Red Snapper Findings from the NMFS Panama City Laboratory Trap & Camera Fishery-Independent Survey – 2004-2011

D.A. DeVries, W. Ingram, C.L. Gardner, and P. Raley

SEDAR31-DW28

20 August 2012

Red Snapper Findings from the NMFS Panama City Laboratory Trap & Camera Fishery-Independent Survey – 2004-2011

D.A. DeVries, W. Ingram, C.L. Gardner, and P. Raley
National Marine Fisheries Service
Southeast Fisheries Science Center
Panama City Laboratory
Panama City, FL

August 2012

Panama City Laboratory
Contribution 12-xx

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. 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, but was changed to stratified random in 2010 after side scan sonar surveys that year yielded an order of magnitude increase in that universe. To ensure uniform geographic and bathymetric coverage, 2-stage sampling is used. Five by five minute blocks, stratified by depth zone (< and >30 m) and geographical location, and known to contain reef sites, are randomly chosen first, then 2 sites a minimum of 300 m apart within each selected block (Figure 2).

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 (25 W: 89 E), 86 in '07 (29 W: 57 E), , 98 in '08 (31 W: 66 E), , 143 in '09 (48 W: 97 E), , 162 in '10 (53 W: 109 E), , and 170 in '11 (65 W: 115 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 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).

During 2005 – 2008, visual data were collected using a stationary camera array composed of 4 high definition (HDEF), digital video cameras 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; both SIS's were also upgraded with HDEF, color mpeg cameras.

When only HDEF cameras were used (through 2008), 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 hard drive in the SIS 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, and 99 of 115 in 2011 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 47 in 2009, 52 of 53 in 2010, and 57 of 64 in 2011 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 (video min count) 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 is summarized in Table X for red snapper.

The month effect was dropped from the binomial submodel based on type 3 analyses. However, with the variable removal there was a corresponding increase in AIC (Table 1), but due to the high insignificance of the month variable, it was left out of the model. For the lognormal submodel for nonzero observation of red snapper, the water depth variable was dropped from the model, and there was a corresponding decrease in the AIC value (Table 1).

Results

Red snapper distribution and abundance on the inner and mid shelf have consistently and noticeably differed east and west of Cape San Blas since the Panama City survey began in 2004/5 (Tables 1 and 2, Figures 4 and 5)(DeVries et al. 2008, 2009). Red snapper has been, by far, the most commonly encountered exploited reef fish west of Cape San Blas (the Cape) every year, occurring in 47 – 88 % of trap catches and 91 – 100 % of video samples (Table 1, Fig. 6). In contrast, east of the Cape, red snapper have been much less common, especially during 2004-08, when they occurred in 0-8 % of trap sets and 9-26 % of video samples. Since 2009, up through 2011, those numbers have been considerably higher: 20-36 % for traps and 26-46 % for video (Table 1, Fig. 4). Some of the increase reflects 1) the difference in the distribution of depths sampled in each area, e.g., only a small proportion of sites <20 m have been sampled west of the Cape, while in the east through 2009, very few sites >20 m were sampled; as well as 2) the expansion of sampling to deeper depths over time (Fig. 3). Figure 4 clearly shows that red snapper east of Cape San Blas were rarely observed in depths <20 m. Although the sampling depth differences and changes likely explain some of the increases in occurrence, it also appears to reflect an expansion of the population into Apalachee Bay, as occurrence increased noticeably even in shallow (<20 m) areas, especially an area in northwest Apalachee Bay in 2009 (Fig. 5).

Overall modal size of red snapper taken in traps was fairly stable 2005-2007, ranging from 300 to 350 mm TL, then steadily increased through 2011, when it was 375 to 425 mm TL (Fig. 7). Along with this increase in modal size, the lower (left hand) tail of the distribution also shifted, increasing from around 200-225 mm in 2005 to about 325 mm in 2011. Part of this shift may reflect the expansion of the sampling depth range west of the Cape during those years, as a comparison of size structure in depths < and > 30 m (Fig. 8) clearly showed smaller average sizes in shallower depths. However, the shift in size structure co-occurred with increasingly restrictive management measures and mirrored the steady increases in average sizes (and catch rates) of recreationally harvested fish in the area, which suggests it shows a real trend in the population and is not just an artifact of changes in sampling depths.

Not surprisingly, a comparison of size data from trap catches with that from stereo images indicated that the traps do select against most red snapper >650 mm TL, although fish that large appear to be uncommon in the survey area based on the few stereo measurements obtained (Fig. 9). For the most part, in 2011, west of the Cape, the size distributions were surprisingly similar between the two gears, except for the rare large fish detected only with the video gear. Earlier (2007-09) size data from scaling lasers suggested traps were selecting against the smallest individuals, perhaps an inhibiting effect of larger, more aggressive fish entering the trap first. In 2009, unexpectedly, the distribution of the laser measurements was shifted to the left (smaller) of that from the stereo data, with an obviously smaller mode; while the distributions of the trap fish and that from the stereo images, like in 2011, were very similar. Given the problem of potentially measuring the same fish more than once with lasers, length data from stereo images taken from a frame with no more than the min count of that site is likely to be more unbiased.

Age structure in trap catches during 2005-2007 was dominated by 2 and 3 yr olds, with an obvious mode at age 2 ; one and four yr olds were uncommon, except for age ones in 2005 (Fig. 10). In 2008 and 2009, two and three yr olds still dominated the age structure; and in 2008, for the first time, four yr olds were quite common and a few fish to age 8 were caught. The 2006 and 2007 year classes, equating to the 2 and 3 yr old modal group in 2009, continued to dominate the age structure as 3 and 4 yr olds in 2010 and 4 and 5 yr olds in 2011, suggesting these two year classes were fairly strong. In 2010, age ones were no longer present, and by 2011, as the distribution continued to shift to older ages, age 2 fish were also virtually nonexistent.

Video indices of abundance

Figure 11 and its adjoining table summarize indices of red snapper developed from the Panama City video data, 2005-2011, using a delta-lognormal model. The index, scaled to a mean of one over the time series, peaked in 2009; and based on the age frequency data from trap catches (Fig. 9), the fish were primarily from the 2006 and 2007 year classes. The index declined in 2010 and 2011, perhaps as the influence of the apparently strong 06 and 07 cohorts waned. Figures 12 and 13 provide diagnostics for each of the submodels in the index development; and the QQ plots in each indicate the approximately normal distribution of the residuals of corresponding submodels.

Literature Cited

DeVries, D.A, J.H. Brusher, C.L. Gardner, and G.R. Fitzhugh. 2008. NMFS Panama City Laboratory trap & camera survey for reef fish. Annual Report of 2007 results. Panama City Laboratory Contribution 08-14. 20 pp.

DeVries, D.A., J. H. Brusher, C. L. Gardner, and G. R. Fitzhugh. 2009. NMFS Panama City Laboratory trap and camera survey for reef fish. Annual report of 2008 results. Panama City Laboratory, Contribution Series 09-10. 22 p.

Gledhill, C., and A. David. 2003. Survey of fish assemblages and habitat within two marine protected areas on the West Florida shelf. NMFS, Southeast Fisheries Science Center. Report to the Gulf of Mexico Fishery Management Council.
Gledhill, C. and W. Ingram. 2004. SEAMAP Reef Fish survey of Offshore Banks. 14 p. plus appendices. NMFS, Southeast Fisheries Science Center, Mississippi Laboratories. SEDAR 7 – DW 15.
GMFMC. 2001. October 2001 report of the Reef Fish Stock Assessment Panel. Gulf of Mexico Fishery Management Council, Tampa, FL. 34 pp.
LO, N. C. H., L.D. Jacobson, and J.L. Squire. 1992. Indices of relative abundance from fish spotter data based on delta-lognormal models. Can. J. Fish. Aquat. Sci. 49: 2515-1526.
McGovern, J. C., G.R. Sedberry and P.J. Harris. 1998. The status of reef fish stocks off the southeast United States, 1983-1996. Gulf and Caribbean Fisheries Institute 50: 871-895.
Mahmoudi, B. 2005. State-Federal Cooperative Reef fish Research and Monitoring Initiative in the Eastern Gulf of Mexico. Workshop report. March 3-4 2005, Florida Fish and Wildlife Research Institute, St. Petersburg, Florida.
Sokal, R.R., and F.J. Rohlf. 1969. Biometry. W.H. Freeman and Company, San Francisco, CA. 776 p.
Steel, R.G.D., and J.H. Torrie. 1960. Principles and procedures of statistics. McGraw-Hill Book Company, New York, NY. 481 p.
Zieman, J.C., and R.T. Zieman. 1989. The ecology of the seagrass meadows of the west coast of Florida: A community profile. Biological Report 85(7.25). U.S. Fish and Wildlife Service. 155 p.

Tables

Table 1. Annual % frequencies of occurrence of red snapper in trap and video samples east and west of Cape San Blas, and total number of sites sampled. Data from all sites were included for trap estimates; censored data sets were used to calculate video frequencies.

	Chevron trap				Video			
	Total sites sampled		% Freq. Occur.		Total sites sampled		% Freq. Occur.	
Year	East	West	East	West	East	West	East	West
2004	53	33	3.8	63.6				
2005	77	24	6.5	87.5	31		9.7	
2006	89	25	7.9	88.0	47	24	25.5	95.8
2007	57	26	7.0	69.2	29	23	13.8	100.0
2008	51	29	0	86.2	56	29	8.9	96.6
2009	53	30	35.8	86.7	62	42	45.2	100.0
2010	53	17	18.9	47.1	95	52	46.3	92.3
2011	50	32	20.0	84.4	99	57	26.3	91.2

Table 2. Mean annual video min counts, standard errors, and sample sizes of red snapper east and west of Cape San Blas, 2005-2011. Estimates calculated using censored data sets (see Methods).

	Total sites sampled		Mean nominal min count		Standard error	
Year	East	West	East	West	East	West
2005	31		0.129		0.077	
2006	47	24	0.830	7.583	0.581	1.103
2007	29	23	0.345	10.348	0.974	1.537
2008	56	29	0.089	6.345	0.428	0.826
2009	62	42	1.452	10.452	0.723	1.375
2010	95	52	1.305	6.942	0.409	0.915
2011	99	57	0.515	5.491	0.318	0.672

Table 3. Backward selection procedure for building delta-lognormal submodels for red snapper 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 = 3474.0)*						*Lognormal Submodel Type 3 Tests (AIC = 745.4)*			
Effect	*Num DF*	*Den DF*	*Chi-Square*	*F Value*	*Pr > ChiSq*	*Pr > F*	*Num DF*	*Den DF*	*F Value*	*Pr > F*
Year	6	633	21.16	3.53	0.0017	0.0019	6	326	5.02	<.0001
Month	5	633	3.99	0.80	0.5507	0.5511	5	326	5.25	0.0001
Region	1	633	47.87	47.87	<.0001	<.0001	1	326	131.16	<.0001
Depth	1	633	61.15	61.15	<.0001	<.0001	1	326	0.13	0.7202
Model Run #2*	*Binomial Submodel Type 3 Tests (AIC = 3537.3)*						*Lognormal Submodel Type 3 Tests (AIC = 737.2)*			
Effect	*Num DF*	*Den DF*	*Chi-Square*	*F Value*	*Pr > ChiSq*	*Pr > F*	*Num DF*	*Den DF*	*F Value*	*Pr > F*
Year	6	638	20.02	3.34	0.0027	0.0030	6	327	5.70	<.0001
Month	dropped						5	327	5.74	<.0001
Region	1	638	73.88	73.88	<.0001	<.0001	1	327	132.23	<.0001
Depth	1	638	61.63	61.63	<.0001	<.0001	dropped			

Figures

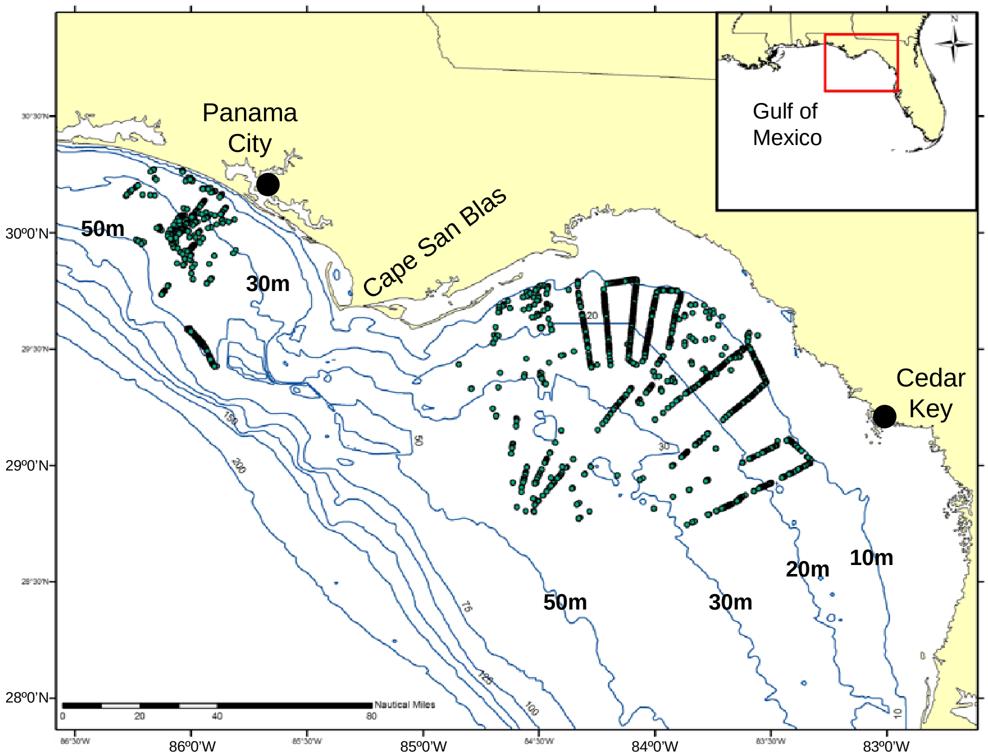


Figure 1. Locations of all natural reefs in the sampling universe of the Panama City NMFS reef fish video survey as of March 2012. Total sites: 2359, 722 west of and 1637 east of Cape San Blas.

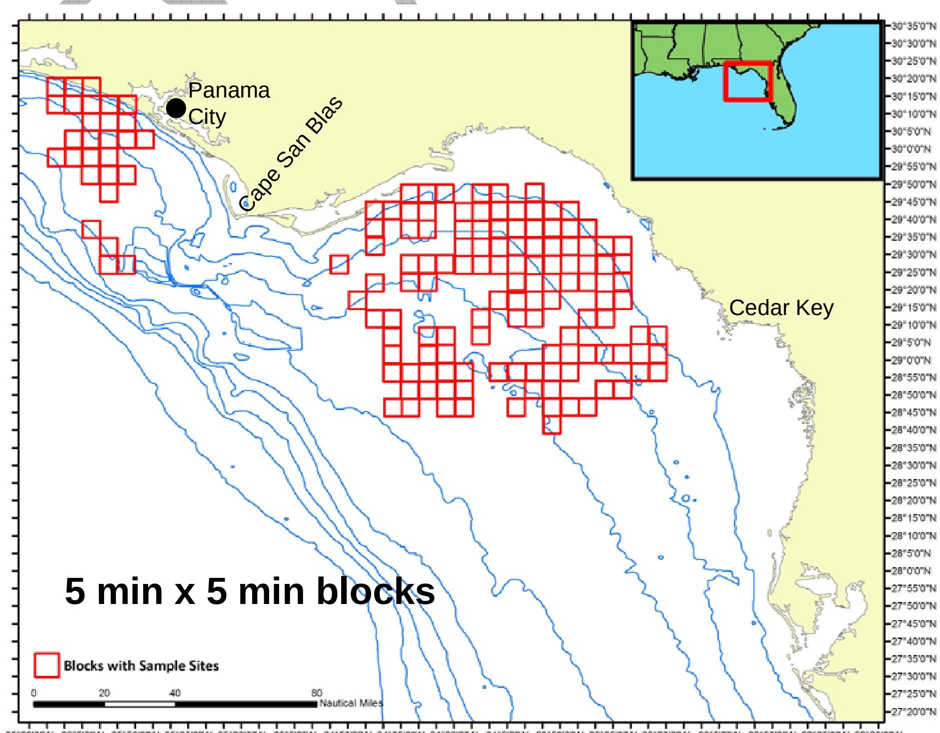


Figure 2. Sampling blocks, as of 2012, of the Panama City reef fish survey.

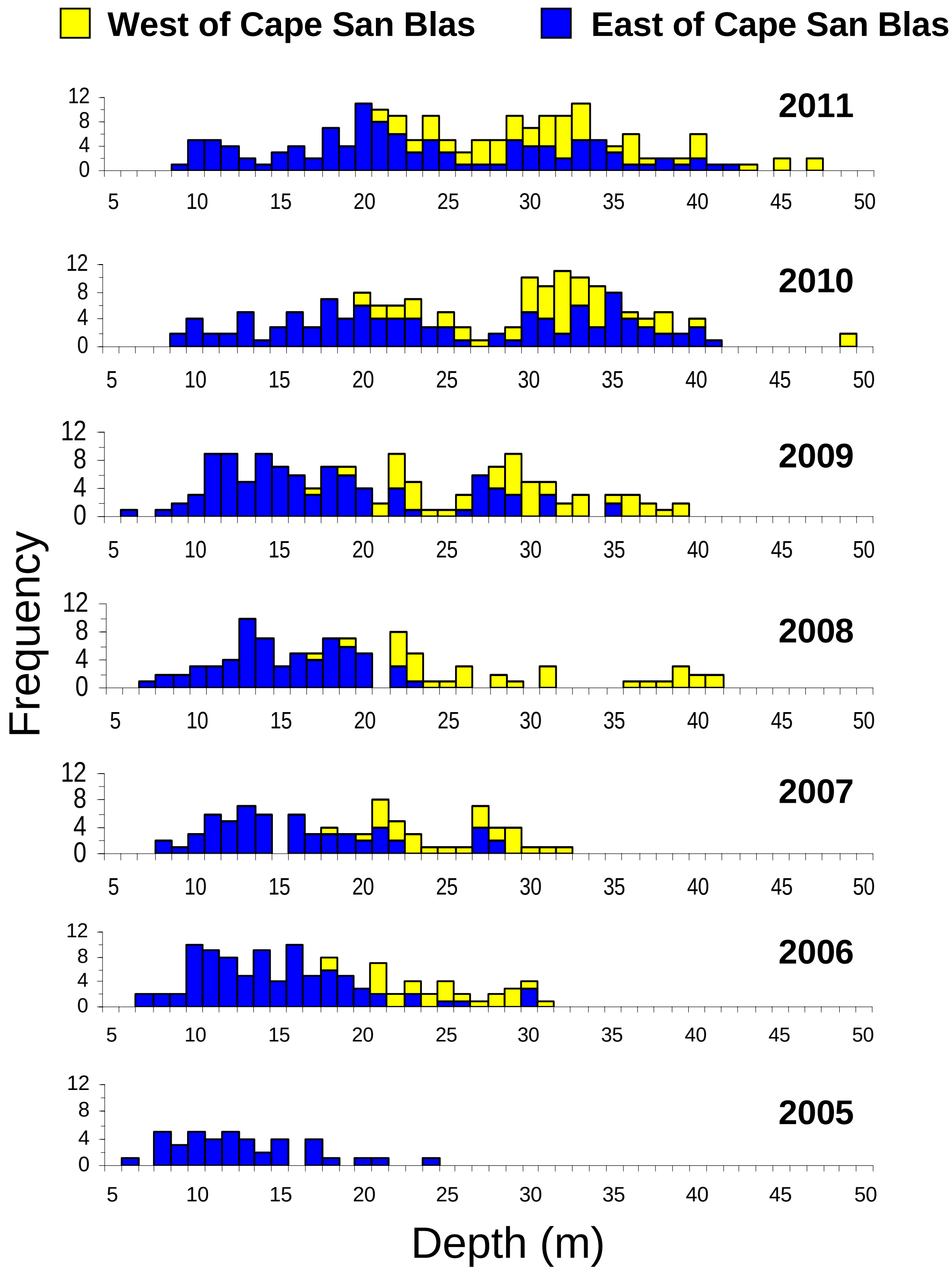


Figure 3. Annual depth distribution of Panama City reef fish survey video sample sites east and west of Cape San Blas, 2005-2011.

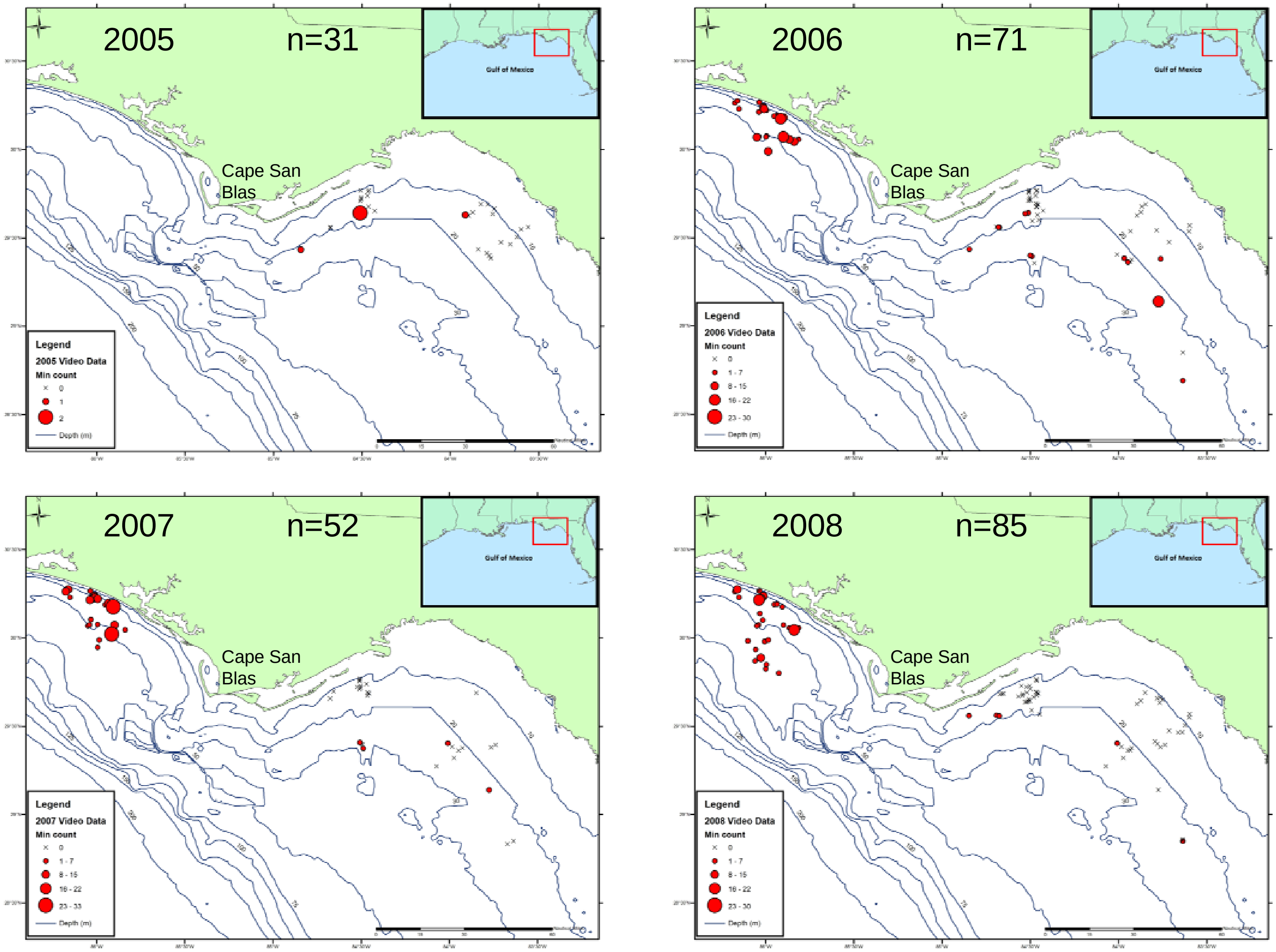


Figure 4. Annual distribution and relative abundance (min counts) of red snapper observed in the Panama City NMFS reef fish survey, 2005-2008, with stationary, high definition video cameras. Sites sampled with video gear, but where no red snapper were observed, are indicated with an X. Sample sizes refer to the total number of sites surveyed.

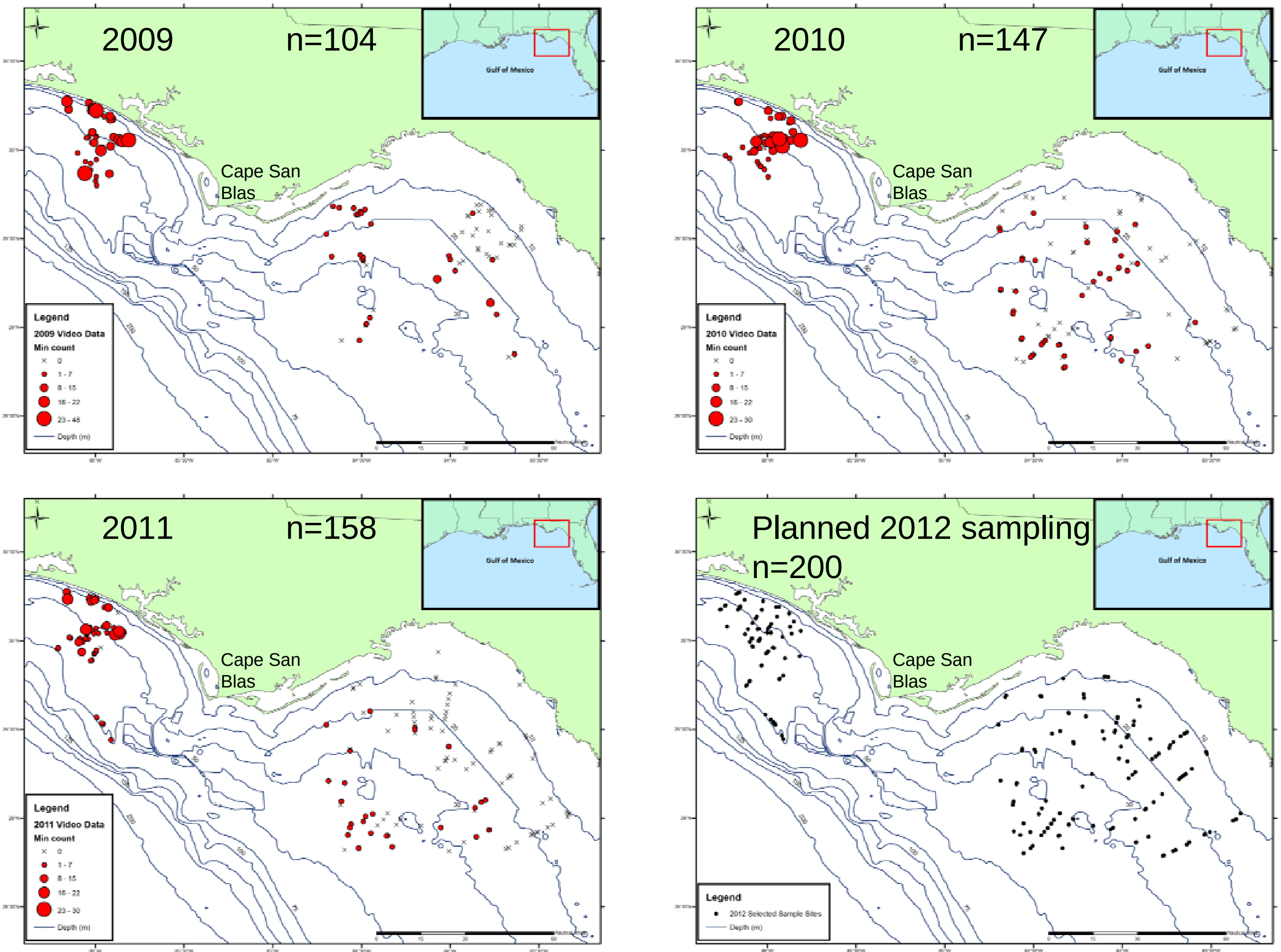


Figure 5. Annual distribution and relative abundance (min counts) of red snapper observed in the Panama City NMFS reef fish survey, 2009-2011, with stationary, high definition video or mpeg cameras. Sites sampled with video gear, but where no red snapper were observed, are indicated with an X. Sample sizes refer to the total number of sites surveyed.

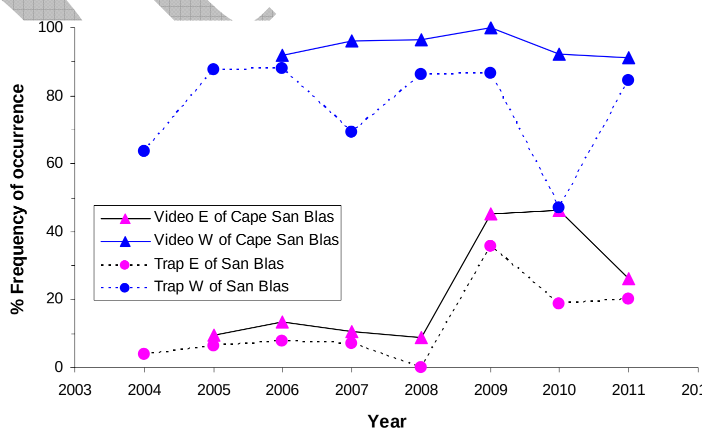


Figure 6. Annual percent frequency of occurrence of red snapper in video and trap samples east and west of Cape San Blas, 2005-2011. All data was included for trap estimates; censored data sets were used to calculate video frequencies.

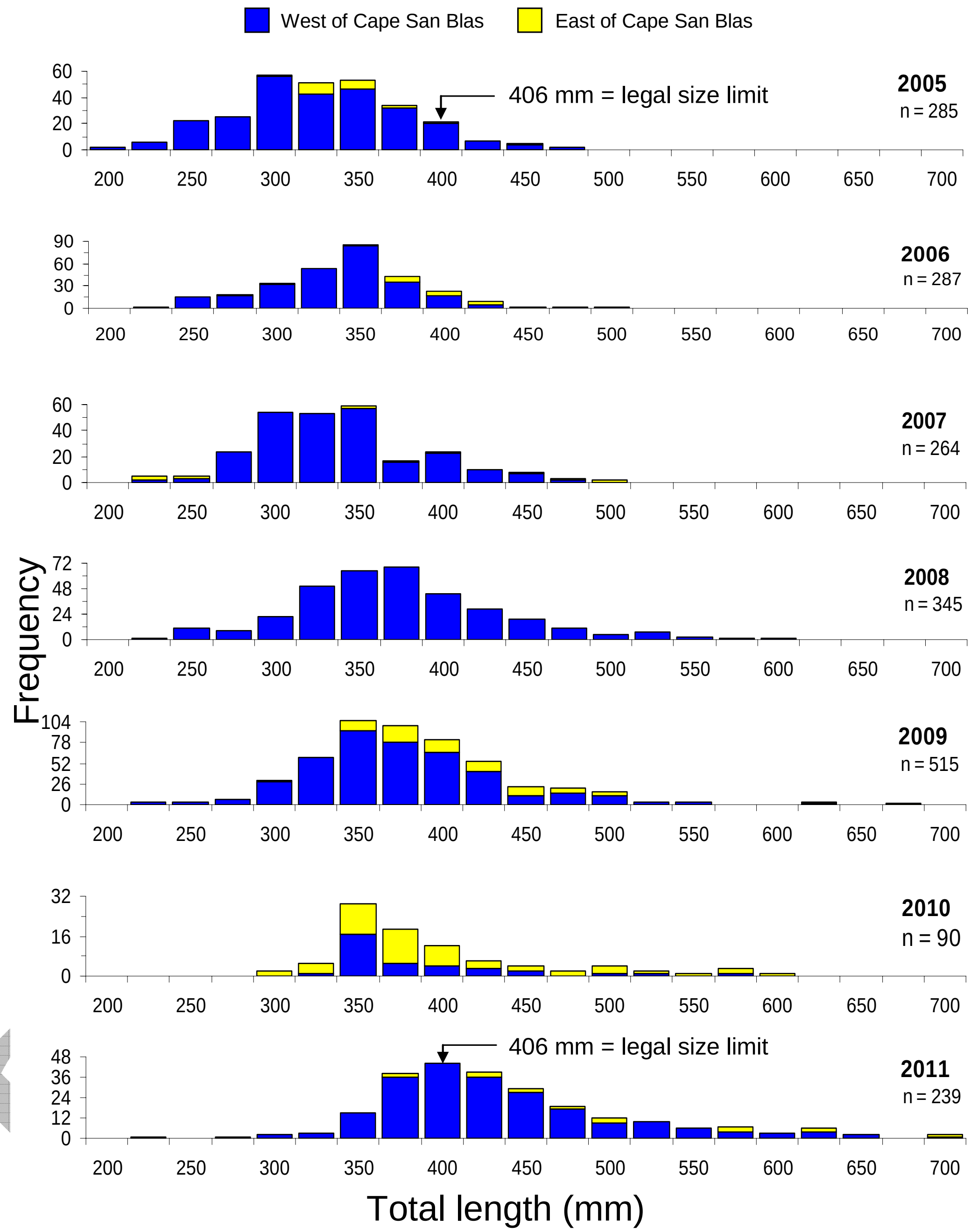
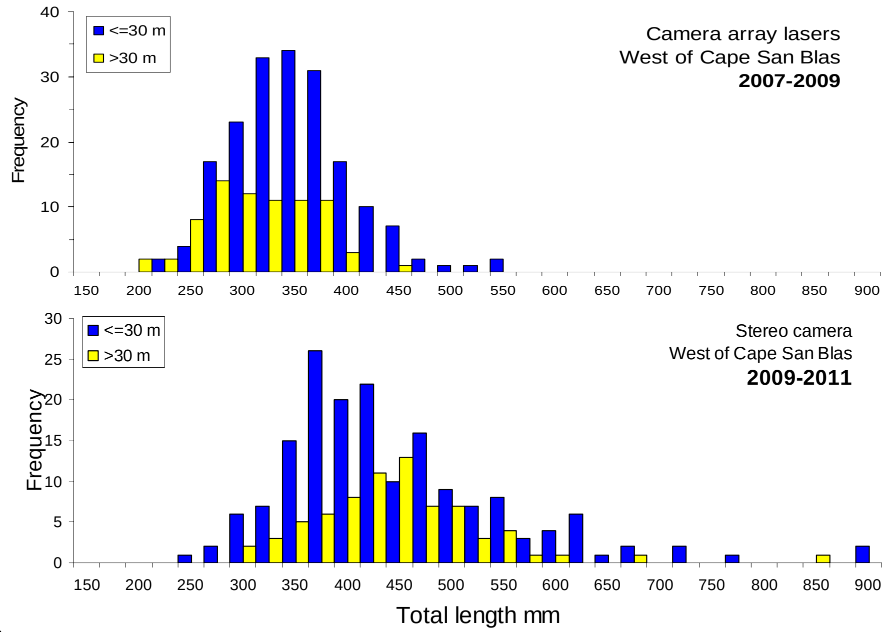


Figure 7. Annual size structure of trap-caught red snapper east and west of Cape San Blas, 2005-2011.



.
Figure 8. Size structure of red snapper by depth zone (<=30m and >30m) observed west of Cape San Blas based on scaling lasers, 2007-2009, and from stereo camera images, 2009-2011.

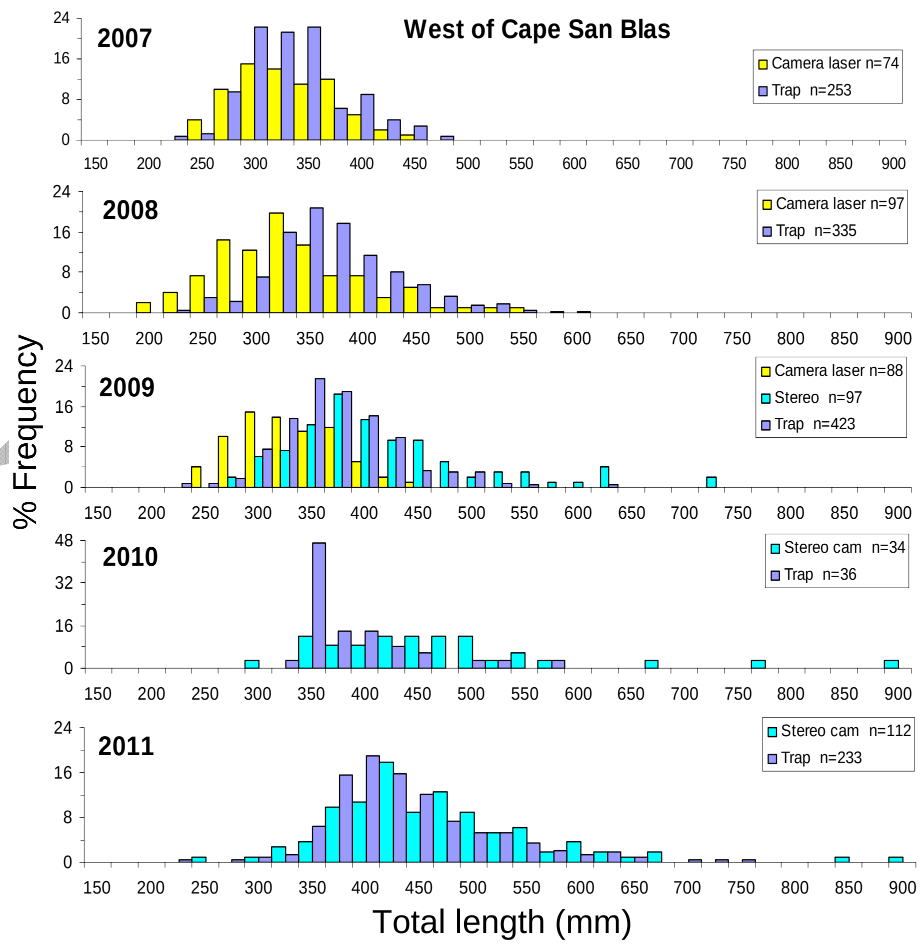


Figure 9. Annual size distributions of red snapper west of Cape San Blas, 2007-2011 collected in chevron traps or measured in high definition video or stereo still images.

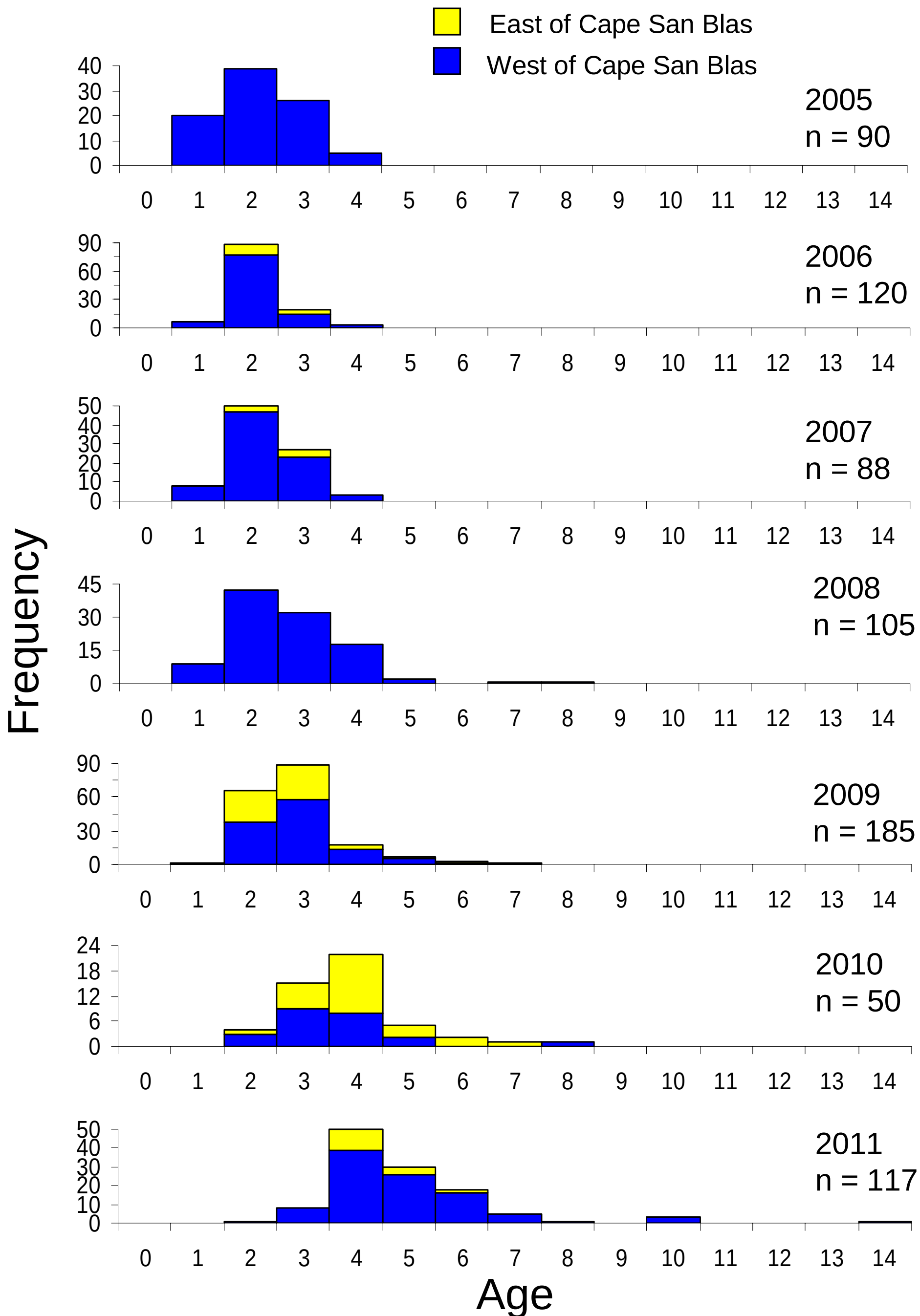
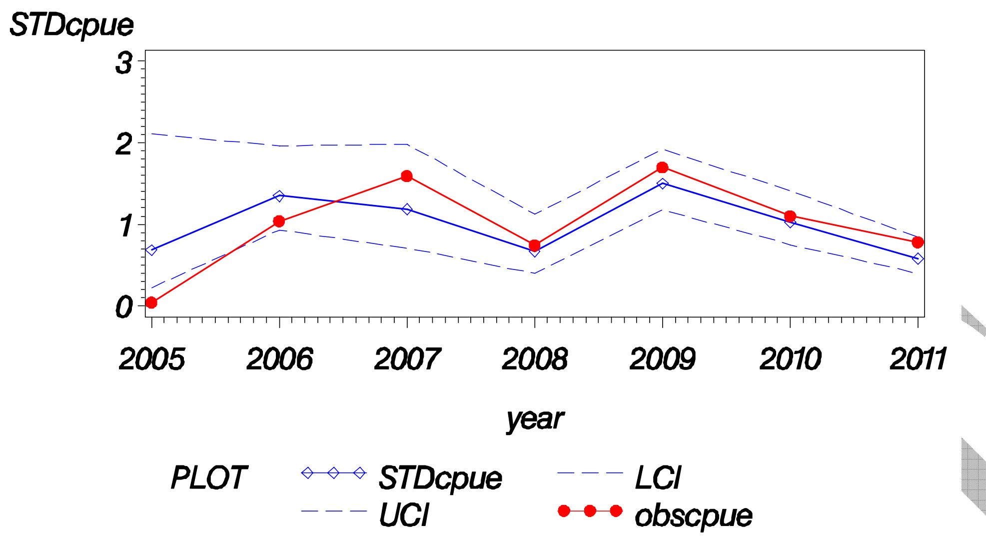


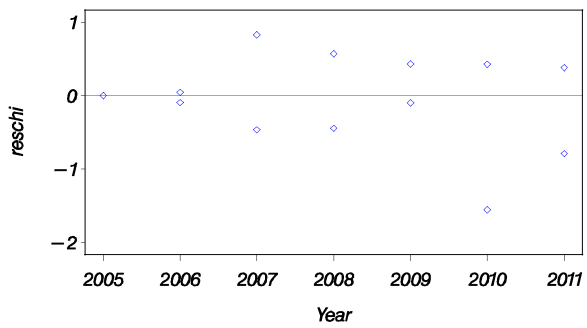
Figure 10. Annual age structure of trap-caught red snapper, 2005-2011, east and west of Cape San Blas.



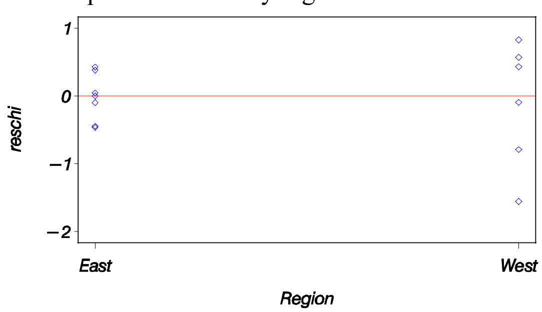
Survey Year	*Frequency*	*N*	*Index*	*Scaled Index*	*CV*	*LCL*	*UCL*
2005	0.09677	31	2.60619	0.68621	0.60876	0.22324	2.10931
2006	0.49296	71	5.12543	1.34953	0.18824	0.92917	1.96005
2007	0.51923	52	4.50183	1.18533	0.26125	0.70902	1.98164
2008	0.38824	85	2.54182	0.66926	0.26481	0.39762	1.12648
2009	0.67308	104	5.70097	1.50107	0.12231	1.17641	1.91532
2010	0.62585	147	3.90904	1.02925	0.15843	0.75121	1.41020
2011	0.50633	158	2.20034	0.57935	0.18720	0.39970	0.83974

Figure 11. PC Video abundance indices for red snapper. 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 table above, 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.

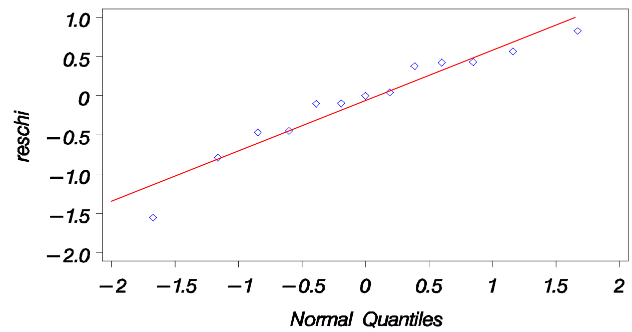
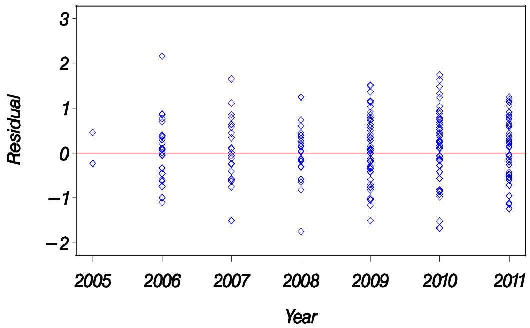
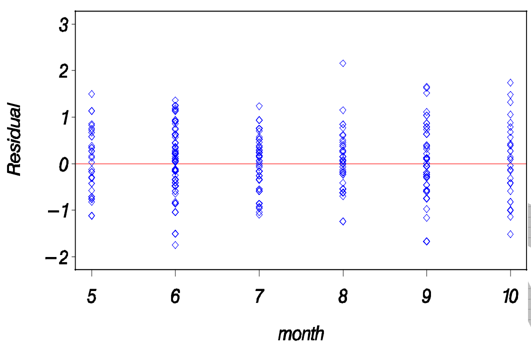


Figure 12. Diagnostic residual plots of the binomial submodel for red snapper observed during PC Video Surveys in the northeastern Gulf of Mexico.

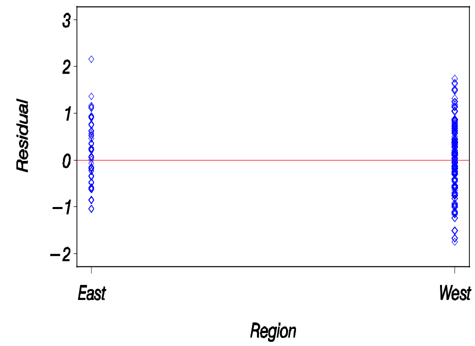
a. Chi-square residuals by year.



b. Chi-square residuals by month.



c. Chi-square residuals by region.



d. QQplot of chi-square residuals.

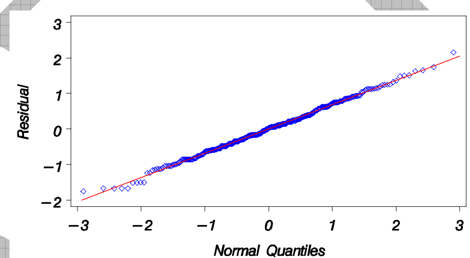


Figure 13. Diagnostic residual plots of the lognormal submodel for red snapper observed during PC Video Surveys in the northeastern Gulf of Mexico.