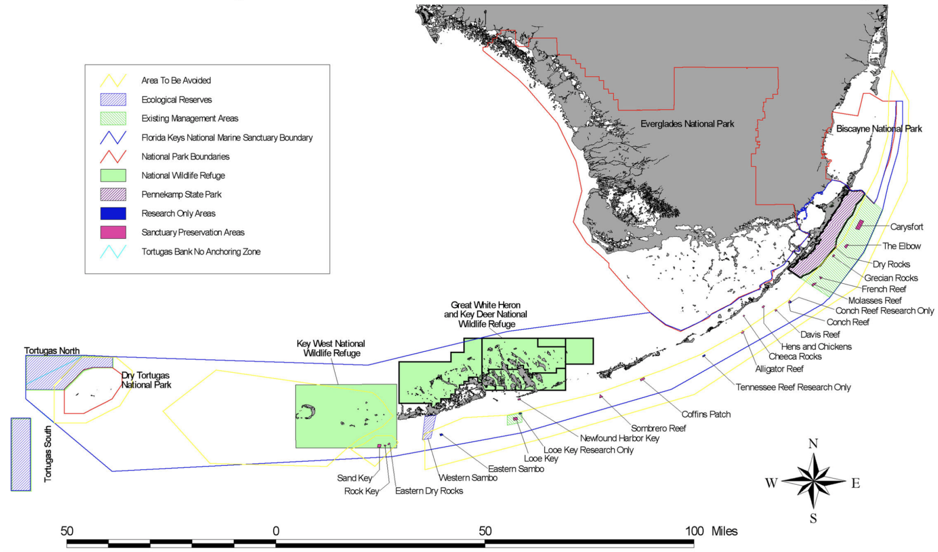

Fig. 1. Florida Keys National Marine Sanctuary and Zones Boundaries (FKNMS 2011).

As the Florida Keys has undergone heavy extraction and exploitation there are multiple agencies working together to conduct annual monitoring of the coral reef ecosystems. This multi-agency collaboration includes National Oceanic Atmospheric Administration (NOAA), National Park Service (NPS), U.S. Department of the Interior, University of Miami, and Florida Fish and Wildlife Conservation Commission for the National Coral Reef Monitoring Program in South Florida. This program focuses on monitoring diverse coral reef ecosystems in three regions labeled: SEFCRI (Southeast Florida Reef Initiative which includes Miami-Dade, Broward, Palm Beach and Martin counties), the Florida Keys, and Dry Tortugas (Fig. 2).

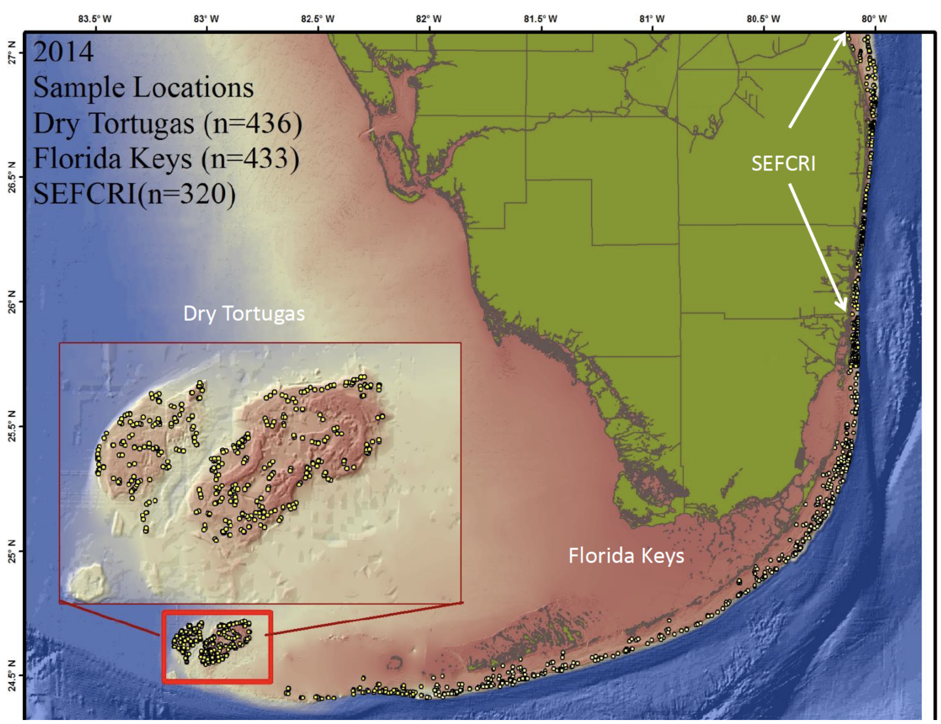

Fig. 2. South Florida regions (SEFCRI, Florida Keys, and Dry Tortugas) within the National Coral Reef Monitoring Program.

Methods

Site Information

Sites within the Florida Keys National Marine Sanctuary focus on hardbottom reef habitats between 1-30m depth. There is greater effort towards sites with higher complexity habitat as these habitats have higher densities of fish and higher variance as well. In this way scientists are able to capture more accurate data on reef fish and still be able to detect changes inside and outside of marine reserves. As our interest for this project is in marine reserves we narrowed our region of interest to only protected sites within the Florida Keys National Marine Sanctuary (Fig. 1).

Field Methods

Data collected in the FKNMS include fish sampling and benthic sampling. The fish sampling methods include 4 divers (2 buddy pairs) conducting Reef Visual Census (RVC) point count method in a 15m diameter cylinder (177m^2) to record the total number of all fish species present and the minimum, maximum and mean fork length of all individuals within 15 minutes. Dives are only conducted if diver visibility is greater than 5m. Additional data is collected on the benthos often by a separate team per site to record data such as coral cover, rugosity measurements, and additional biotic and abiotic measurements. This "modern" survey method began in 2001 therefore for this analysis we chose data from 2001-2018, although visual surveys of reef fish species have been conducted since 1978 and is available online dating back since 1999.

Statistical Analysis

The South Florida Reef Visual Census data is publicly available by NOAA at:
`https://github.com/jeremiaheb/rvc`

We started by pulling the data from the GitHub and began filtering it to include data that we were looking to analyze. We filtered from years 2001-2018, although the Florida Keys region was surveyed every other year after 2012, so data is missing for years 2013, 2015, and 2017. We then filtered to only include sites that are marine reserves ("no-take") and this reduced the dataset by more than half. We did additional filters to include rows with fish length and count greater than 0 as much of the data set was filled with rows on fish species they were hoping to observe, but didn't, and so "length" and "count" columns were filled with 0 as a value.

As fish "length" and "count" alone aren't metrics that managers can use to analyze the fish community, we decided to convert this data to biomass. To do so we needed to calculate the weight length relationship:
`W(g) = a*L(mm)^b`

for each fish species using the 'a' and 'b' coefficients that are unique to each fish species. This data was pulled from the taxonomic dataset also from the above GitHub, which is based on fish literature (Marks & Klomp 2003). Data sets were merged and the biomass column was created for each fish species. After, we could subset the data by fish families of interest and continue to apply statistical methods to plot fish biomass curves to see if there were changes as the number of years of protection increased.

Results

To test if marine reserves are working efficiently managers look towards data to show if goals are being met. Marine reserve goals can include increasing species biomass, increasing species richness, increasing the abundance of mature fish, assisting population connectivity, restoring ecological function, etc. Therefore, we first plotted how species richness had changed over time. To do this, we had to create a new data frame of only unique fish species over time, and plot cumulative fish species per year (Fig. 3).

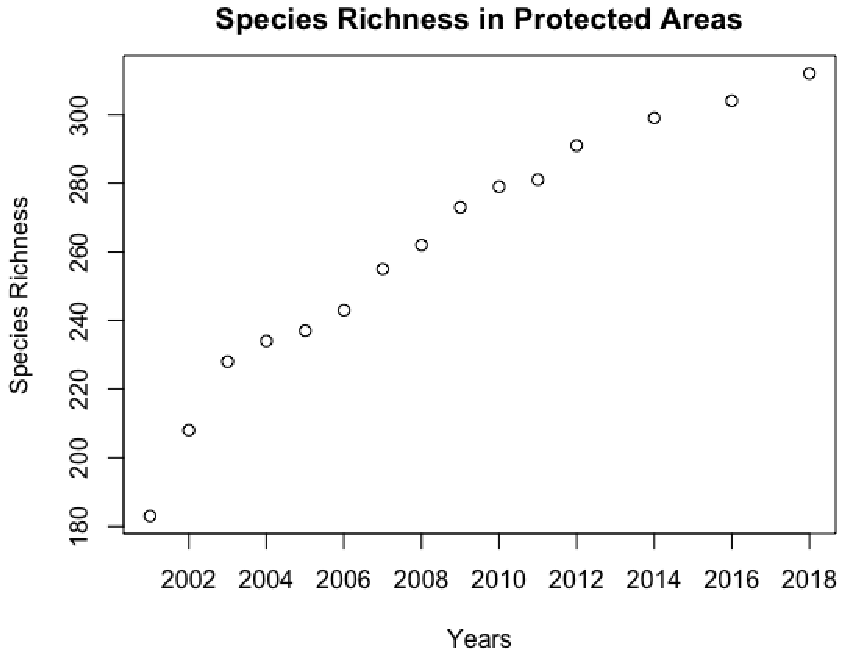

Fig. 3. Fish species richness in Florida Keys protected areas between 2001-2018.

In 2001 the total number of fish species started at 183 and increased every year with 312 total fish species observed in 2018. From 2001 to 2002 there was an additional 25 new fish species observed and from 2016 to 2018 (2017 year skipped in monitoring) there were 3 additional fish species observed, so these data points are not following a linear trend. The data seems to be following an asymptotic curve and we would expect this as the number of years increases in a marine reserve it is not realistic for an infinite number of new fish species to be colonizing new areas; but as these sites were once heavily exploited now fish species have fishing pressures removed and are able to move to new areas as populations increase.

After plotting our species richness curve, we wanted to see if there could also be a positive trend with fish biomass. To do this we applied an ARIMA (Auto Regressive Integrated Moving Average) time series analysis approach as we have data spanning 15 years in protected areas. We began by subsetting our data to a commercially important fish family, groupers, as we believed this may show significant changes and/or trends, as groupers were once heavily exploited in the Florida Keys. The goliath grouper (*Epinephelus itajara*) is a protected species and other grouper species such as the black grouper (*Mycteroperca bonaci*) have annual closed seasons, usually during spawning so as to not negatively affect their populations. The grouper family, Serranidae, includes many genera that were observed in all years in protected areas. Plotting the biomass in kg over time for Serranidae showed biomass was not increasing over time as expected, and there could be many reasons for this (Fig. 4).

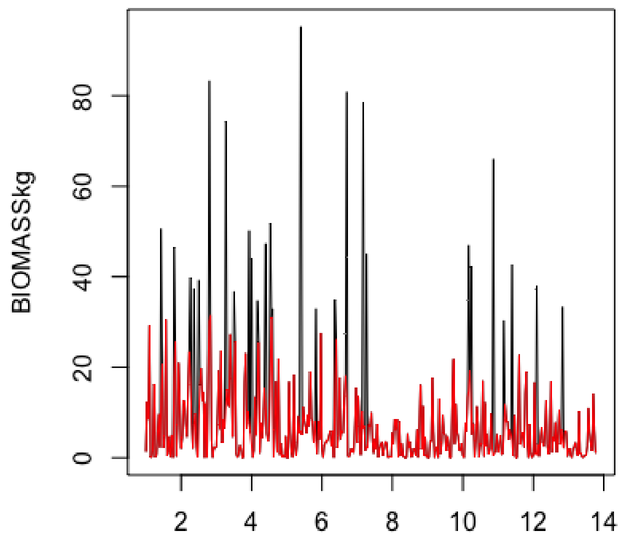

Fig. 4. Serranidae biomass (kg) plotted over time with the ‘tsclean’ function layering over the original data to contrast outliers.

Decomposing the time series to separate the data into the season, trend and irregular components graphically showed that there is a decreasing trend in the Serranidae biomass data that is not seasonal (Fig. 5).

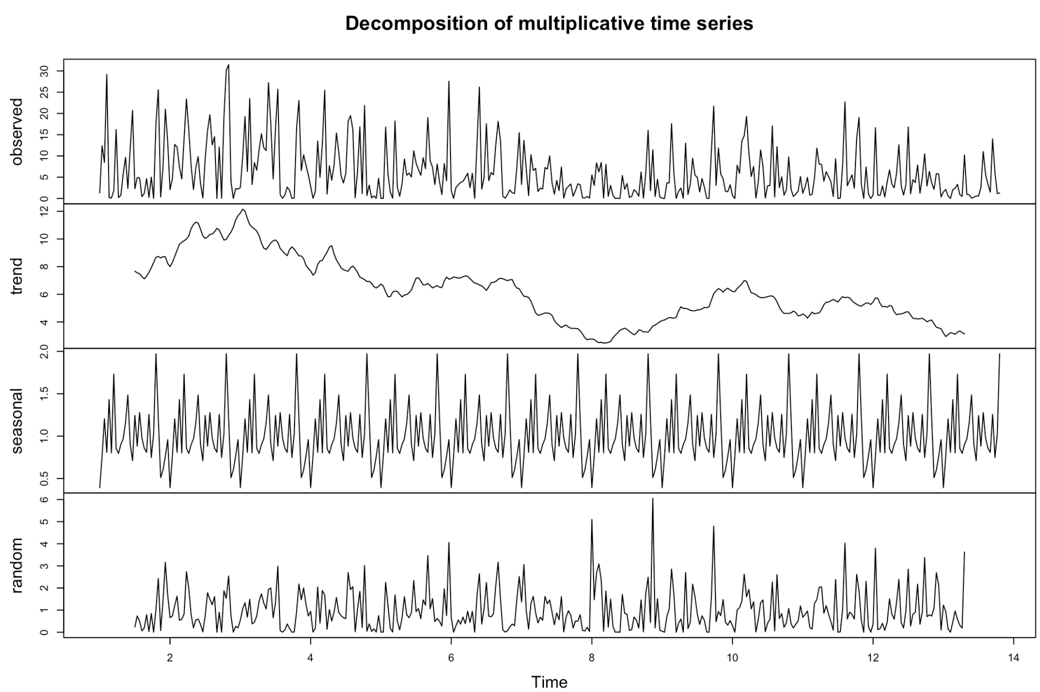

Fig. 5. Grouper family decomposition of time series analysis data into the season, trend and random components.

Outliers or residuals are clearly shown in the graphs. To make sure our data meets our assumptions to run an ARIMA we test for stationary. The series is considered stationary when its mean, variance, and autocovariance are time invariant, which we do with the Dickey-Fuller ADF test (Table 1). Our data had a p-value less than 0.01 which is less than 0.05 and therefore is stationary. Additional tests on our data set were conducted such as the Ljung-Box, a test of independence at all lags, testing for the overall randomness based on a number of lags. Our p-value of 0.2029 was greater than 0.05 and therefore our ARIMA model had no autocorrelation.

Augmented Dickey-Fuller Test		
Dickey-Fuller	Lag order	p-value
-6.4821	7	0.01
Warning message: p-value smaller than printed p-value		

Table 1. Testing for stationarity by the Augmented Dickey-Fuller Test of Serranidae biomass (kg) data.

We fit two ARIMA models, the second where we adjusted the lags of our data to see which would fit better. To test if we had a good model fit for our data, and compare the fits, we use the Akaike Information Criteria (AIC), this quantifies the goodness of fit of the model and the simplicity of the model, by estimating the amount of information lost by a model. The lower the AIC, the better fit for the data. Our second model had a better AIC.

Discussion

We had hypothesized for how we expected the fish community to change with respect to the number of years a marine reserve existed. If a marine reserve is being properly enforced, generally, we would expect that our management is improving the system and moving towards our ecological goals of increasing species biomass, increasing species richness, increasing the abundance of mature fish, assisting population connectivity, restoring ecological function, etc. Fig. 3 showed that we are meeting one of these metrics and species richness was increasing over time, although with a trend that there will be a year where this will no longer continue to increase. We then picked an important harvested fish family, Serranidae, as groupers are heavily harvested for recreational and commercial fisheries. Our ARIMA showed the trends in our biomass data and how this was surprisingly, decreasing over time. Further analysis would need to be conducted to find which explanatory variable could best explain the decrease. One hypothesis could be that although years of protection has increased, the state of our ecosystems has decreased due to chronic global stressors. Therefore, the state of our ecosystems has continued to decrease and this will add stress to fisheries. In Florida, USA the total tourism value is $1,156.8 million per year, and globally reef tourism is calculated around US$35.8 billion dollars every year (Spalding et al. 2017). There is a lot at stake to successfully manage our coastal resources and implementing a strategic ecological design will support our ecological goals of protecting, restoring and sustaining marine biodiversity (Gell & Roberts, 2003; Lester et al., 2009; Edgar et al., 2014).

Although marine reserves can vastly benefit different ecosystems and the species within them, they alone will not undo all anthropogenic stressors occurring within the systems. Global stressors such as climate change and sea level rise are the most significant threats to the survival of coral reefs and the species within them. With unpredictable weather patterns and increasing ocean temperatures the speed and direction of major currents are being changed and this will affect the physical transport of larvae and propagules (Cowen & Sponaugle 2009). Scientists, managers, and political leaders must come together and take aggressive action to reduce global carbon emissions if we hope to continue to have healthy coral reefs and benefit from the vital ecosystem services they provide (Moberg & Folke 1999, Bruno et al. 2019).

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