Greater Amberjack *Seriola dumerili* Findings from the NMFS Panama City Laboratory Camera & Trap Fishery-Independent Survey 2006-2018

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SEDAR70-WP-12

2 July 2020

Please cite this document as:

Overly, K.E., C.L. Gardner and A.G. Pollack. 2020. Greater Amberjack *Seriola dumerili* Findings from the NMFS Panama City Laboratory Camera & Trap Fishery-Independent Survey 2006-2018. SEDAR70-WP-12. SEDAR, North Charleston, SC. 11 pp.

Greater Amberjack *Seriola dumerili* Findings from the NMFS Panama City Laboratory Camera & Trap Fishery-Independent Survey 2006-2018

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July 2020

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Contribution 20-**

Survey history and overview

In 2002, the Panama City NMFS lab began development of a fishery-independent trap survey (PC survey) of natural reefs on the inner shelf in the northeast Gulf of Mexico, off Panama City, FL. The primary objective of the PC survey was establishing an age-based annual index of abundance for young (age 0-3), pre-recruit gag, scamp, and red grouper. Secondary objectives included examining regional catch, recruitment, demographic, and distribution patterns of other exploited reef fish species. Initially, the PC survey used the same chevron trap configuration and soak time that has been used by the South Atlantic MARMAP program for over 30 years (McGovern et. al. 1998), as traps are efficient at capturing a broad size range of several species of reef fish (Nelson et. al.1982, Collins 1990). However, an in-house study in 2003 indicated that traps with a throat entrance area 50% smaller than that in the MARMAP traps were much more effective at meeting our objective of capturing sufficient numbers of all three species of grouper. Video data from our study and consultations with fishermen suggested that the presence of larger red grouper in a trap tend to deter other species from entering. Beginning in 2004, the 50% trap throat size became the standard. That same year the survey was expanded east of Panama City to Apalachee Bay off the Big Bend region of Florida (Fig. 1), an area separated from the shelf off Panama City by Cape San Blas - an established hydrographic and likely zoogeographic boundary (Zieman and Zieman 1989).

Beginning in 2005, the collection of visual (stationary video) data was added to the survey to provide insight on trap selectivity, more complete information on community structure, relative abundance estimates on species rarely or never caught in the trap, and additional, independent estimates of abundance on species typically caught in the traps. Video sampling was only completed in Apalachee Bay in 2005, but was expanded to the entire survey in 2006. Additionally, the target species list was expanded to include the other exploited reef fishes common in the survey area, i.e., red, vermilion, gray, and lane snapper; gray triggerfish, red porgy, white grunt, black seabass, greater amberjack, and hogfish in 2005. From 2005 through 2008 each site was sampled with the camera array, directly followed by a single trap. Beginning in 2009, trap effort was reduced ~50%, with one deployed at every other video site. This was done to increase the number of video samples, and thereby the accuracy and precision of the video abundance estimates. Camera arrays are much less selective and provide abundance estimates for many more species than traps, and those estimates are usually much less biased (DeVries et al. 2009). At each site, a CTD cast was made to collect temperature, salinity, oxygen, and turbidity profiles.

Through 2009, sampling was systematic because of a very limited sampling universe. In 2010, the design was changed to 2-stage unequal probability sampling design after side scan sonar surveys that year yielded an order of magnitude increase in the sampling universe. Five by five minute blocks known to contain hard bottom reef sites, and proportionally allocated by region, sub-region, and depth (10-20, 20-30, 30+ m) to ensure uniform geographic and bathymetric coverage, are randomly selected first. Then, two known reef sites, a minimum of 250 m apart within each selected block are randomly selected. Alternates are also selected for use and are utilized when another boat is found to be fishing the selected site or no hard bottom can be found with sonar at the designated location.

Depth coverage was 8-30 m during 2004-07 and steadily expanded to 8-52 m in 2008. The coverage was expanded again in 2017 and now ranges from 7-58 m. Sampling effort has also increased since 2004 with a minimum of 59 and maximum of 186 video samples per year. All sampling has occurred between May and November, but primarily during June through August.

Methods

Sampling was conducted during the daytime from one hr after sunrise until one hr before sunset. Chevron traps were baited each new drop, with three previously frozen Atlantic mackerel *Scomber scombrus,* and soaked for 1 to 1.5 hr. Traps were dropped as close as possible to the exact location sampled by the camera array. All trap-caught fish were identified, counted, and measured to maximum total (TL) and fork length (FL) (FL only for gray triggerfish and TL only for black seabass). Both sagittal otoliths were collected from a max of five randomly subsampled specimens of snappers (gray, lane, red, and vermilion), groupers (gag, red, and scamp), black seabass, red porgy, hogfish, white grunt, and gray triggerfish (first dorsal spine for the latter).

Visual data were collected using a stationary camera array composed of four Hi 8 video cameras (2005 only) or four high definition (HD) digital video cameras (2006-2008) mounted orthogonally 30 cm above the bottom of an aluminum frame. From 2007 until 2009, parallel lasers (100 mm spacing) mounted above and below each camera were used to estimate the sizes of fish which crossed the field of view perpendicular to the camera. In 2009 and 2010, one of the HD cameras was replaced with a stereo imaging system (SIS) consisting of two high resolution black and white still cameras mounted 8 cm apart, one digital video (MPEG) color camera, and a computer to automatically control these cameras as well as store the data. The SIS provides images from which fish measurements can be obtained with the Vision Measurement System (VMS) software (2009-2014) and SeaGIS software (2015-2017). Beginning in 2011, a second SIS facing 180º from the other was added, reducing the number of HDs to two; and both SIS's were also upgraded with HD, color MPEG cameras. In 2012 the two digital video cameras were replaced with HD GoPro cameras. The camera array was unbaited in 2005 through 2008, but since 2009 has been freshly baited each drop with one previously frozen Atlantic mackerel placed in a mesh bag near the center.

Before stereo camera systems were used (prior to 2009), soak time for the array was 30 min to allow sediment stirred up during camera deployment to dissipate and ensure tapes with an un-occluded view of at least 20 min duration (Gledhill and David 2003). With the addition of stereo cameras in 2009, soak time was increased to 45 min to allow sufficient time for the SIS to be settled on the bottom before starting its hard drive, and to insure the hard drive had time to shut down before retrieval. In mid-2013, stereo cameras were upgraded with solid state hard drives, enabling soak time to be reduced back to 30 min. Prior to 2009, tapes of the four HD cameras were scanned, and the one with the best view of the habitat was analyzed in detail. If none was obviously better, one was randomly chosen. In 2009 only the three HD video cameras were scanned and the one with the best view of the reef was analyzed. Starting in 2010, all four cameras – the HDs and the SIS MPEGs, which have virtually the same fields of view (64 vs 65º), were scanned, and again, the one with the best view of the habitat was analyzed. Beginning in 2012, when a video from a GoPro camera was selected to be read, predetermined, equal portions of each edge of the video were digitally cropped so that only the central 65° of the field of view was visible due to the GoPro's much larger field of view (122 vs 65º). The videos were viewed, beginning twenty minutes prior to pick up of the camera array, to ensure the cloud of sediment disturbed by the landing of the array had dissipated. All fish captured on videotape and identifiable to at least genus were counted. Data on habitat type and reef morphometrics were also recorded. If the quality of the MPEG video derived from the SIS was less than desirable, fish identifications were confirmed on the higher quality and concurrent stereo still frames. The estimator of abundance was the maximum number of a given species in the field of view at any time during the 20 min analyzed (= min count; Gledhill and Ingram 2004, or MaxN; Ellis and DeMartini 1995). Stereo measurements were taken from a still frame showing the min count of a given species (but not necessarily the same frame the actual min count came from) to eliminate the possibility of measuring the same fish more than once. Even for deployments where the SIS did not provide a good view of the reef habitat, the stereo files were examined to obtain fish measurements using VMS or SeaGIS, and again, those measurements were only taken from a still frame showing the min count of a given species. In contrast, when scaling lasers were used to obtain length data, there was no way to eliminate the possibility of double measuring a given fish, although this was probably not a serious problem, as usable laser hits were typically rare for any one sample.

Because of the significant differences we observed in both species composition and abundance of many reef fishes east and west of Cape San Blas, and because of the Cape's known status as a hydrographic and likely zoogeographic boundary (Zieman and Zieman 1989), many of the results presented herein are shown separately for the two areas.

Censored data sets were used in deriving the indices of relative abundance from video data. All video samples were screened, and those with no visible hard or live bottom and no visible species of fish strongly associated with hard bottom habitat, as well as samples where the view was obscured because of poor visibility, video out of focus, etc., were excluded from calculations of relative abundance. In 2014, ten video samples from an area with an ongoing red tide bloom which reduced visibility past a readable threshold were also censored.

The CPUE and proportion positive findings for the trap survey were based on all samples except those from sites which had already been sampled in a given year and ten sites in 2014 located in an ongoing red tide bloom that greatly reduced visibility.

Index Construction

Delta-lognormal modeling methods were used to estimate relative abundance indices for greater amberjack (Pennington 1983, Bradu and Mundlak 1970). The main advantage of using this method is allowance for the probability of zero catch (Ortiz *et al.* 2000). The index computed by this method is a mathematical combination of yearly abundance estimates from two distinct generalized linear models: a binomial (logistic) model which describes proportion of positive abundance values (i.e. presence/absence) and a lognormal model which describes variability in only the nonzero abundance data (*cf.* Lo *et al*. 1992).

The delta-lognormal index of relative abundance (I_y) was estimated as:

(1) $I_y = c_y p_y$,

where c_y is the estimate of mean CPUE for positive catches only for year y, and p_y is the estimate of mean probability of occurrence during year y. Both c_y and p_y were estimated using generalized linear models. Data used to estimate abundance for positive catches (c) and probability of occurrence (p) were assumed to have a lognormal distribution and a binomial distribution, respectively, and modeled using the following equations:

(2) $\ln(c) = X\beta + \varepsilon$

and

(3) $p = \dfrac{e^{\mathbf{X\beta+\varepsilon}}}{1+e^{\mathbf{X\beta+\varepsilon}}}$,

respectively, where c is a vector of the positive catch data, p is a vector of the presence/absence data, X is the design matrix for main effects, β is the parameter vector for main effects, and ε is a vector of independent normally distributed errors with expectation zero and variance σ^2. Therefore, c_y and p_y were estimated as least-squares means for each year along with their corresponding standard errors, SE (c_y) and SE (p_y), respectively. From these estimates, I_y was calculated, as in equation (1), and its variance calculated using the delta method approximation

(4) $V(I_y) \approx V(c_y)p_y^2 + c_y^2 V(p_y)$.

A covariance term is not included in the variance estimator since there is no correlation between the estimator of the proportion positive and the mean CPUE given presence. The two estimators are derived independently and have been shown to not covary for a given year (Christman, unpublished).

The submodels of the delta-lognormal model were built using a backward selection procedure based on type 3 analyses with an inclusion level of significance of $\alpha = 0.05$. Binomial submodel performance was evaluated using AIC, while the performance of the lognormal submodel was evaluated based on analyses of residual scatter and QQ plots in addition to AIC. Variables that could be included in the submodels were:

Submodel Variables

Year: 2006 – 2018
Depth: 6 – 58 meters (continuous)
Month: May, June, July, August, September, October, November
Region: East of Cape San Blas, West of Cape San Blas

Results and Discussion

Index of Abundance

For the PC Video Survey abundance index of greater amberjack, year, depth, month, and region were retained in the binomial submodel, while year and region were retained in the lognormal submodel. A summary of the factors used in the analysis is presented in Appendix Table 1. Table 2 summarizes the final set of variables used in the submodels and their significance. The AIC for the binomial and lognormal submodels were 8043.0 and 799.3, respectively. The diagnostic plots for the lognormal submodel are shown in Figure 3, and indicated the distribution of the residuals is approximately normal. Annual abundance indices are presented in Table 2 and Figure 4.

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Table 1. Summary of backward selection procedure for building delta-lognormal submodels for greater amberjack Panama City Video Survey index of relative abundance from 2006 to 2018 in the eastern Gulf of Mexico.

Model Run #1	*Binomial Submodel Type 3 Tests (AIC 8043.0)*						*Lognormal Submodel Type 3 Tests (AIC 811.5)*			
Effect	*Num DF*	*Den DF*	*Chi-Square*	*F Value*	*Pr > ChiSq*	*Pr > F*	*Num DF*	*Den DF*	*F Value*	*Pr > F*
Year	12	1606	69.78	5.82	<.0001	<.0001	12	294	2.17	0.0133
Depth	1	1606	9.58	9.58	0.0020	0.0020	1	294	0.32	0.5732
Month	6	1606	30.62	5.10	<.0001	<.0001	6	294	0.72	0.6374
Region	1	1606	13.44	13.44	0.0002	0.0003	1	294	5.80	0.0166
Model Run #2	*Binomial Submodel Type 3 Tests (AIC 8043.0)*						*Lognormal Submodel Type 3 Tests (AIC 806.7)*			
Effect	*Num DF*	*Den DF*	*Chi-Square*	*F Value*	*Pr > ChiSq*	*Pr > F*	*Num DF*	*Den DF*	*F Value*	*Pr > F*
Year	12	1606	69.78	5.82	<.0001	<.0001	12	300	2.26	0.0094
Depth	1	1606	9.58	9.58	0.0020	0.0020	1	300	0.70	0.4043
Month	6	1606	30.62	5.10	<.0001	<.0001		Dropped		
Region	1	1606	13.44	13.44	0.0002	0.0003	1	300	8.24	0.0044
Model Run #3	*Binomial Submodel Type 3 Tests (AIC 8043.0)*						*Lognormal Submodel Type 3 Tests (AIC 799.3)*			
Effect	*Num DF*	*Den DF*	*Chi-Square*	*F Value*	*Pr > ChiSq*	*Pr > F*	*Num DF*	*Den DF*	*F Value*	*Pr > F*
Year	12	1606	69.78	5.82	<.0001	<.0001	12	301	2.32	0.0075
Depth	1	1606	9.58	9.58	0.0020	0.0020		Dropped		
Month	6	1606	30.62	5.10	<.0001	<.0001		Dropped		
Region	1	1606	13.44	13.44	0.0002	0.0003	1	301	11.31	0.0009

Table 2. Indices of greater amberjack abundance developed using the delta-lognormal (DL) model for Panama City Video Survey from 2006-2018 in the eastern Gulf of Mexico. The nominal frequency of occurrence, the number of samples (*N*), the DL Index (number per trawl-hour), the DL indices scaled to a mean of one for the time series, the coefficient of variation on the mean (CV), and lower and upper confidence limits (LCL and UCL) for the scaled index are listed.

Survey Year	Frequency	*N*	DL Index	Scaled Index	CV	LCL	UCL
2006	0.10959	73	0.88634	0.99107	0.44587	0.42291	2.32257
2007	0.21154	52	0.95136	1.06378	0.36569	0.52376	2.16060
2008	0.15294	85	1.16252	1.29990	0.34361	0.66641	2.53556
2009	0.35577	104	1.58011	1.76683	0.19313	1.20497	2.59068
2010	0.17007	147	0.56986	0.63720	0.26343	0.37957	1.06971
2011	0.06962	158	0.15516	0.17349	0.39003	0.08174	0.36824
2012	0.29333	150	1.31707	1.47270	0.17839	1.03366	2.09823
2013	0.35577	104	1.09981	1.22977	0.20218	0.82408	1.83519
2014	0.24390	164	0.73203	0.81853	0.19898	0.55193	1.21392
2015	0.20833	168	0.95662	1.06966	0.21570	0.69827	1.63861
2016	0.15205	171	0.62147	0.69491	0.26353	0.41387	1.16680
2017	0.13333	150	0.62468	0.69850	0.32968	0.36742	1.32792
2018	0.07921	101	0.96913	1.08365	0.43043	0.47510	2.47164

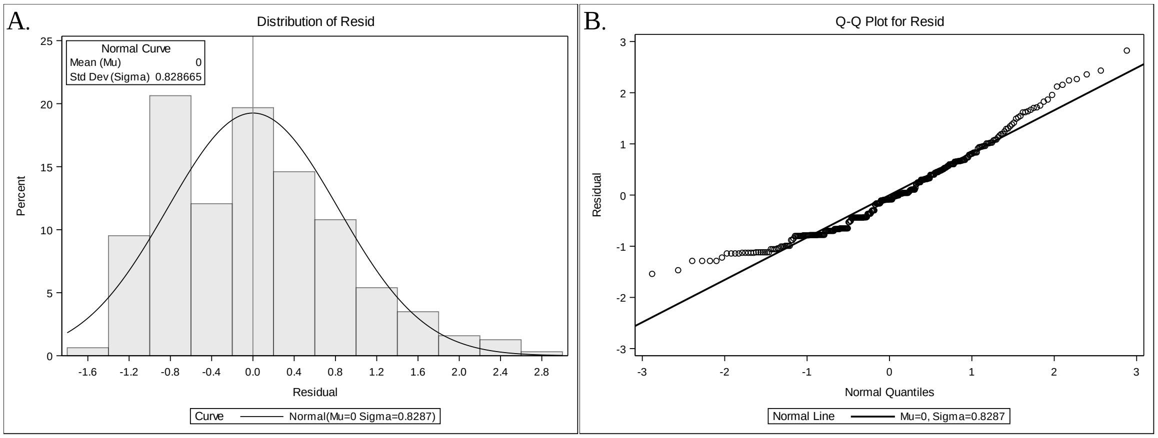

Figure 3. Diagnostic plots for lognormal component of greater amberjack Panama City Video Survey model: **A.** the frequency distribution of log (CPUE) on positive stations and **B.** the cumulative normalized residuals (QQ plot).

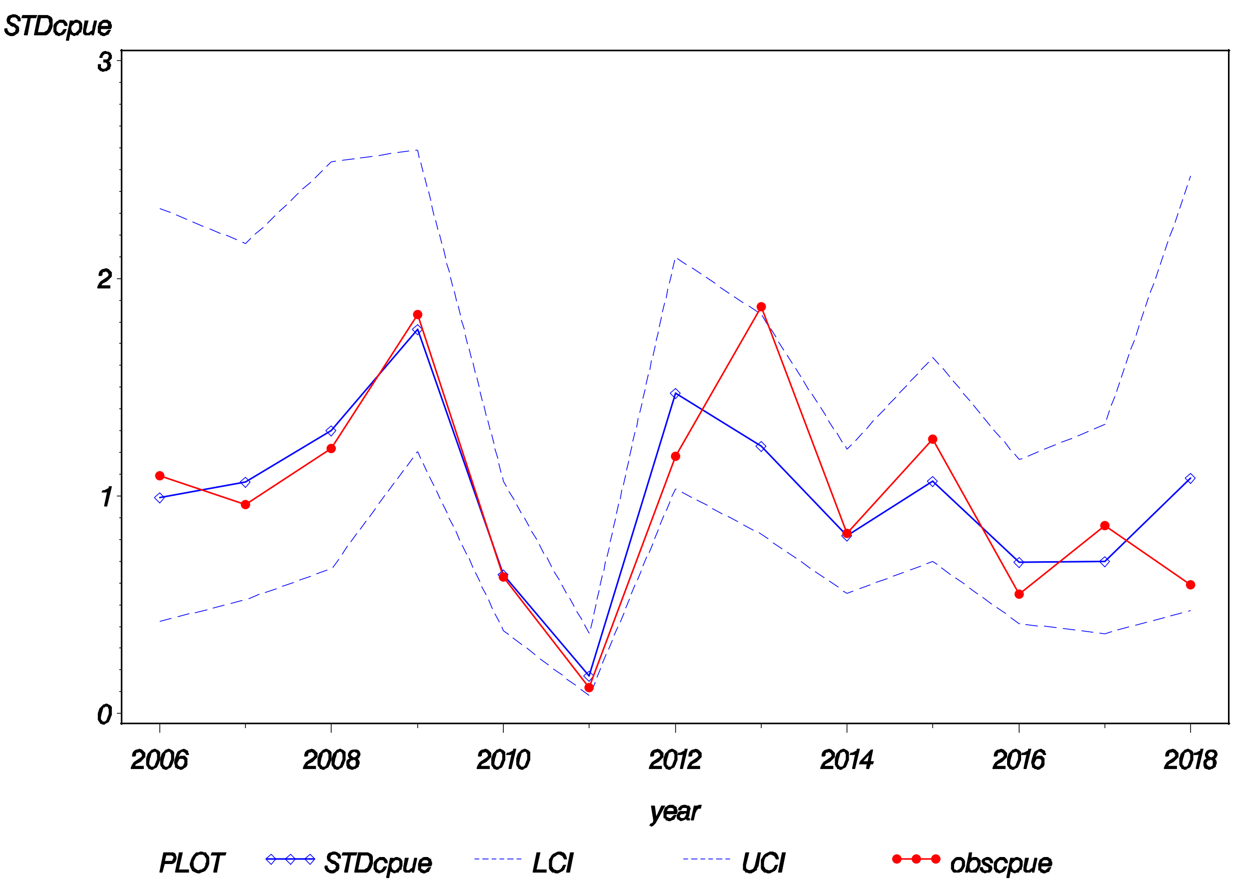

Figure 4. Annual index of abundance for greater amberjack from the Panama City Video Survey from 2006 – 2018.