Greater Amberjack *Seriola dumerili* Findings from the NMFS Panama City Laboratory Trap & Camera Fishery-Independent Survey – 2004-2012

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Panama City Laboratory
Contribution 13-3

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 of the eastern Gulf of Mexico off Panama City, FL, with the primary objective of 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. The chevron trap is efficient at capturing a broad size range of several species of reef fish (Nelson et. al.1982, Collins 1990), and has been used by the South Atlantic MARMAP program for over 20 yr (McGovern et. al. 1998). Initially the PC survey used the same trap configuration and soak time used by MARMAP (McGovern et. al. 1998), but 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 tended 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 (Figure 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 done in Apalachee Bay that first year, but was expanded to the entire survey in 2006. Also in 2005 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, and hogfish. From 2005 through 2008 each site was sampled with the camera array followed immediately by a single trap. Beginning in 2009 trap effort was reduced ~50%, with one deployed at about every other video site, starting with the first site of the day. This was done so the number of video samples, and thereby the accuracy and precision of the video abundance estimates, could be increased. 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). All sampling has occurred between May and early October, but primarily during June through September. At each site, a CTD cast was made to collect temperature, salinity, oxygen, and turbidity profiles.

The survey sampling design was systematic through 2009 because of a very limited sample site universe. In 2010 the design was changed to 2 stage random after side scan sonar surveys that year yielded an order of magnitude increase in that universe. Five by five minute blocks known to contain reef sites, and proportionally allocated by region, subregion, and depth (10-20, 20-30, 30+ m) to ensure uniform geographic and bathymetric coverage, are randomly chosen first. Then 2 known reef sites a minimum of 300 m apart within each selected block are randomly selected (Figure 2). Alternates are also selected (if another boat was fishing the site or no hard bottom could be seen with sonar).

Depth coverage was ~8-30 m during 2004-07, and since then was steadily expanded to ~8 – 47 m (Fig. 3). Sampling effort has also increased since 2004. Sample sizes were 59 in 2004 (33 W: 26 E), 101 in '05 (24 W: 77 E), 113 in '06 (24 W: 89 E), 86 in '07 (29 W: 57 E), 98 in '08 (32 W: 66 E), 143 in '09 (47 W: 96 E), 162 in '10 (53 W: 109 E), 180 in '11 (65 W: 115 E), and 178 in '12 (61 W: 117 E). In 2004 and 2005 some sites were sampled twice: 9 in 04 and 23 in 05; thereafter each site was only sampled once in a given year.

Methods

Sampling is conducted only during daytime from 1 hr after sunrise until 1 hr before sunset. Chevron traps, identical to that used in the MARMAP program (McGovern et al. 1998) except for a 50% smaller throat opening, are baited each set with 3 previously frozen Atlantic mackerel *Scomber scombrus*, and soaked for 1.5 hr. Traps are fished as close as possible to the exact location sampled by the camera array that day. All trap-caught fish are identified, counted and measured to maximum total and fork length (FL only for gray triggerfish and TL only for black seabass). Both sagittal otoliths are collected from 4-5 randomly subsampled specimens of all 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 4 High 8 video cameras (2005) or 4 high definition (HDEF), digital video cameras (2006-08) mounted orthogonally 30 cm above the bottom of an aluminum frame. From 2007 to 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 HDEF 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. Beginning in 2011, a second SIS facing 180º from the other SIS was added, reducing the number of HDEFs to two; and both SIS's were also upgraded with HDEF, color mpeg cameras. The camera array was unbaited 2005-2008. Since 2009 the array 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 unoccluded 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. Prior to 2009, tapes of the 4 HDEF cameras were scanned, with the one with the best view of the habitat analyzed in detail. If none was obviously better, one was randomly chosen. In 2009 only the 3 HDEF video cameras were scanned and the one with the best view of the reef was analyzed. Starting in 2010, all 4 cameras – the HDEFs 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. Twenty min of the tape were viewed, beginning when the cloud of sediment disturbed by the landing of the array has dissipated. All fish captured on videotape were identified to the lowest discernable taxon. 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 (a common problem), fish identifications were confirmed on the much 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), and VMS measurements were only taken from a still frame showing the min count of a given species 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 files were examined to obtain fish measurements using VMS, and again, those measurements were only taken from a still frame showing the min count of a given species. In contrast, when using the scaling lasers on the array 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 in both species composition and abundance for many reef fishes east and west of Cape San Blas, especially in the inner and mid-shelf depths sampled by the Panama City survey, 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. Prior to 2010, the year we began using side scan sonar to locate reefs, lack of knowledge of reef habitat locations east of the Cape necessitated making a much higher proportion of "exploratory" camera and trap drops there versus west of the Cape. To compensate, more overall effort was expended in the east. Some of these "exploratory" sample sites turned out to be sand, mostly sand, or very marginal reef habitat at best, yielding little or no reef fish data. In addition, the gear occasionally missed the intended reef site. Inclusion of data from those sites would have reduced the precision of the abundance estimates and confounded any analyses. For that reason, video data – both habitat classification and fish counts – from all sites were screened, and those with no evidence that hard or live bottom was in close proximity, as well as sites where the view was obscured for some reason (poor visibility, bad camera angle), were censored (excluded) from calculations of relative abundance. As a result of this screening, of video samples east of the Cape, only 31 of 41 in 2005, 47 of 89 in 2006, 23 of 57 in 2007, 56 of 66 in 2008, 62 of 97 in 2009, 95 of 109 in 2010, 99 of 115, in 2011, and 101 of 115 in 2012 met the reef and visibility criteria and were retained. In contrast, west of the Cape, 24 of 25 sites in 2006, 29 of 29 in 2007, 29 of 31 in 2008, 42 of 48 in 2009, 52 of 53 in 2010, 57 of 65 in 2011, and 49 of 59 in 2012 were retained for analyses.

Indices of relative abundance from video data

The delta-lognormal index of relative abundance (I_y) as described by Lo *et al*. (1992) was estimated as

(1) $I_y = c_y p_y$,

where c_y is the estimate of mean CPUE for positive observations only for year y; 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) $\ln(\mathbf{c}) = \mathbf{X\beta} + \boldsymbol{\varepsilon}$

and

(3) $\mathbf{p} = \dfrac{e^{\mathbf{X\beta}+\boldsymbol{\varepsilon}}}{1+e^{\mathbf{X\beta}+\boldsymbol{\varepsilon}}}$, 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, $\boldsymbol{\beta}$ is the parameter vector for main effects, and $\boldsymbol{\varepsilon}$ is a vector of independent normally distributed errors with expectation zero and variance σ^2.

We used the GLIMMIX and MIXED procedures in SAS (v. 9.1, 2004) to develop the binomial and lognormal submodels, respectively. Similar covariates were tested for inclusion for both submodels: water depth, survey region [two regions in the northeastern GOM: East (east of Cape san Blas) and

West (west of east of Cape san Blas)], month and year. A backward selection procedure was used to determine which variables were to be included into each submodel based on type 3 analyses with a level of significance for inclusion of $\alpha = 0.05$. If year was not significant then it was forced into each submodel in order to estimate least-squares means for each year, which are predicted annual population margins (i.e., they estimate the marginal annual means as if over a balanced population).

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 (5), and its variance calculated as

$$(4) \qquad V(I_y) \approx V(c_y)p_y^2 + c_y^2 V(p_y) + 2c_y p_y \mathrm{Cov}(c, p),$$

where

$$(5) \qquad \mathrm{Cov}(c, p) \approx \rho_{\mathrm{c,p}} \left[\mathrm{SE}(c_y)\mathrm{SE}(p_y)\right],$$

and $\rho_{c,p}$ denotes correlation of c and p among years.

The backward selection procedure used to develop the delta-lognormal model for greater amberjack is summarized in Table 2. Month and depth variables were dropped from the binomial submodel based on type 3 analyses. For the lognormal submodel for nonzero observations of greater amberjack, month and region variables were dropped (Table 2).

Results

Greater amberjacks have consistently been observed with stationary video gear across the northern portion of the inner and mid- West Florida shelf since 2006, although they have been much more common west of Cape San Blas than east (Tables 1, Fig. 4 and 5)(DeVries et al. 2008, 2009, 2012). Annual frequency of occurrence 2005-2012 ranged from 21-55 % (mean = 33 %) west of the Cape compared to 2-32 % (mean = 12 %) in the east (Table 1, Fig. 6). Virtually no greater amberjacks have ever been taken in chevron traps in the Panama City lab survey. Greater amberjack were not uniformly distributed across all depths sampled east of the Cape, but were in the west (Fig. 7a and b); 24% of all sites in the east were shallower than 14 m but only 9% of positive amberjack sites occurred in those depths, and none were found <10.1 m.

The video survey strongly targets pre-recruit greater amberjacks - about 98% (100% east of the Cape and 97% in the west) were <762 mm FL, the recreational minimum size limit during 2009-2012 (Fig. 8). Greater amberjack east of the Cape tended to be smaller than those in the west ($\bar{x}$ = 383 vs. 475 mm FL), with 87% <450 mm in the east vs. 53% in the west. The minimum size was also much smaller in the east – 154 vs. 305 mm. Although sample sizes were small in 2010 and 2011, annual size structure data did hint at the progression of a modal group from 350-500 mm in 2009 to 500-650 mm the following year (Fig. 9). In 2012 a group of fish 150- 350 mm, smaller than seen in the previous 3 years, dominated the size structure east of the Cape but was absent in the west (Fig. 9). There was no apparent relationship between size and depth in greater amberjack observed in the video survey (Fig. 10).

Although no age data were available from the survey, a comparison of the overall size distribution of greater amberjack measured from survey stereo images with age-specific size distributions derived from Florida specimens, ages 0-3, aged in other studies (subsample of age data described in Allman et al. 2013), strongly suggests that the majority observed were age 1, with fewer age 0's and 2's, and no age 3's (Fig. 11). Most, if not all, of the likely age 0 fish observed were in the strong mode of very small fish that was only present in 2012 (Fig. 9).

Video indices of abundance

Overall nominal mean min count indices and frequency of occurrence data from the video survey, 2005-2012, revealed considerable annual fluctuations in the abundance of younger (ages 0-2 yr but likely primarily age 1) greater amberjack on the northern portion of the west Florida shelf, and both metrics showed similar trends 2008-2012 (Fig. 12). During 2006-2010 the trends were opposite east and west of Cape San Blas (Table 1, Fig. 13). Nominal CPUE in the east increased 5 fold from 2007 to 2009 but dropped 99% by 2011 (94 fold difference), while in the west there was a 7 fold difference between the peak CPUE in 2006 and the lowest in 2011, followed by a 6 fold increase from 2011 to 2012 (Fig. 13). Nominal mean min count was lowest in both areas in 2011, but was highest in the east in 2009 vs. 2006 in the west (Fig. 13). Much of the difference in nominal mean min counts (CPUE) east and west of the Cape in the early years (2005-2007) was likely related to the much shallower depths sampled in the east prior to 2009 (Fig. 3). The complete absence of greater amberjacks in 2005, when only the east was sampled, probably reflects the fact that 61% of the sites sampled that year were 14 m or shallower depths (Fig. 14), and 34% were in 10 m or less – depths where no amberjack were ever observed.

The delta-lognormal unscaled and scaled indices of relative abundance of greater amberjack derived from the Panama City lab video min count data are shown and summarized in Figure 15. Figures 16 and 17 provide diagnostics for each of the submodels in the index development. The QQ plot sub-figure in each of the aforementioned figures indicates the approximately normal distribution of the residuals of corresponding submodels.

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Tables

Table 1. Annual sample sizes, % frequencies of occurrence, raw mean nominal video min counts, and standard errors of greater amberjack east and west of Cape San Blas, 2005-2012. Estimates calculated using censored data sets (see Methods).

	Total sites sampled		% Frequency of occurrence		Mean nominal min count		Standard error	
Year	East	West	East	West	East	West	East	West
2005	31		0.0		0.000		0.000	
2006	49	24	6.1	20.8	0.449	1.750	0.352	1.034
2007	29	23	10.3	34.8	0.310	1.348	0.217	0.568
2008	56	29	8.9	27.6	0.875	1.172	0.417	0.525
2009	62	42	32.3	40.5	1.903	0.833	0.580	0.193
2010	95	52	7.4	34.6	0.242	0.981	0.119	0.282
2011	100	58	2.0	15.5	0.020	0.224	0.014	0.078
2012	101	49	16.8	55.1	0.653	1.551	0.273	0.420

Table 2. Backward selection procedure for building delta-lognormal submodels for greater amberjack observed during PC Video Surveys in the northeastern Gulf of Mexico. ** indicates the model chosen for the index.

Model Run #1	*Binomial Submodel Type 3 Tests (AIC = 3832.6)*						*Lognormal Submodel Type 3 Tests (AIC = 371.7)*			
Effect	*Num DF*	*Den DF*	*Chi-Square*	*F Value*	*Pr > ChiSq*	*Pr > F*	*Num DF*	*Den DF*	*F Value*	*Pr > F*
Year	6	755	47.97	7.99	<.0001	<.0001	6	135	2.54	0.0231
Month	5	755	4.36	0.87	0.4989	0.4995	5	135	0.98	0.4299
Region	1	755	22.68	22.68	<.0001	<.0001	1	135	0.07	0.7897
Depth	1	755	0.60	0.60	0.4380	0.4382	1	135	2.69	0.1036
Model Run #2	*Binomial Submodel Type 3 Tests (AIC = 3840.6)*						*Lognormal Submodel Type 3 Tests (AIC = 370.4)*			
Effect	*Num DF*	*Den DF*	*Chi-Square*	*F Value*	*Pr > ChiSq*	*Pr > F*	*Num DF*	*Den DF*	*F Value*	*Pr > F*
Year	6	760	45.34	7.56	<.0001	<.0001	6	136	2.57	0.0216
Month	dropped						5	136	1.43	0.2191
Region	1	760	33.81	33.81	<.0001	<.0001	dropped			
Depth	1	760	0.90	0.90	0.3439	0.3442	1	136	3.57	0.0608
Model Run #3**	*Binomial Submodel Type 3 Tests (AIC = 3831.9)*						*Lognormal Submodel Type 3 Tests (AIC = 371.8)*			
Effect	*Num DF*	*Den DF*	*Chi-Square*	*F Value*	*Pr > ChiSq*	*Pr > F*	*Num DF*	*Den DF*	*F Value*	*Pr > F*
Year	6	761	45.97	7.66	<.0001	<.0001	6	141	2.39	0.0316
Month	dropped						dropped			
Region	1	761	53.00	53.00	<.0001	<.0001	dropped			
Depth	dropped						1	141	8.85	0.0034

Figures

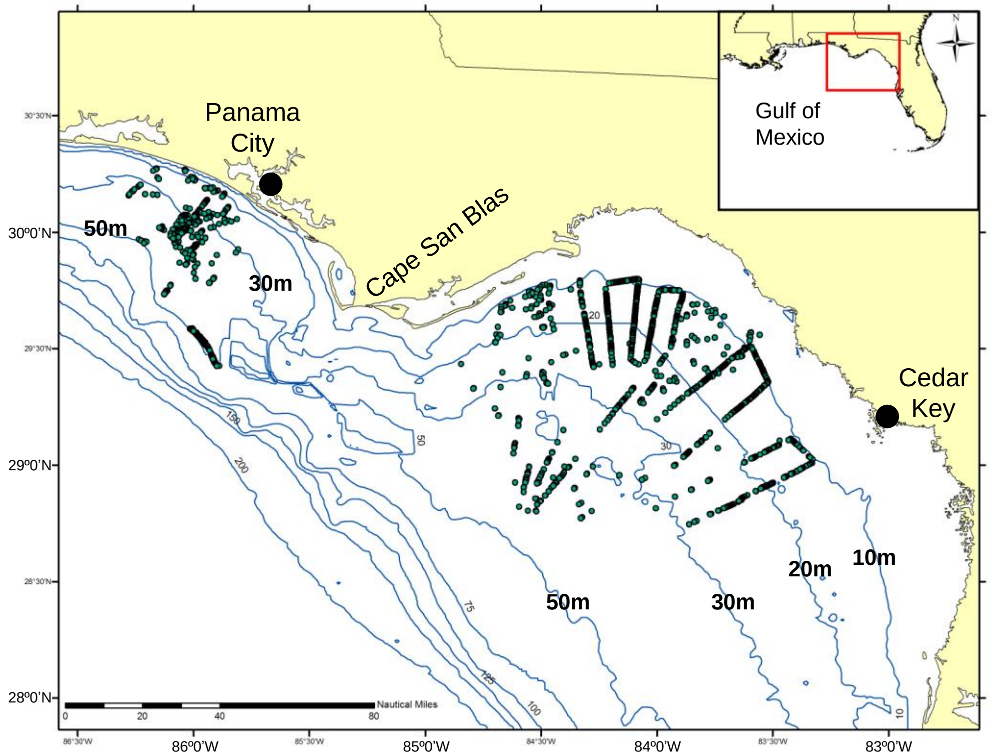


Figure 1. Locations of all natural reefs in the sampling universe of the Panama City NMFS reef fish video survey as of March 2012. Total sites: 2359, 722 west of and 1637 east of Cape San Blas.

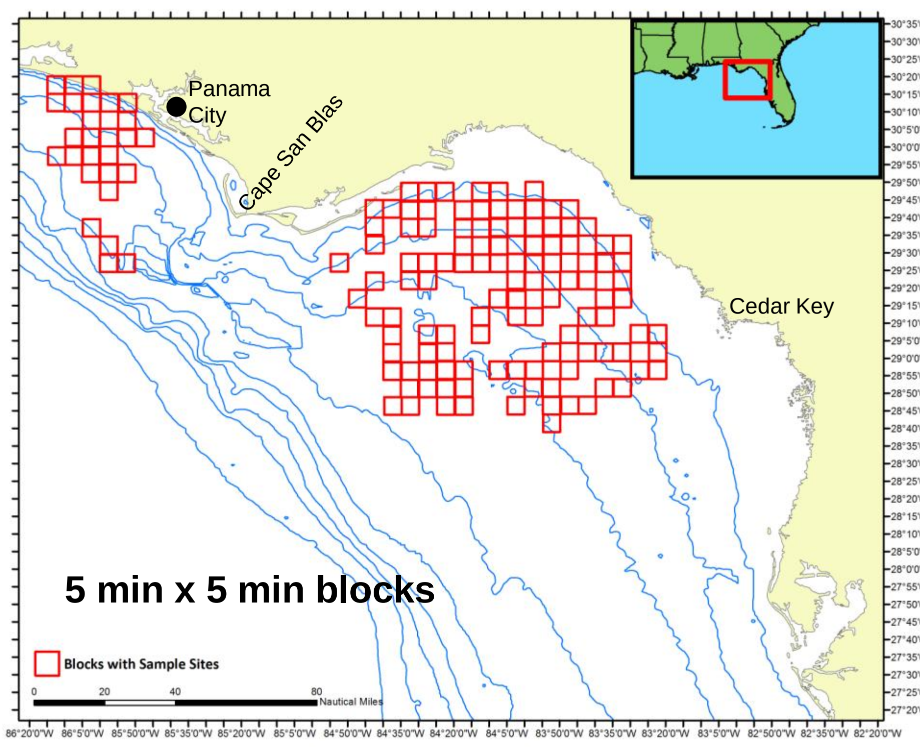


Figure 2. Sampling blocks, as of 2012, of the Panama City reef fish survey.

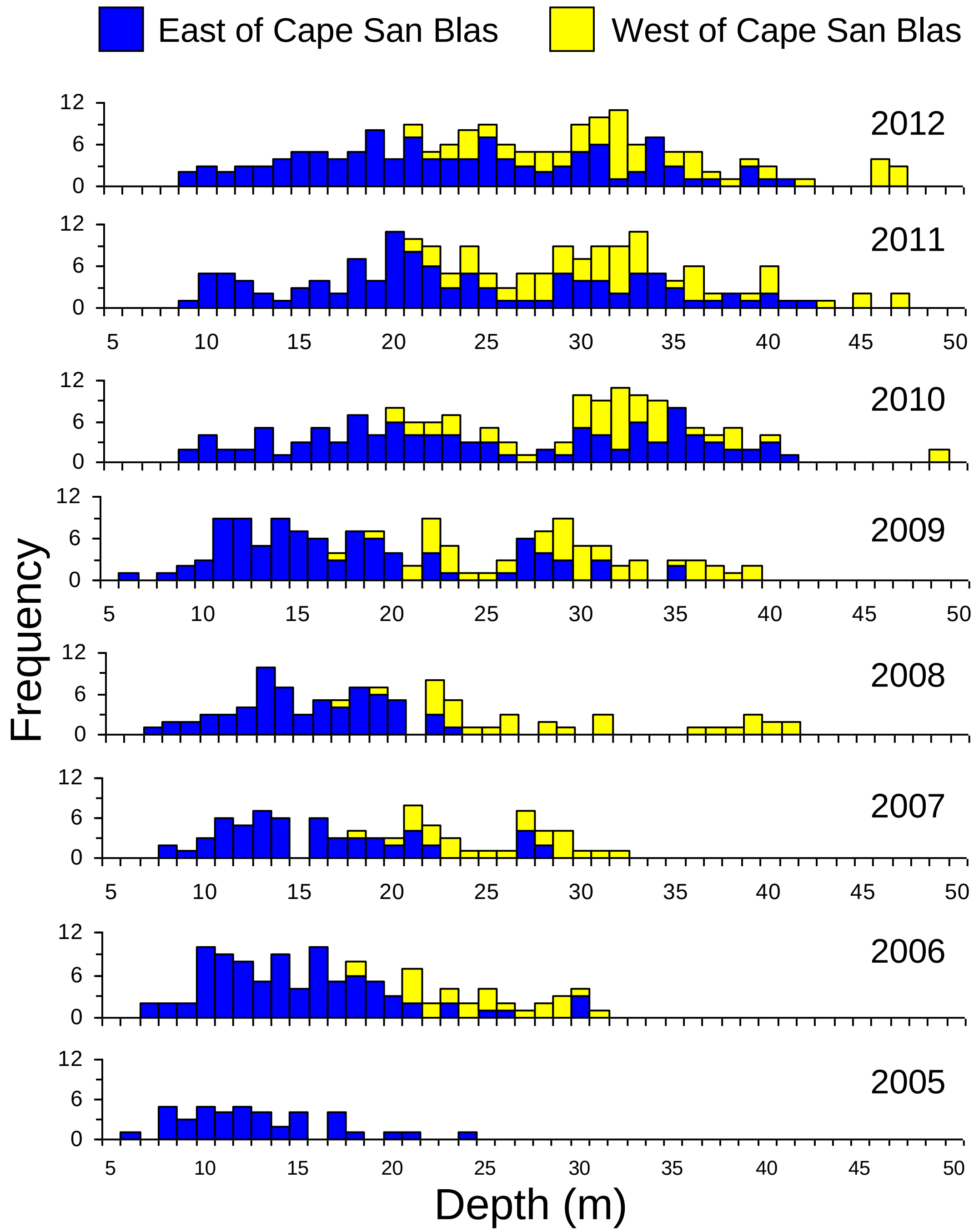


Figure 3. Annual depth distribution of Panama City reef fish survey video sample sites east and west of Cape San Blas, 2005-2012.

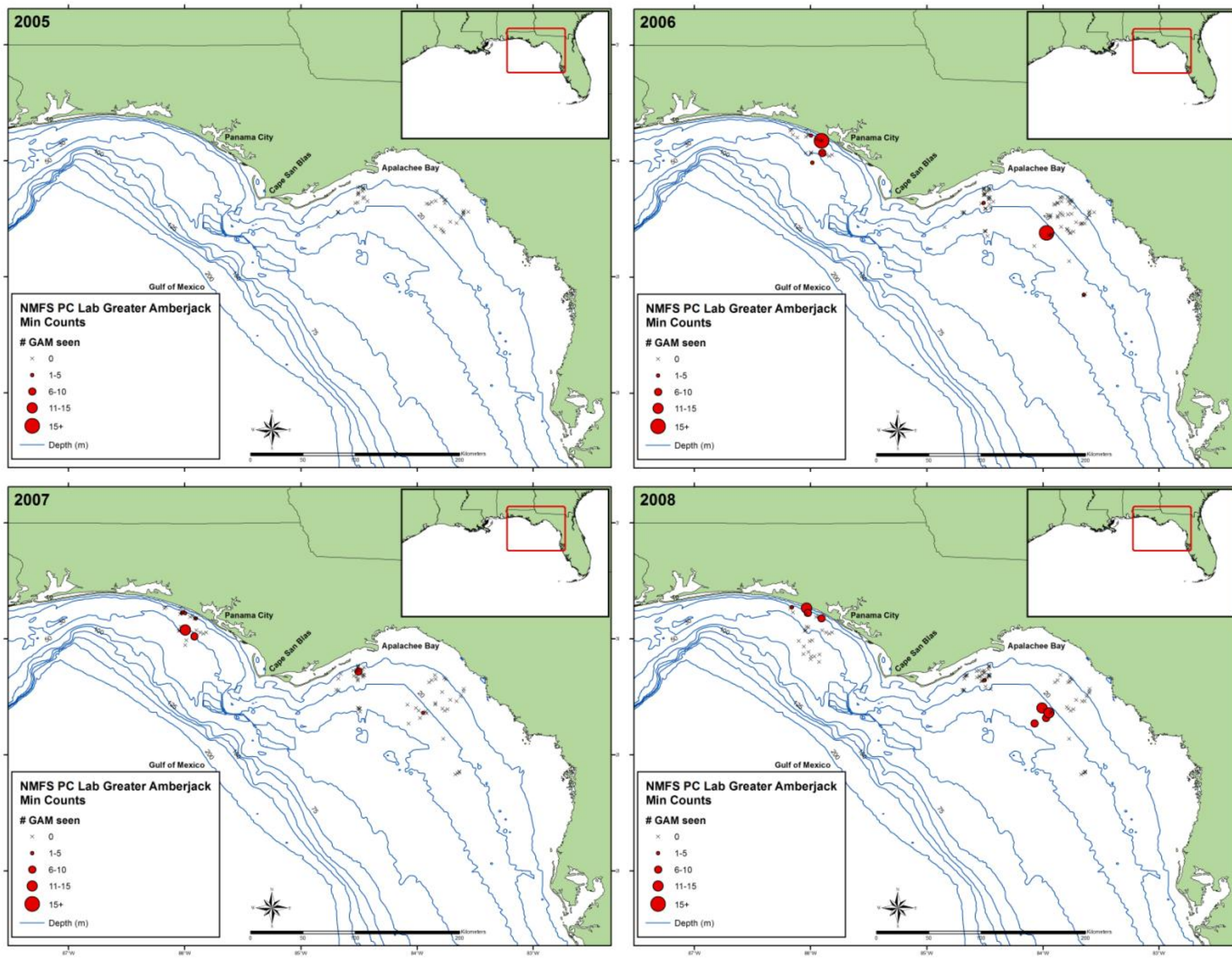


Figure 4. Annual distribution and relative abundance (min counts) of greater amberjack observed in the Panama City NMFS reef fish survey, 2005-2008, with stationary, high definition video cameras. Sites sampled with video gear, but where no greater amberjack were observed, are indicated with an X.

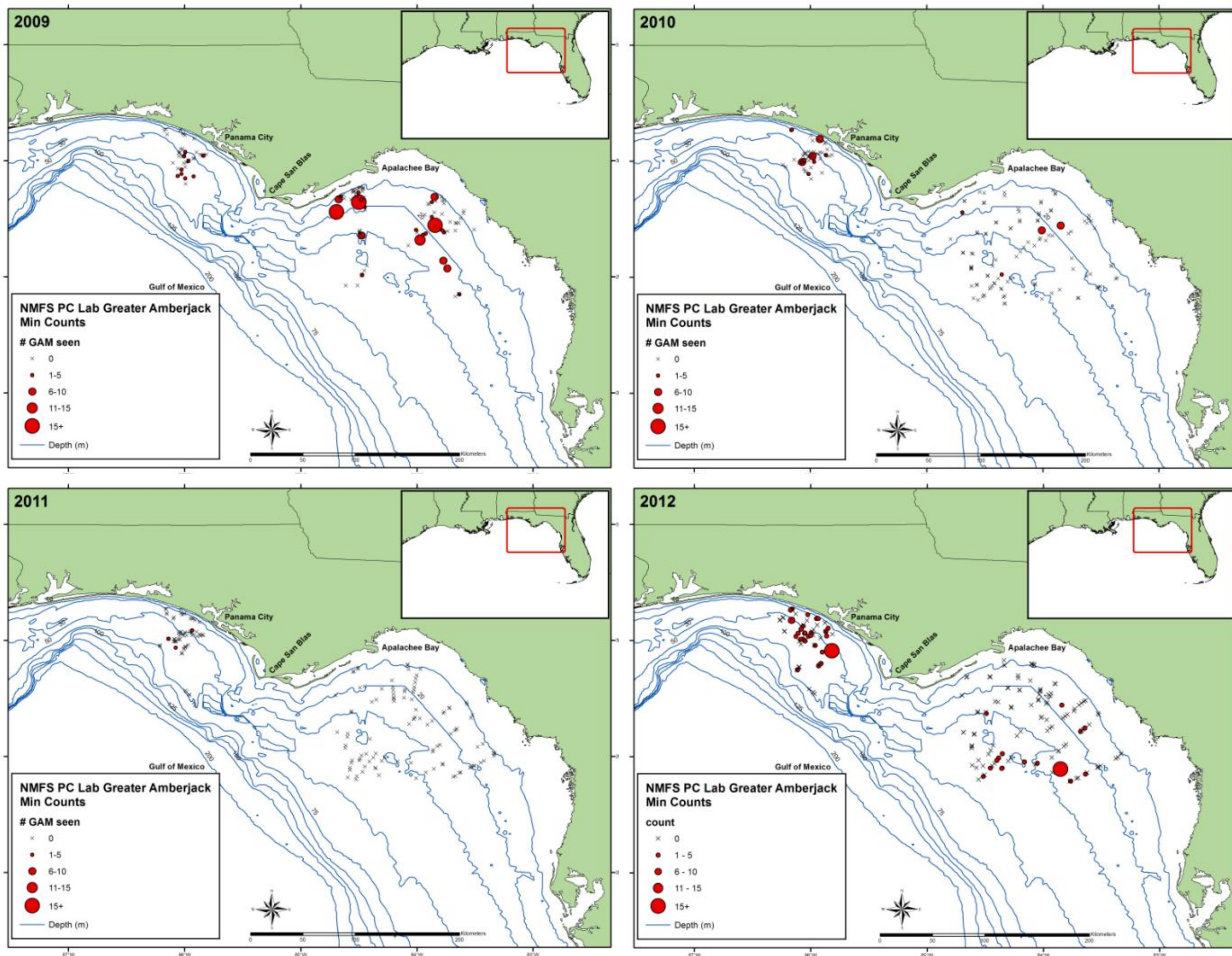


Figure 5. Annual distribution and relative abundance (min counts) of greater amberjack observed in the Panama City NMFS reef fish survey, 2009-2012, with stationary, high definition video or mpeg cameras. Sites sampled with video gear, but where no greater amberjack were observed, are indicated with an X.

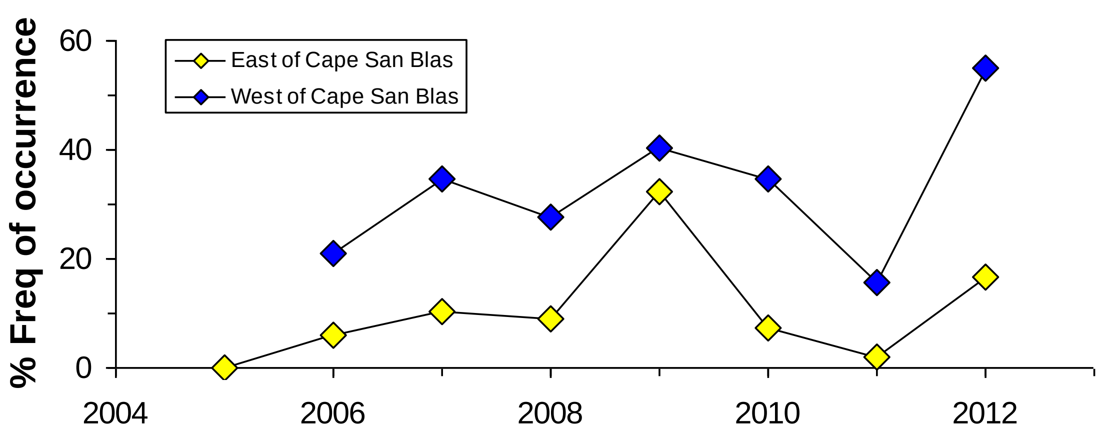


Figure 6. Annual percent frequency of occurrence of greater amberjack in video samples east and west of Cape San Blas, 2005-2012, using censored data sets.

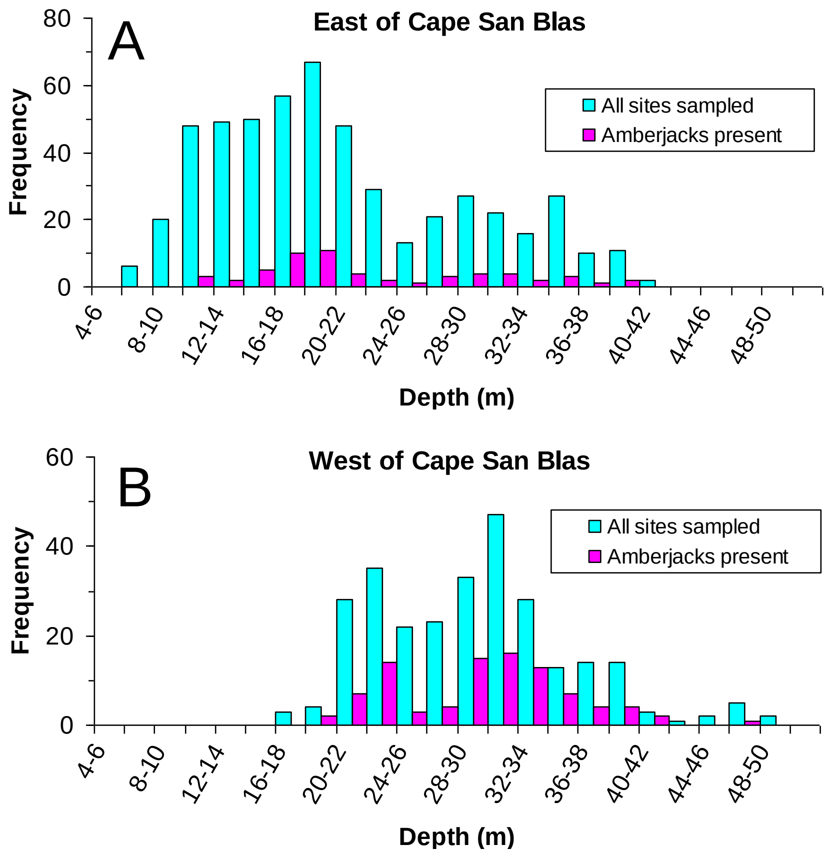


Figure 7. A. Depth distribution of all video sample sites and all sites with greater amberjack east of Cape San Blas, 2005-2012. B. Depth distribution of all video sample sites and all sites with greater amberjack west of Cape San Blas, 2006-2012.

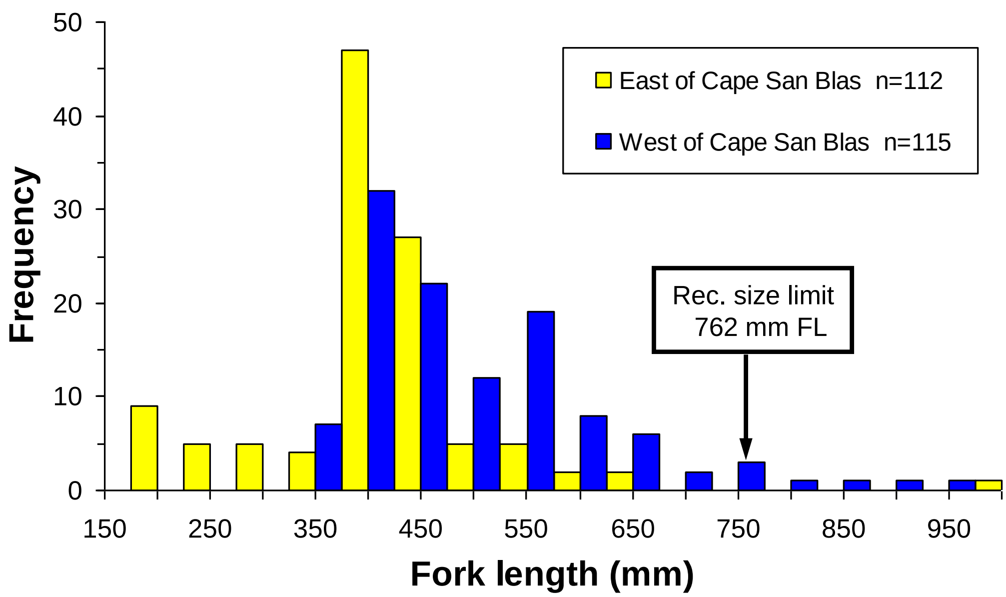


Figure 8 Overall size distributions of greater amberjack east and west of Cape San Blas observed with stereo cameras and measured using VMS, 2009-2012.

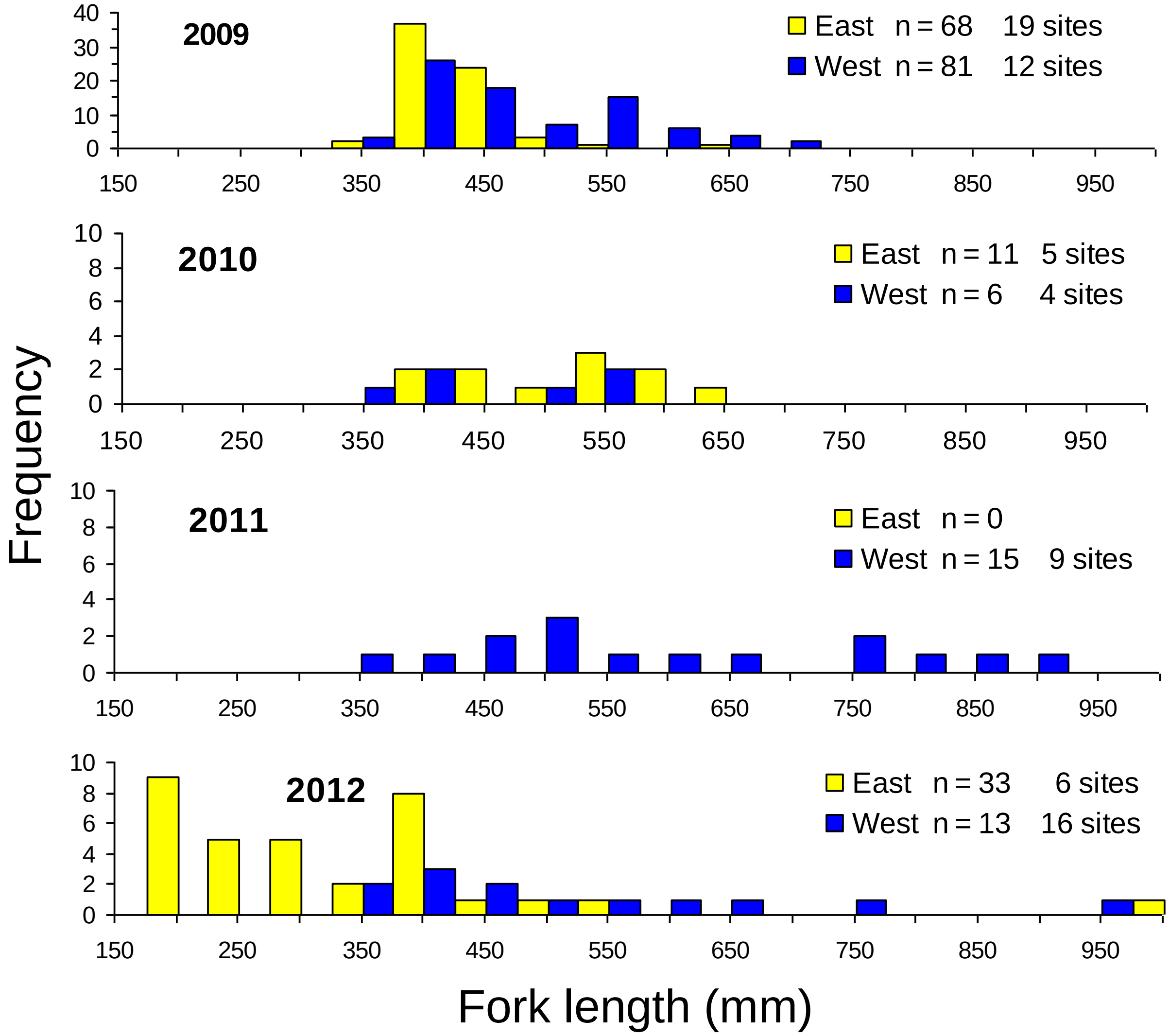


Figure 9. Annual size distributions of greater amberjack, 2009-2012, east and west of Cape San Blas observed with stereo cameras and measured using VMS.

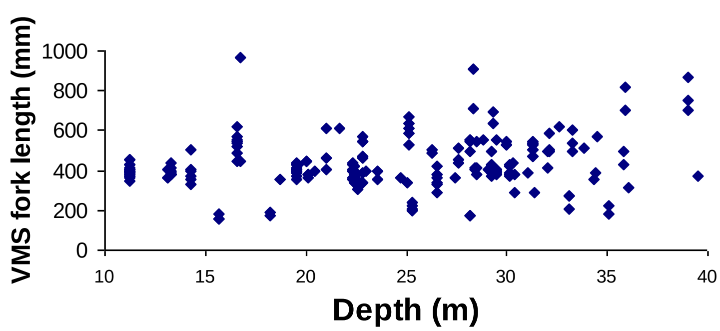


Figure 10. Fork length vs. depth relationship of all greater amberjack observed in the Panama City lab video survey with stereo cameras and measured using VMS, 2009-2012.

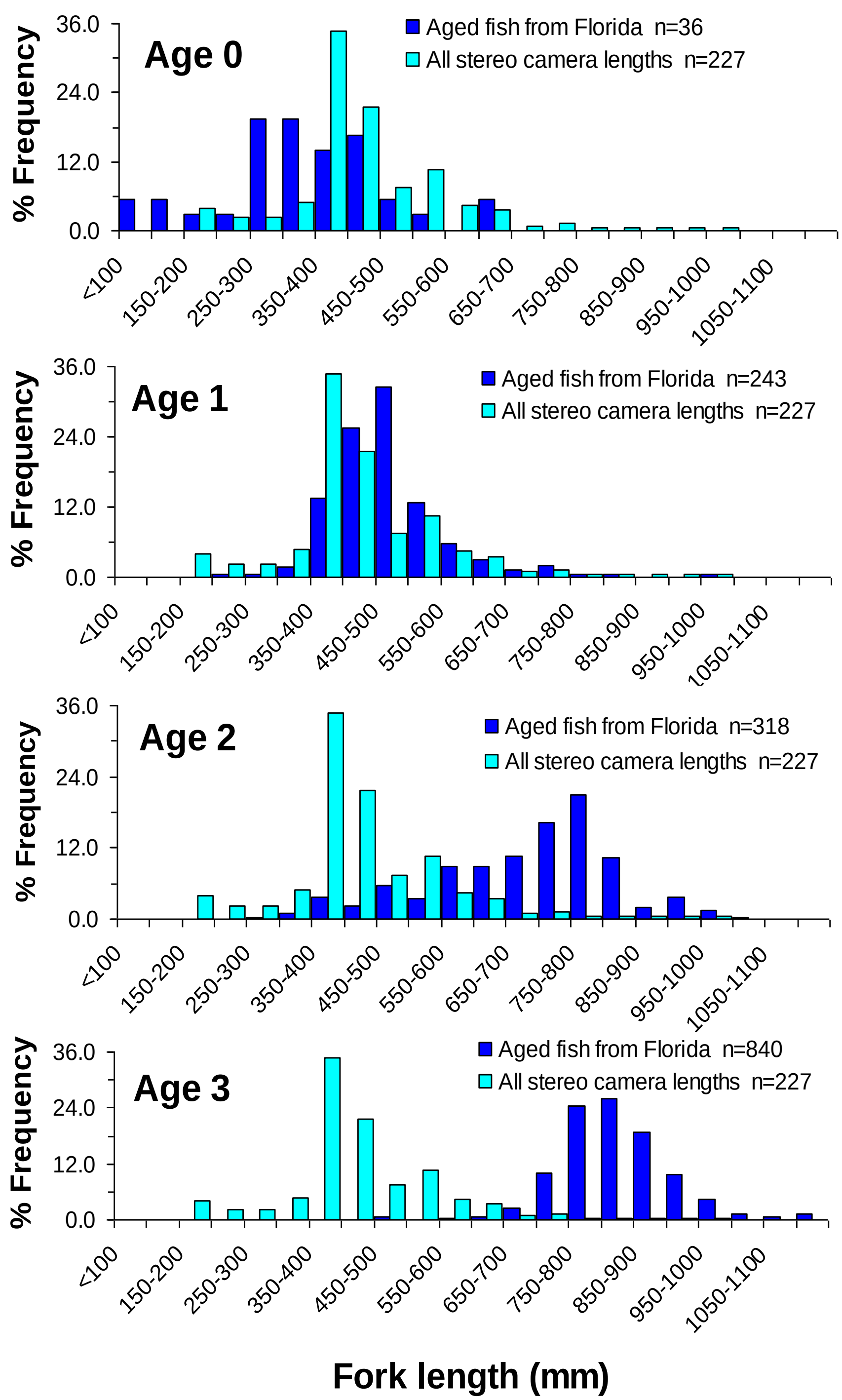


Figure 11. Comparison of age-specific size structure of greater amberjack, ages 0-3, from fish collected in Florida, 1980-2012 (data set described in Allman et al. 2013) with size distribution of all fish measured from Panama City survey stereo images, 2009-2012.

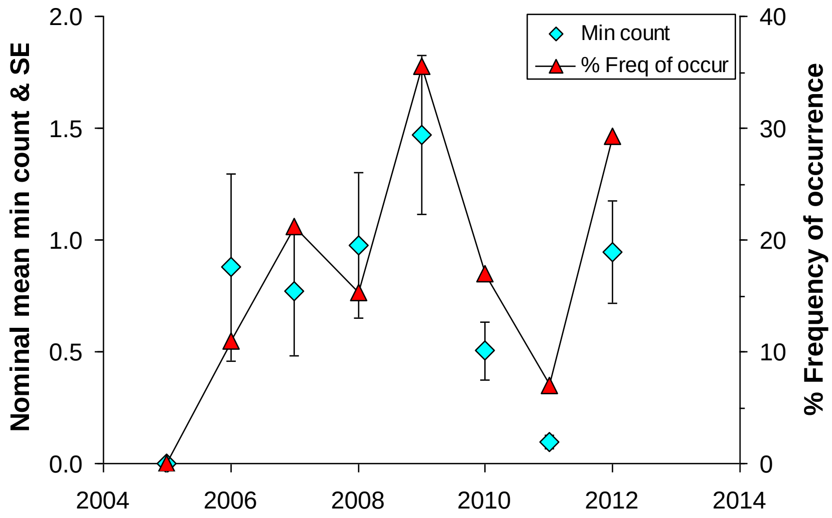


Figure 12. Overall (east + west of Cape San Blas) annual nominal mean video min counts ± SE and % frequencies of occurrence of greater amberjack, 2005-2012. Only 41 sites east of Cape San Blas were sampled in 2005, and 37 of those were shallower than 18 m.

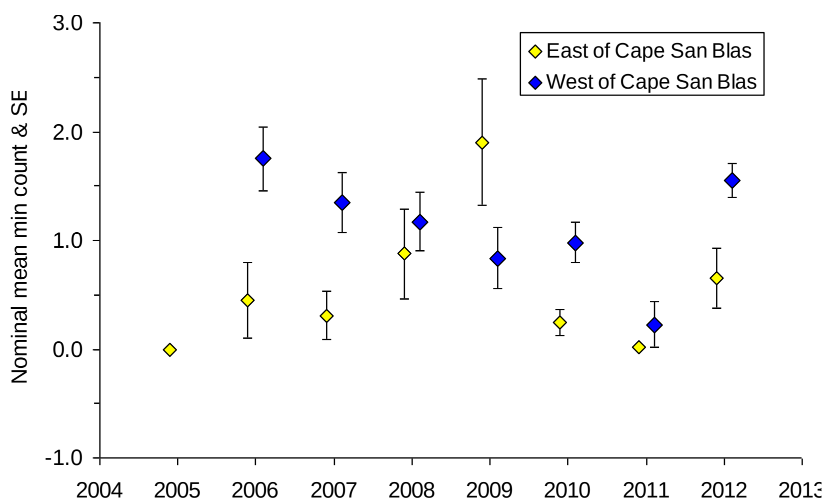


Figure 13. Nominal mean video min counts and standard errors of greater amberjack east and west of Cape San Blas, 2005-2012.

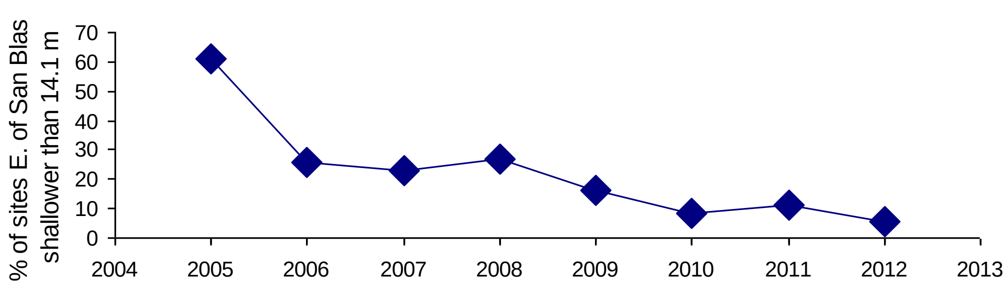
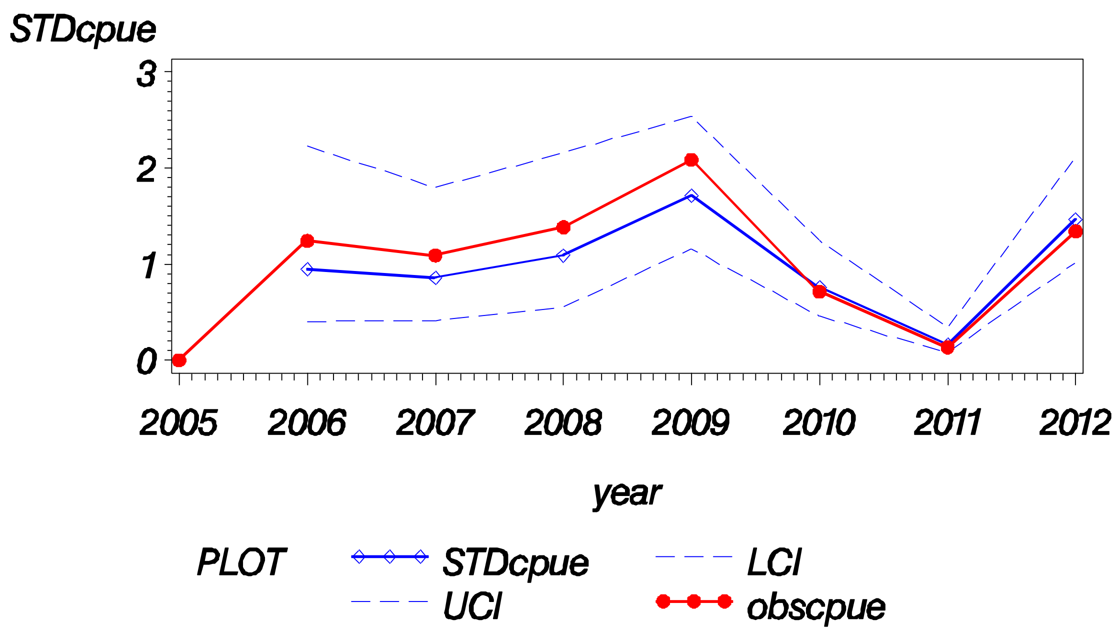


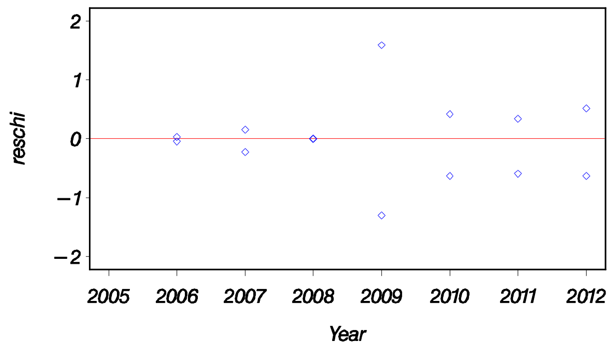
Figure 14. Annual proportion of video sample sites east of Cape San Blas in depths <14.1 m.



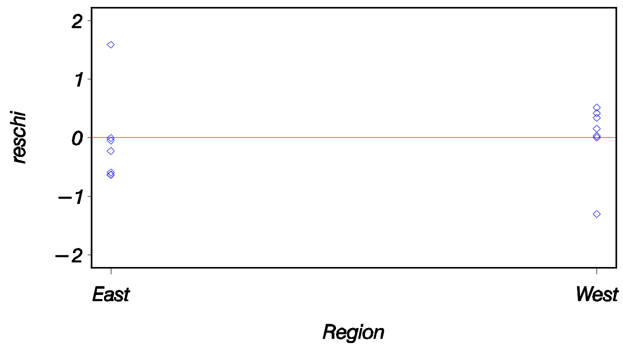
Survey Year	*Frequency*	*N*	*Index*	*Scaled Index*	*CV*	*LCL*	*UCL*
2005	0.00000	31					
2006	0.10959	73	0.69957	0.94676	0.44831	0.40228	2.22820
2007	0.21154	52	0.63627	0.86111	0.37975	0.41329	1.79414
2008	0.15294	85	0.80661	1.09163	0.35090	0.55219	2.15804
2009	0.35577	104	1.26600	1.71335	0.19799	1.15752	2.53610
2010	0.17007	147	0.55889	0.75638	0.25408	0.45866	1.24734
2011	0.06962	158	0.12025	0.16274	0.38881	0.07684	0.34467
2012	0.29333	150	1.08473	1.46803	0.18495	1.01728	2.11851

Figure 15. Panama City lab video abundance indices for greater amberjack. STDcpue is the index scaled to a mean of one over the time series. Obscpue is the average nominal CPUE, and LCI and UCI are 95% confidence limits. In the accompanying table, the *frequency* listed is nominal frequency, *N* is the number of video stations, *Index* is the abundance index in CPUE units, *Scaled Index* is the index scaled to a mean of one over the time series, *CV* is the coefficient of variation on the index value, and *LCL* and *UCL* are 95% confidence limits.

a. Chi-square residuals by year.



b. Chi-square residuals by region.



c. QQplot of chi-square residuals.

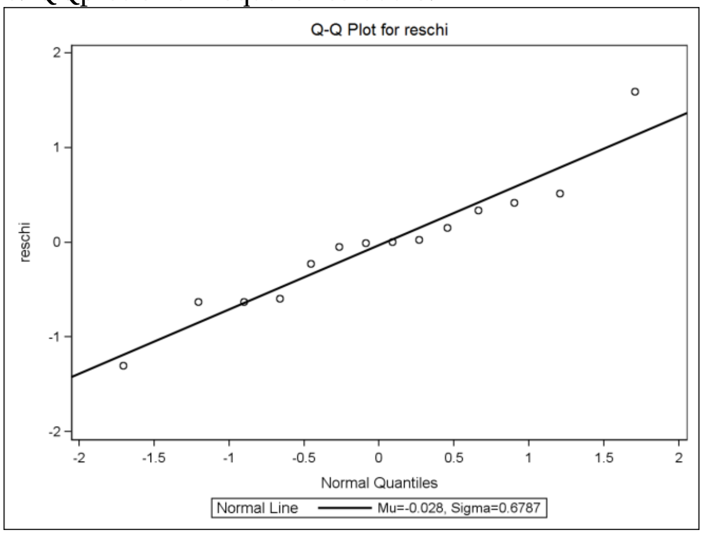


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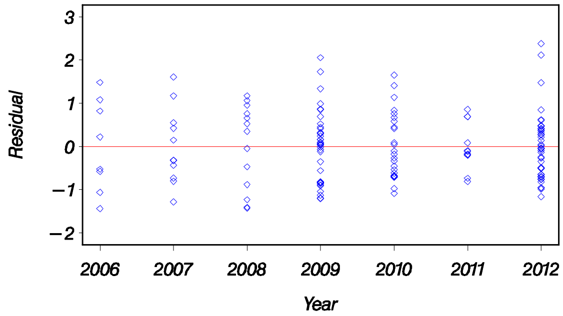
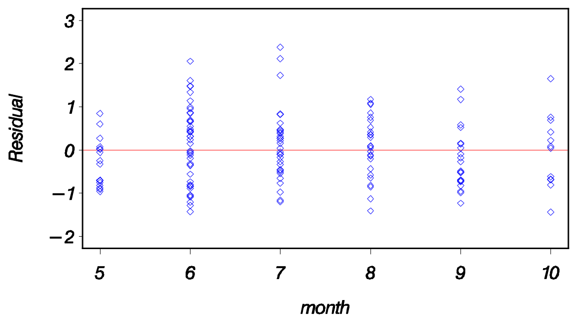
Figure 16. Diagnostic residual plots of the binomial submodel for greater amberjack observed during Panama City lab video surveys in the northeastern Gulf of Mexico.

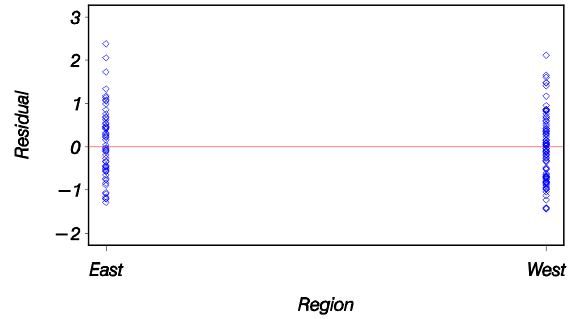
a. Chi-square residuals by year.



b. Chi-square residuals by month.



c. Chi-square residuals by region.



d. QQplot of chi-square residuals.

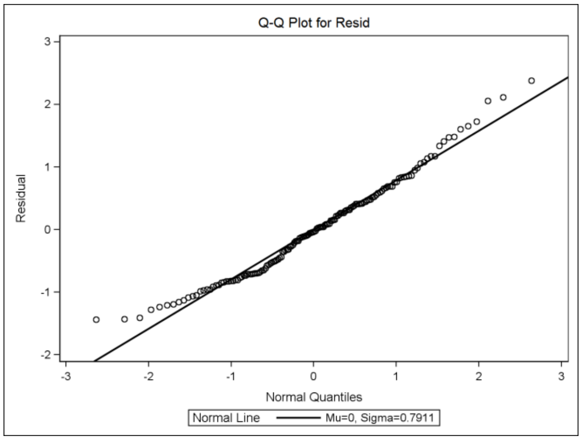


Figure 17. Diagnostic residual plots of the lognormal submodel for greater amberjack observed during Panama City lab video surveys in the northeastern Gulf of Mexico.