Almaco jack *Seriola rivoliana* Findings from the NMFS Panama City Laboratory Trap & Camera Fishery-Independent Survey 2004-2014

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SEDAR49-DW-15

29 April 2016

Please cite this document as:

DeVries, D.A., C.L. Gardner, P. Raley, and K. Overly. 2016. Almaco jack *Seriola rivoliana* Findings from the NMFS Panama City Laboratory Trap & Camera Fishery-Independent Survey 2004-2014. SEDAR49-DW-15. SEDAR, North Charleston, SC. 18 pp.

Almaco jack *Seriola rivoliana* Findings from the NMFS Panama City Laboratory Trap & Camera Fishery-Independent Survey 2004-2014

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April 2016

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Contribution 16-05

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 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 random after side scan sonar surveys that year yielded an order of magnitude increase in that universe (Figure 1). Five by five minute blocks known to contain 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 chosen first. Then two known reef sites a minimum of 250 m apart within each selected block are randomly selected (Figure 2). Alternates are also selected for use when another boat is found to be fishing the site or no hard bottom can be found with sonar at that site.

Depth coverage was ~8-30 m during 2004-07, and then subsequently steadily expanded to ~8–52 m (Figure 3). Sampling effort has also increased since 2004. Sample sizes were 59 in 2004 (33 West: 26 East), 101 in '05 (24 W: 77 E), 114 in '06 (25 W: 89 E), 86 in '07 (29 W: 57 E), 97 in '08 (31 W: 66 E), 143 in '09 (47 W: 96 E), 162 in '10 (53 W: 109 E), 180 in '11 (65 W: 115 E), 178 in '12 (61 W: 117 E), 112 in 2013 (71 W: 41 E), and 184 in 2014 (113E: 71 W). Nine sites in 2004 and 23 in 2005 were sampled twice; thereafter each site was only sampled once in a given year. All sampling occurred between May and October (with the exception of four sites in November, 2013), but primarily during June through August (Figure 4). Sampling east of Cape San Blas in 2013 was greatly reduced (down ~66%) and done later than normal (Oct. and Nov.) due to late receipt of funding, ship mechanical issues, and weather problems.

Methods

Sampling was conducted during daytime from 1 hr after sunrise until 1 hr before sunset. Chevron traps were freshly baited each set with 3 previously frozen Atlantic mackerel *Scomber scombrus*, and soaked for 1.5 hr. Traps were fished 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 and fork length (FL only for gray triggerfish and TL only for black seabass). Both sagittal otoliths were 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 Hi 8 video cameras (2005 only) 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 was added, reducing the number of HDEFs to two; and both SIS's were also upgraded with HDEF, color mpeg cameras. In 2012 the two HDEFs were replaced with hi-def GoPro cameras. The camera array was unbaited 2005-2008, 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 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. 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 4 HDEF 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 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. Beginning in 2012, when a video from a GoPro camera was selected to be read, because they have a much larger field of view than the SIS MPEGs (122 vs 65º), predetermined, equal portions of each edge of the video monitor were covered so that only the central 65° of the field of view was visible. 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 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 (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, or MaxN; Ellis and DeMartini 1995), and VMS 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 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 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, bad camera angle, video out of focus, etc., were censored (excluded) from calculations of relative abundance. In 2014, 10 video samples from an area with an ongoing serious red tide bloom, and which showed no or virtually no evidence of living fish, were also censored. As a result of this screening, of the video samples from east of the Cape, only 41 of 41 in 2005, 84 of 89 in 2006, 48 of 57 in 2007, 61 of 66 in 2008, 68 of 97 in 2009, 97 of 109 in 2010, 100 of 115, in 2011, and 105 of 115 in 2012, 38 of 39 in 2013, and 103 of 113 in 2014 met the reef and visibility criteria and were retained. In contrast, west of the Cape, 25 of 25 sites in 2006, 24 of 29 in 2007, 29 of 31 in 2008, 44 of 47 in 2009, 50 of 53 in 2010, 60 of 64 in 2011, 53 of 59 in 2012, 67 of 72 in 2013, and 71 of 71 in 2014 were retained for analyses.

Results

Since the Panama City lab reef fish video survey began in 2005, almaco jacks *Seriola rivoliana* have consistently and relatively commonly been observed with stationary video gear across the inner and mid-West Florida shelf both east and west of Cape San Blas (Table 1, Fig. 5) (DeVries et al. 2008, 2009, 2012). In contrast, almaco jacks have only been captured in the Panama City lab trap survey once out of 875 deployments from 2004 through 2015 (one individual in 2014), so there are no trap results to present herein. Although almaco jacks have been frequently observed swimming with schools of greater amberjack *S. dumerili* (personal observation by D. DeVries and C. Gardner), our survey has found almost no evidence that this species forms monospecific schools; 71% of positive samples had min counts of one, 17% had two fish, and the highest count was 7 (2 collections composing only 3% of all positive sites) (Fig. 6).

Encounter rates: Almaco jacks were never encountered in video samples shallower than 15.3 m and in only 1 out of 346 samples from depths < 18.2 m (where 30% of all sampling occurred) (Fig. 5 and 7). Encounter rates increased noticeably at the 20 m depth interval, with proportion positives fluctuating between 0.06 and 0.09 to the depth limits of the survey (Fig. 7). Because of the almost complete absence of almaco jacks in our shallow water samples, data summaries are presented both for collections from all depths and for collections only from depths ≥18.2 m.

Almaco jacks were more commonly encountered west of Cape San Blas than east, even when data from <18.2 m depths were excluded. During 2006-2014 the mean annual proportion of positive video samples in depths ≥ 18.2 m was 0.11 (range 0.04 to 0.30) west of Cape San Blas and 0.06 (range 0.00-0.13) in the east (Table 1, Fig. 8); with data from all depths included, those proportion remained unchanged in the west (0.11, range 0.04 to 0.29) but dropped 50% to 0.03 (range 0.00-0.05) in the east (Table 1, Fig. 8). Certainly some of the difference between regions was related to the much larger proportion of shallow (<20 m) sites sampled east of the Cape prior to 2009 (Fig. 3), as exclusion of all sites <18.2 m from the analysis virtually eliminated those differences in 2007 and 2008. However, after 2009, even with the addition of many deep sites in the east and the exclusion of all sites <18.2 m, the proportion of positive observations remained higher west of the Cape than east every year except 2012, averaging 0.09 and 0.04, respectively, 2010-2014 (Fig. 8). The overall mean proportion of positive video samples for east and west combined, 2006-2014, was 0.06 for all depths and 0.09 for depths ≥ 18.2 m (Fig. 9). After the extension of sampling into deeper waters east of Cape San Blas starting in 2009, the trends in proportion positives were basically identical whether sites from all depths were included or just those >18.1 m, although the latter averaged 22% higher (range 19-28%) (Fig. 9).

Abundance trends: Along with higher encounter rates, relative abundance estimates of almaco jack were also higher west of the Cape than east every year except 2012 when data from all depths were examined, and every year except 2009 and 2012 if only depths ≥18.2 m were included (Fig. 10). Using data from all depths, mean video min count for 2006-2014 was 0.17 (range 0.05-0.50) west of the Cape and 0.05 (range 0.0-0.16) in the east. If only data from depths ≥18.2 m were included, mean min count for 2006-2014 west of the Cape was unchanged, i.e., 0.17 (range 0.05-0.50); but east of the Cape the mean increased to 0.12 (range 0.0-0.5), over twice the estimate of 0.5 when all depths were included. As with proportion positives, the differences between east and west narrowed greatly in 2006 and basically disappeared in 2007 and 2008 when the large number of shallow sites <18.2 m were excluded (Fig. 10). Excluding depths <18.2 m brought the mean difference between east and west of the Cape during 2006-2008 down an order of magnitude - from 428% (range 275-717) to 47% (range 4-113).

Relative abundance of almaco jack on the northern West Florida Shelf increased sharply in 2008, suggesting the recruitment of a new, strong year class, but it then dropped back to a much lower level in 2009 and stayed at that level through 2011 (Fig. 11). Relative abundance increased moderately in 2012, remained level in 2013, then fell in 2014 to levels only slightly higher than those observed from 2009 through 2011 (Fig. 11). The similar trends in relative abundance east and west of the Cape, especially the very similar sharp rise in 2008, suggests that the almaco jacks in these two regions belong to same stock. Since 2009, the trends in the overall combined east and west data set were identical whether all depths were included or only depths ≥18.2 m; and mean nominal min counts from the ≥18.2 m data set averaged only 22% larger than those from the all depths data set (Fig. 11).

Size structure: Almaco jacks observed with stereo cameras during 2009-2014 ranged from 227 to 480 mm FL and averaged 372 mm FL (Std error = 16)(Fig. 12). Mean sizes from stereo image measurements were slightly larger east of Cape San Blas than west — 399 vs 349 mm FL, although sample sizes were small (n=12 east and n=13 west) and 95% confidence intervals overlapped considerably. There appeared to be very little relationship between size of almaco jack and depth (Fig. 13). No attempt was made to model the relationship because of the small sample size and the very unequal distribution of data points across the depth range.

Annual GIS plots of video min counts show the spatial and temporal distributions of almaco jack for 2005 – 2014 (Fig. 14), but small sample sizes make it difficult draw any detailed conclusions from them.

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.

DeVries, D.A., C.L. Gardner, P. Raley, and W. Ingram. 2012. NMFS Panama City Laboratorytrap and camera survey for reef fish. Annual report of 2011 results. Panama City Laboratory

Ellis, D.M., and DeMartini, E.E. 1995. Evaluation of a video camera technique for indexing abundances of juvenile pink snapper, *Pristipomoides filamentosus*, and other Hawaiian insular shelf fishes. Fish. Bull. 93(1): 67–441 77.

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.

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 video survey sample sizes, proportion positive occurrences, mean nominal video min counts, and standard errors of almaco jack east and west of Cape San Blas, 2006-2014, for all depths (A) and for all depths ≥18 m (B). Estimates calculated using censored data sets (see Methods).

A. All depths included

	Total sites sampled			Proportion positive occurrences			Mean nominal min count			Standard error		
Year	East	West	Total	East	West	Total	East	West	Total	East	West	Total
2006	84	24	108	0.02	0.29	0.08	0.036	0.292	0.093	0.026	0.095	0.031
2007	47	24	71	0.02	0.08	0.04	0.021	0.083	0.042	0.021	0.058	0.024
2008	60	28	88	0.03	0.14	0.07	0.133	0.500	0.250	0.118	0.289	0.122
2009	68	38	106	0.04	0.05	0.05	0.044	0.053	0.047	0.025	0.037	0.021
2010	97	50	147	0.00	0.10	0.03	0.000	0.140	0.048	0.000	0.064	0.022
2011	99	58	157	0.00	0.07	0.03	0.000	0.086	0.032	0.000	0.045	0.017
2012	105	53	158	0.05	0.04	0.04	0.162	0.057	0.127	0.081	0.042	0.056
2013	34	60	94	0.03	0.13	0.10	0.029	0.200	0.138	0.029	0.074	0.049
2014	96	72	168	0.04	0.10	0.07	0.042	0.111	0.071	0.021	0.042	0.022
Total	690	407	1097	0.03	0.10	0.05	0.051	0.147	0.085	0.017	0.028	0.015

B. Depths <18.2 m excluded

	Total sites sampled			Proportion positive occurrences			Mean nominal min count			Standard error		
Year	East	West	Total	East	West	Total	East	West	Total	East	West	Total
2006	21	23	44	0.10	0.30	0.20	0.143	0.304	0.227	0.104	0.098	0.072
2007	15	24	39	0.07	0.08	0.08	0.067	0.083	0.077	0.067	0.058	0.043
2008	16	27	43	0.13	0.15	0.14	0.500	0.519	0.512	0.438	0.299	0.245
2009	32	37	69	0.06	0.05	0.06	0.063	0.054	0.058	0.043	0.038	0.028
2010	71	50	121	0.00	0.10	0.04	0.000	0.140	0.058	0.000	0.064	0.027
2011	65	58	123	0.00	0.07	0.03	0.000	0.086	0.041	0.000	0.045	0.021
2012	80	53	133	0.06	0.04	0.05	0.213	0.057	0.150	0.106	0.042	0.066
2013	17	60	77	0.06	0.13	0.12	0.059	0.200	0.169	0.059	0.074	0.060
2014	67	71	138	0.06	0.10	0.08	0.060	0.113	0.087	0.029	0.043	0.026
Total	**384**	**403**	**787**	**0.04**	**0.10**	**0.07**	**0.093**	**0.149**	**0.121**	**0.030**	**0.028**	**0.021**

Figures

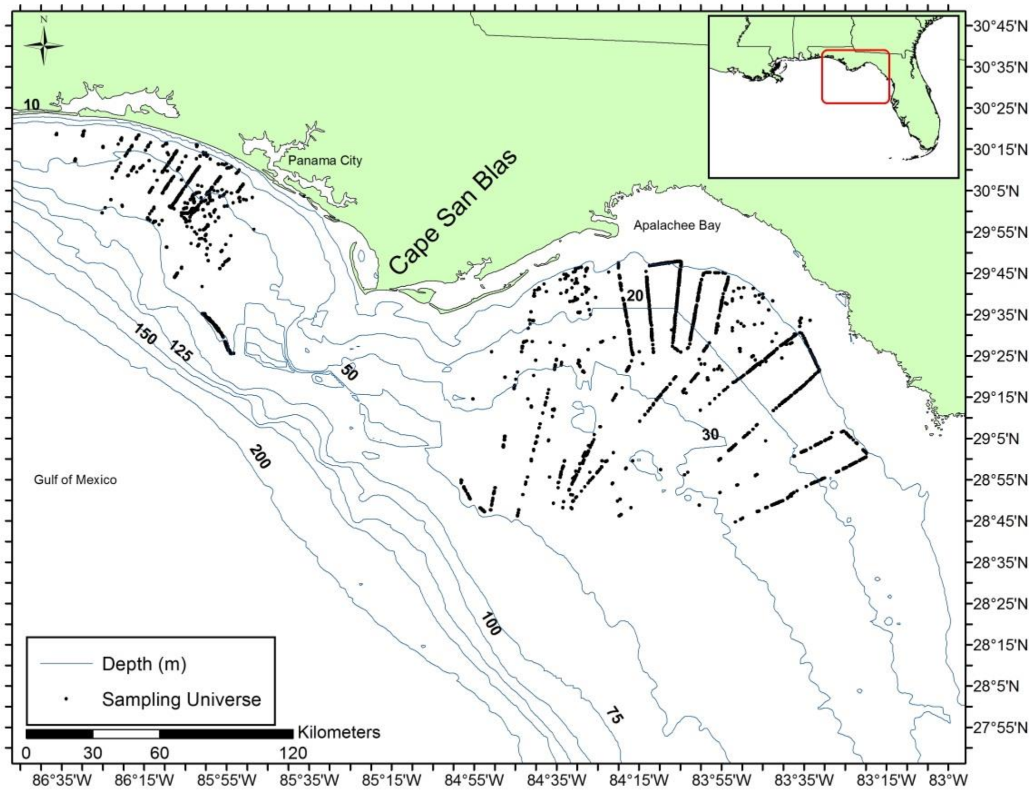


Figure 1. Locations of all natural reefs in the sampling universe of the Panama City NMFS reef fish video survey as of November 2014. Total sites: 2985 – 1105 west of and 1880 east of Cape San Blas. Isobath labels are in meters.

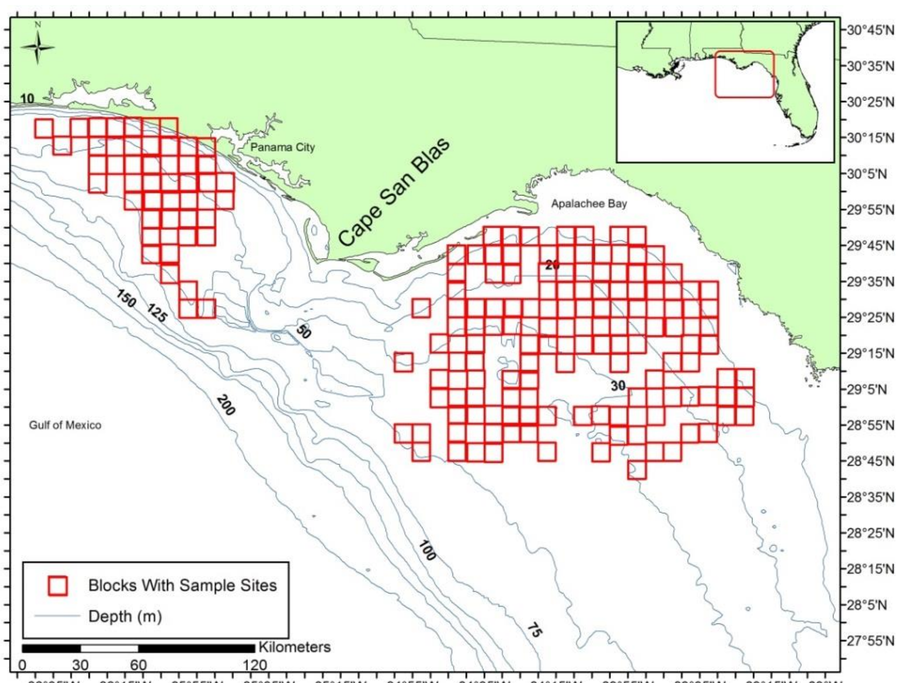


Figure 2. Sampling blocks (5 min lat. x 5 min. long.) of the Panama City reef fish survey as of 2014. Isobath labels are in meters.

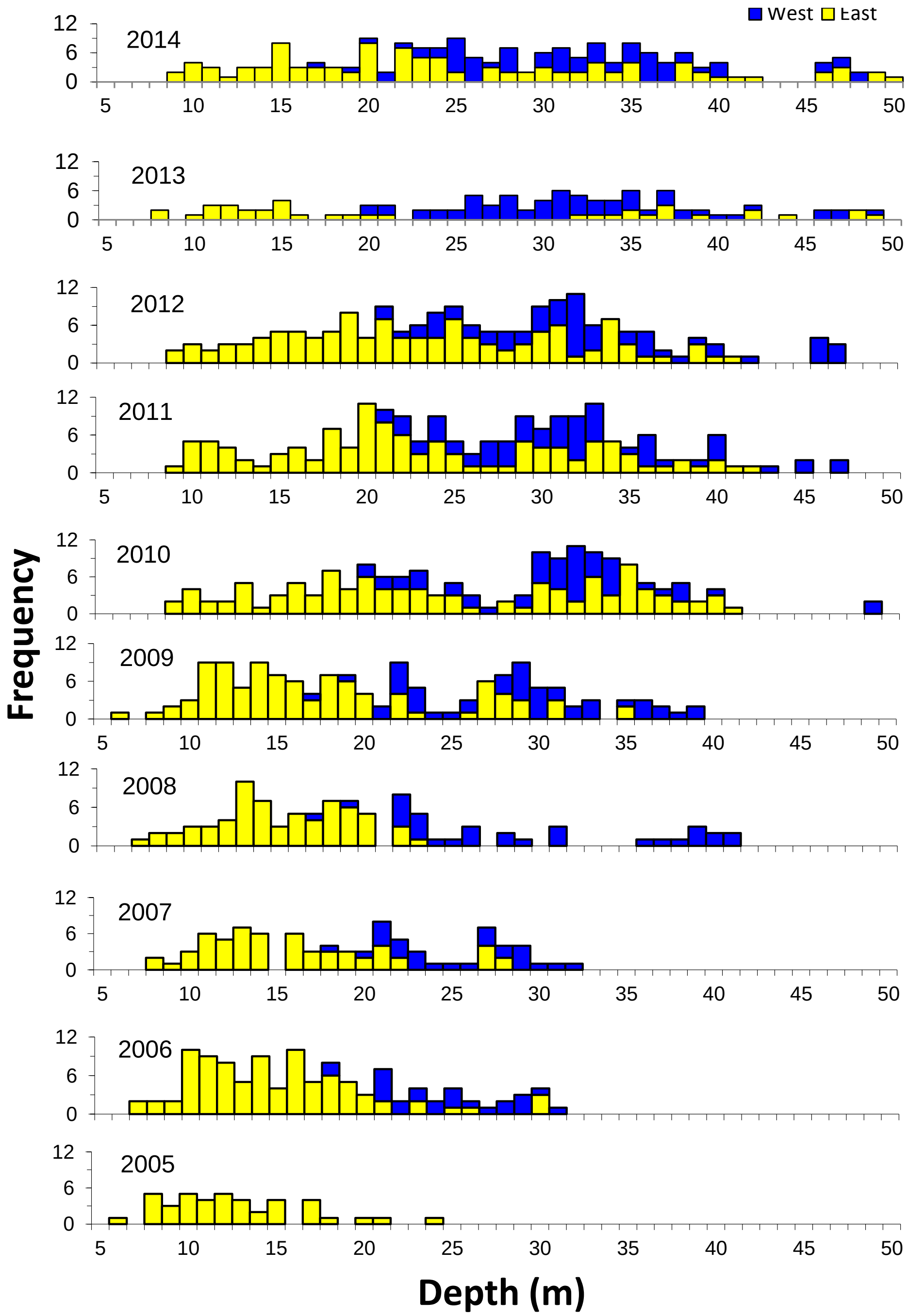


Figure 3. Annual depth distribution of Panama City reef fish survey video sample sites east and west of Cape San Blas, 2005-2014.

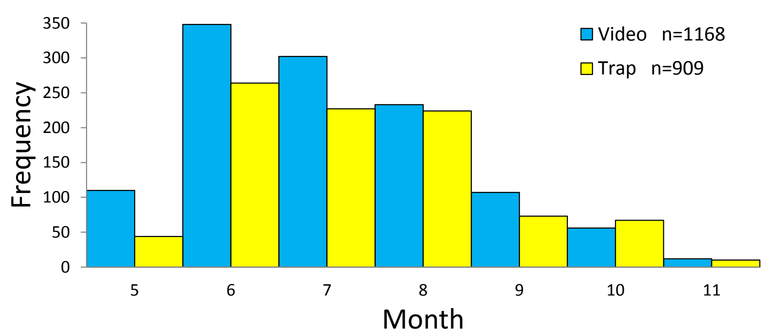


Figure 4. Overall monthly distribution of Panama City reef fish survey video and trap samples (censored data sets only), 2004-2014.

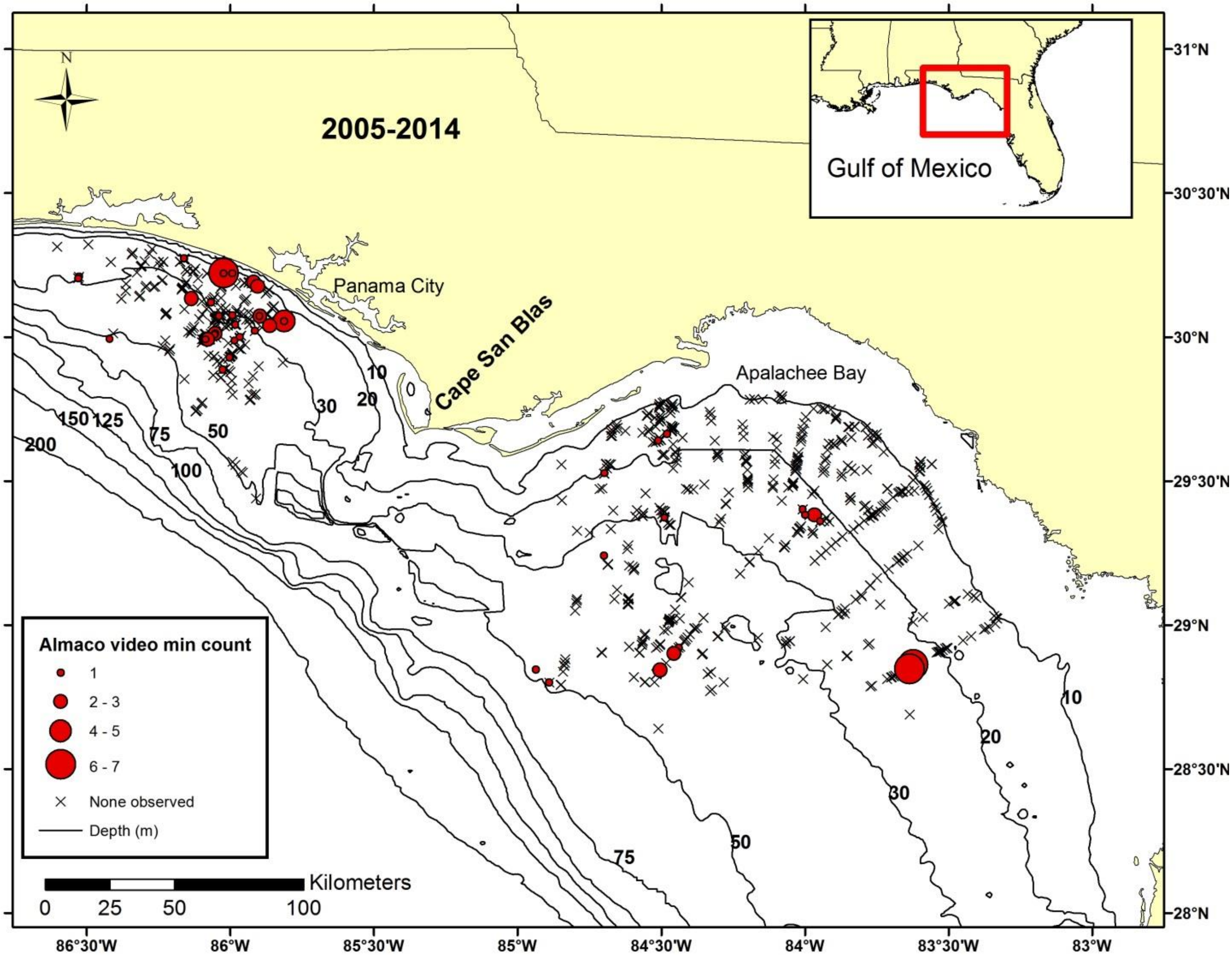


Figure 5. Distribution and relative abundance of almaco jack observed with stationary, high definition video or mpeg cameras (min counts) in the Panama City NMFS reef fish survey, 2004-2014. Sites sampled, but where no almaco jack were caught or observed, are indicated with an X.

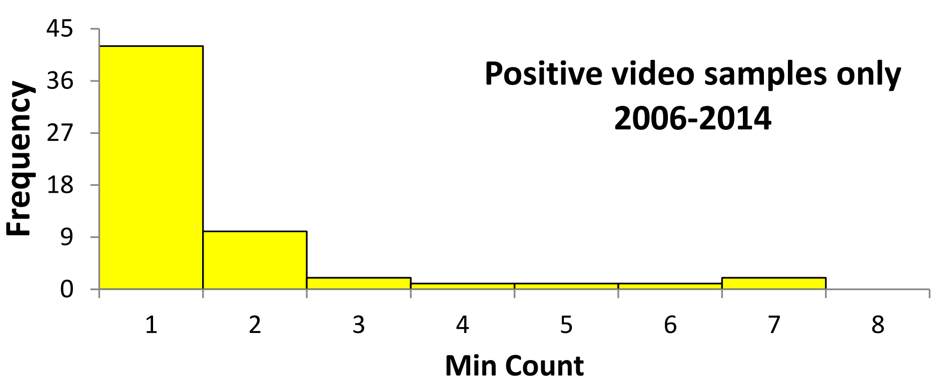


Figure 6. Frequency distribution of non-zero min counts of almaco jack from Panama City reef fish video samples, 2006-2014.

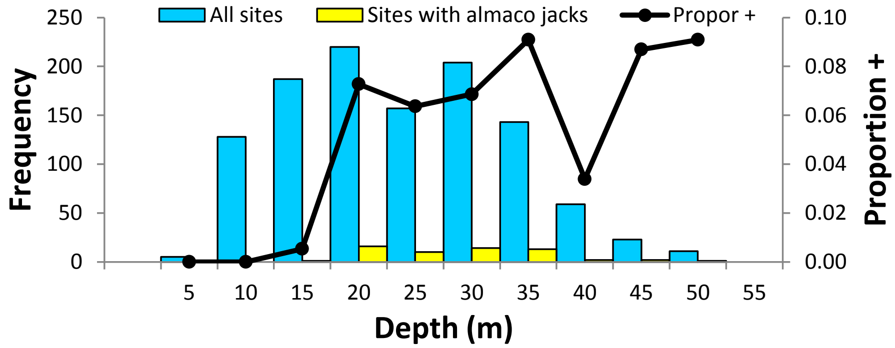


Figure 7. Depth distributions of all video (2004-2014) sample sites vs only sites positive for almaco jack; and overall proportions of positive almaco jack video samples, 2004-2014, by depth.

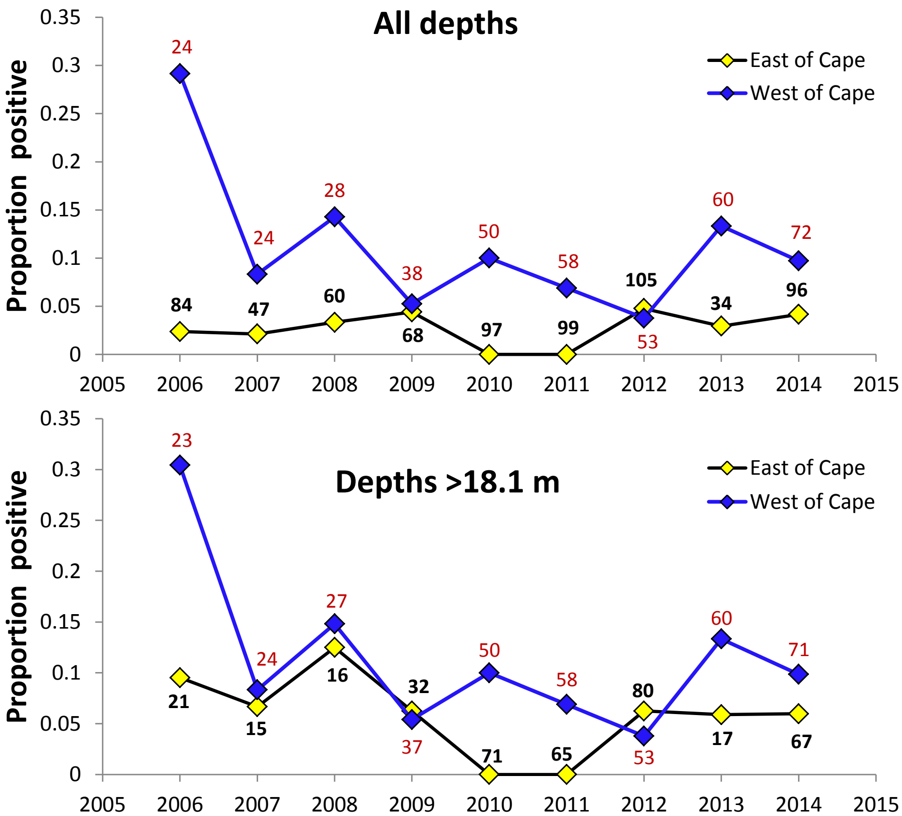


Figure 8. Annual proportions of positive almaco jack video samples, 2006-14, by area (east vs. west of Cape San Blas) based on samples from all depths (upper panel) and on samples only from depths ≥18.1 m (lower panel). Numbers within the plot are total (not just positive) sample sizes for each year.

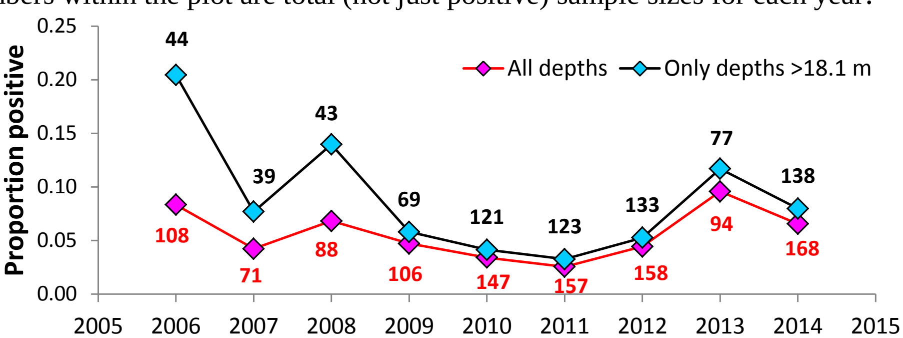


Figure 9. Annual proportions of positive almaco jack video samples, 2006-14, east + west of Cape San Blas, based on samples from all depths and on samples only from depths ≥18.2 m . Numbers within the plot are total (not just positive) sample sizes for each year.

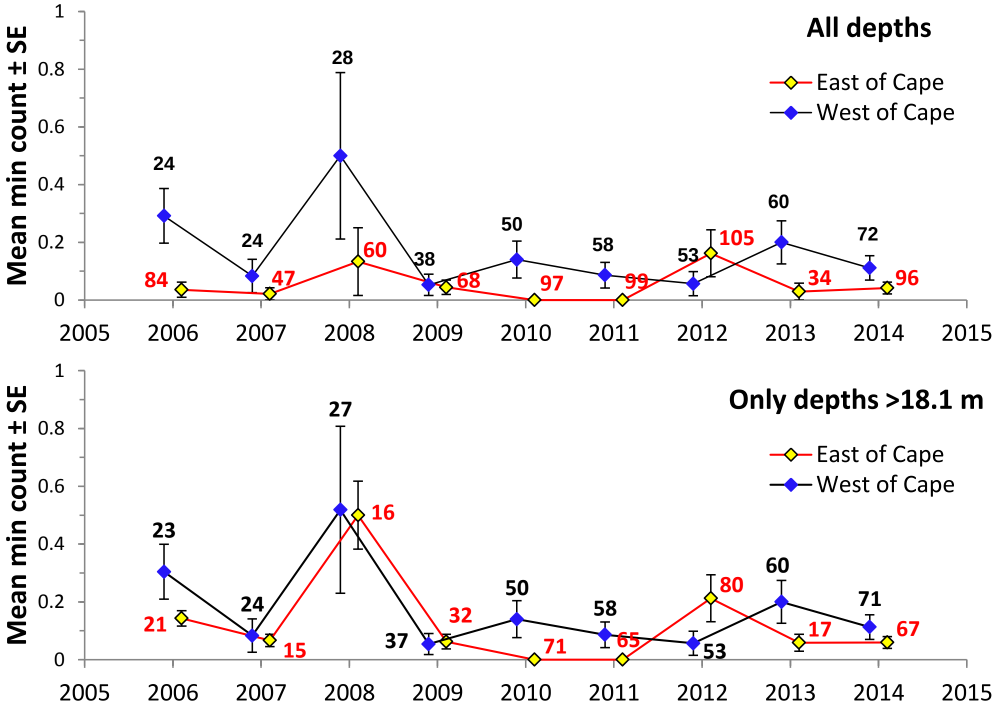


Figure 10. Mean annual nominal video min counts (MaxN) of almaco jack east and west of Cape San Blas, 2006-2014, based on samples from all depths and from only depths ≥ 18.1 m. Numbers within the plot are sample sizes for each year.

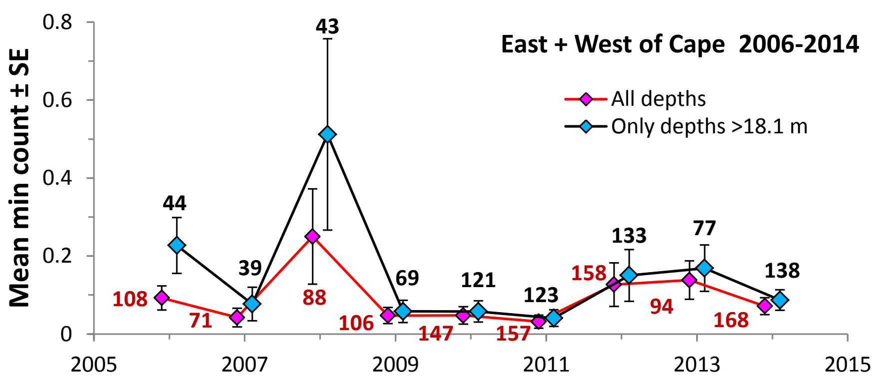


Figure 11. Mean annual nominal video min counts (MaxN) of almaco jack for regions combined, 2006-2014, based on samples from all depths and from only depths ≥ 18.1 m. Numbers within the plot are sample sizes for each year.

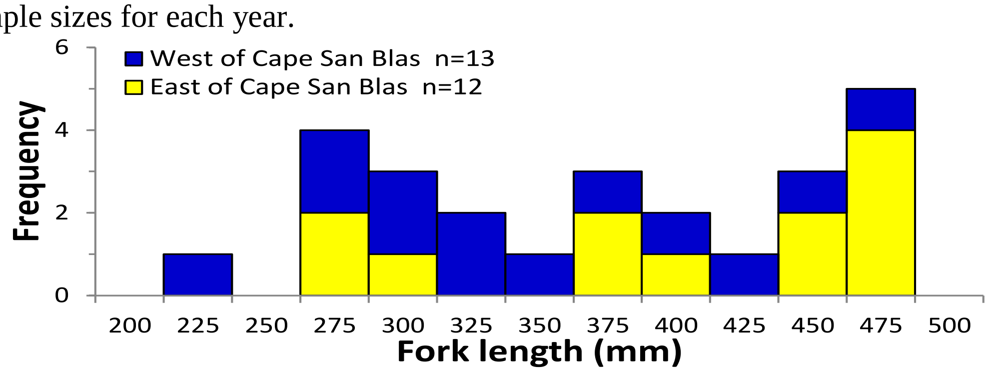


Figure 12. Overall size distributions of all almaco jacks measured in stereo images, 2009-2014, by region.

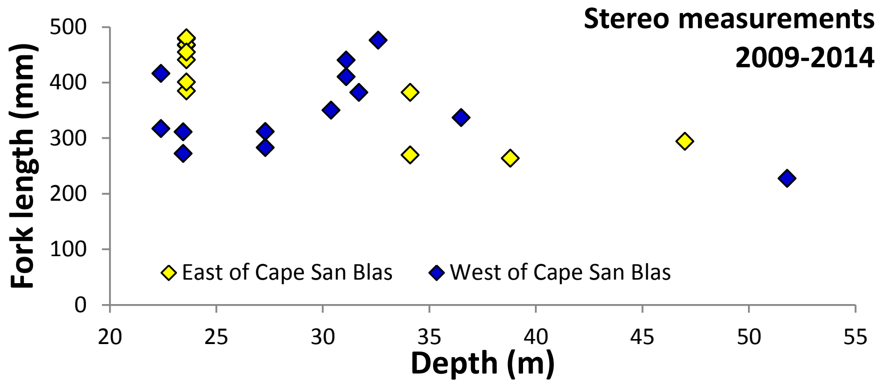


Figure 13. Fork length vs. depth relationship of almaco jacks east and west of Cape San Blas observed with stereo cameras, 2009-14.

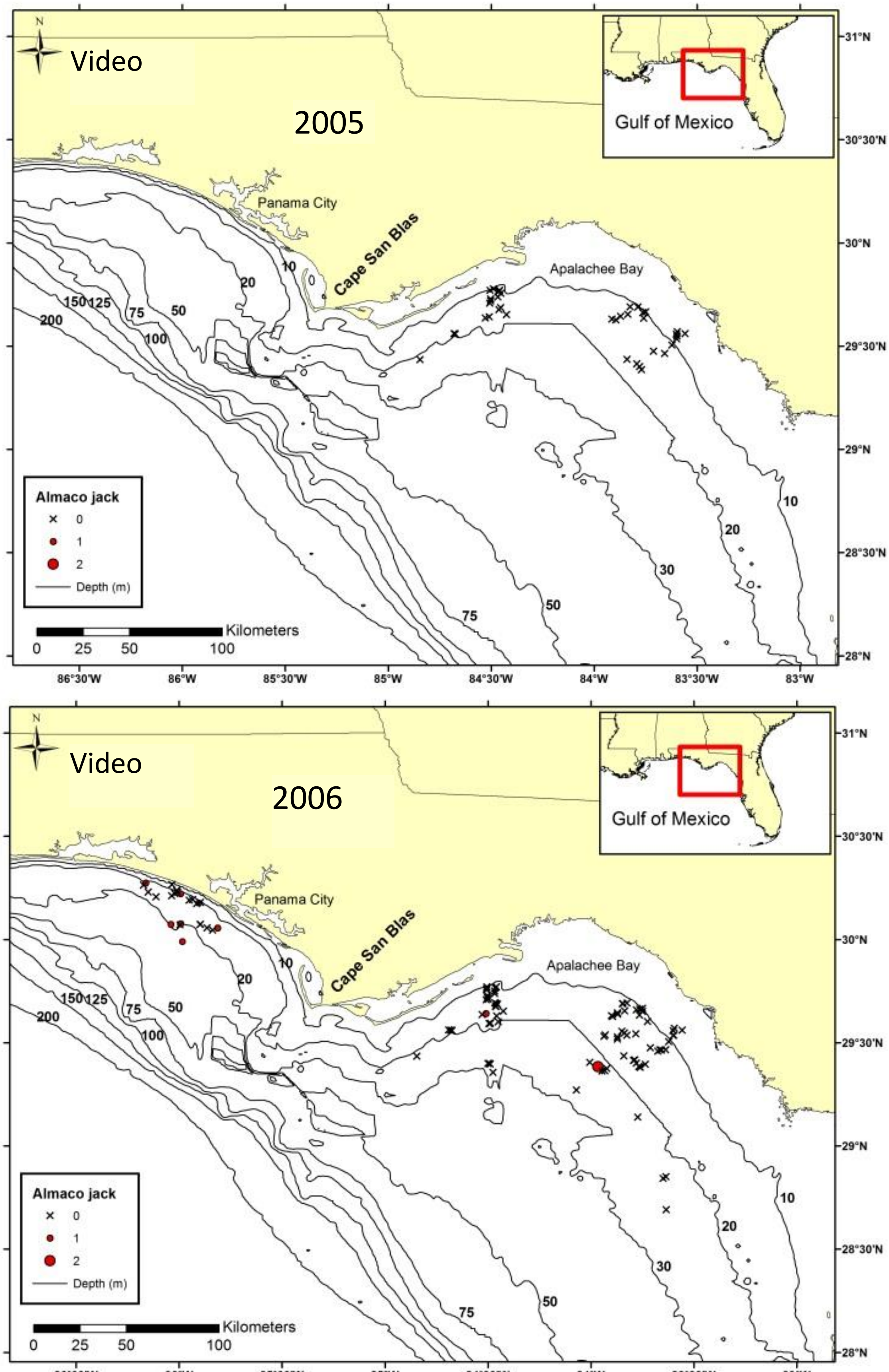
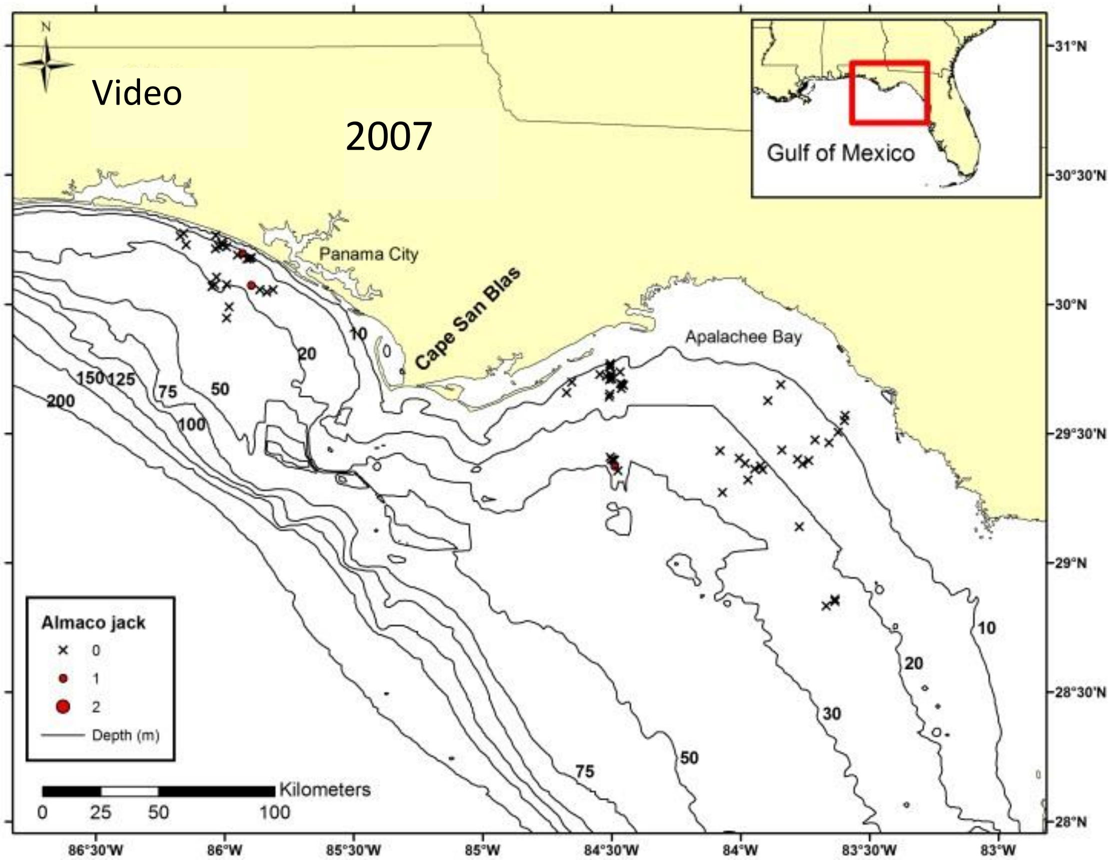


Figure 14. Annual distribution and relative abundance of almaco jack observed with stationary, high definition video or mpeg cameras (min counts) in the Panama City NMFS reef fish survey, 2005-2006. Sites sampled where no almaco jack were caught or observed are indicated with an X.



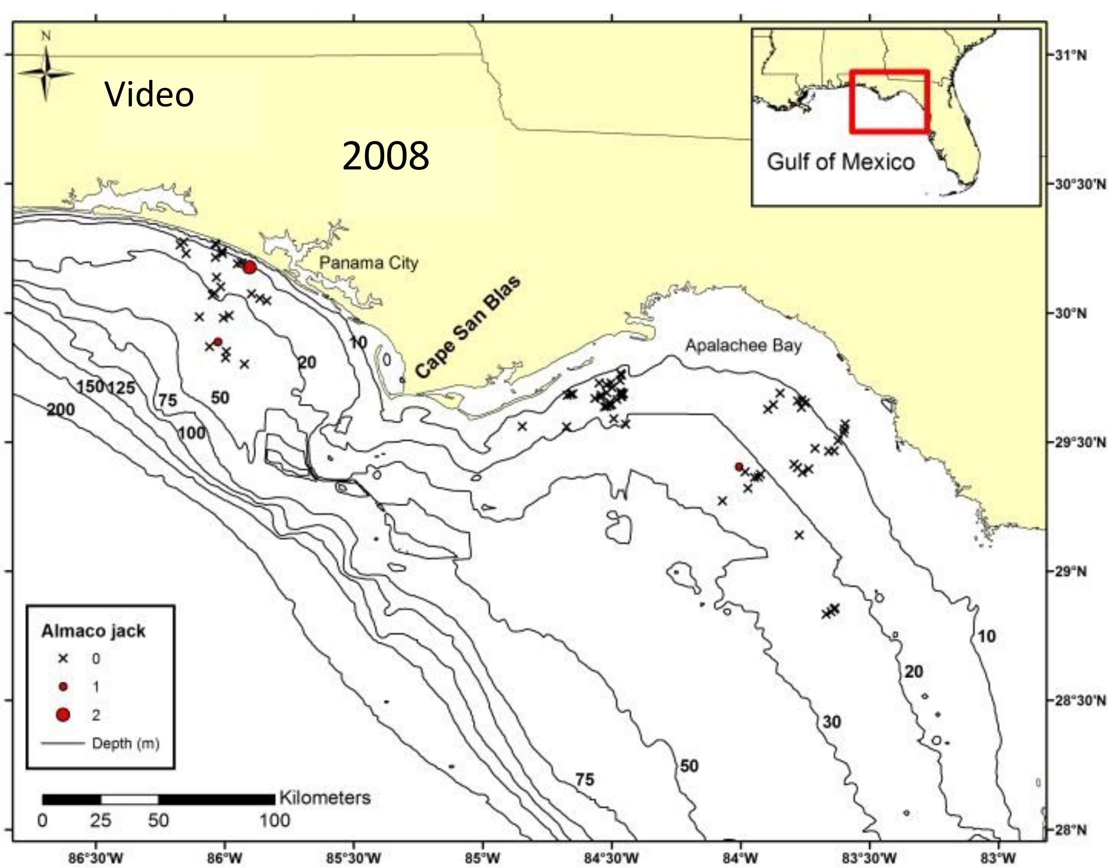


Figure 14 cont. Annual distribution and relative abundance of almaco jack observed with stationary, high definition video or mpeg cameras (min counts) in the Panama City NMFS reef fish survey, 2007 - 2008. Sites sampled where no almaco jack were caught or observed are indicated with an X.

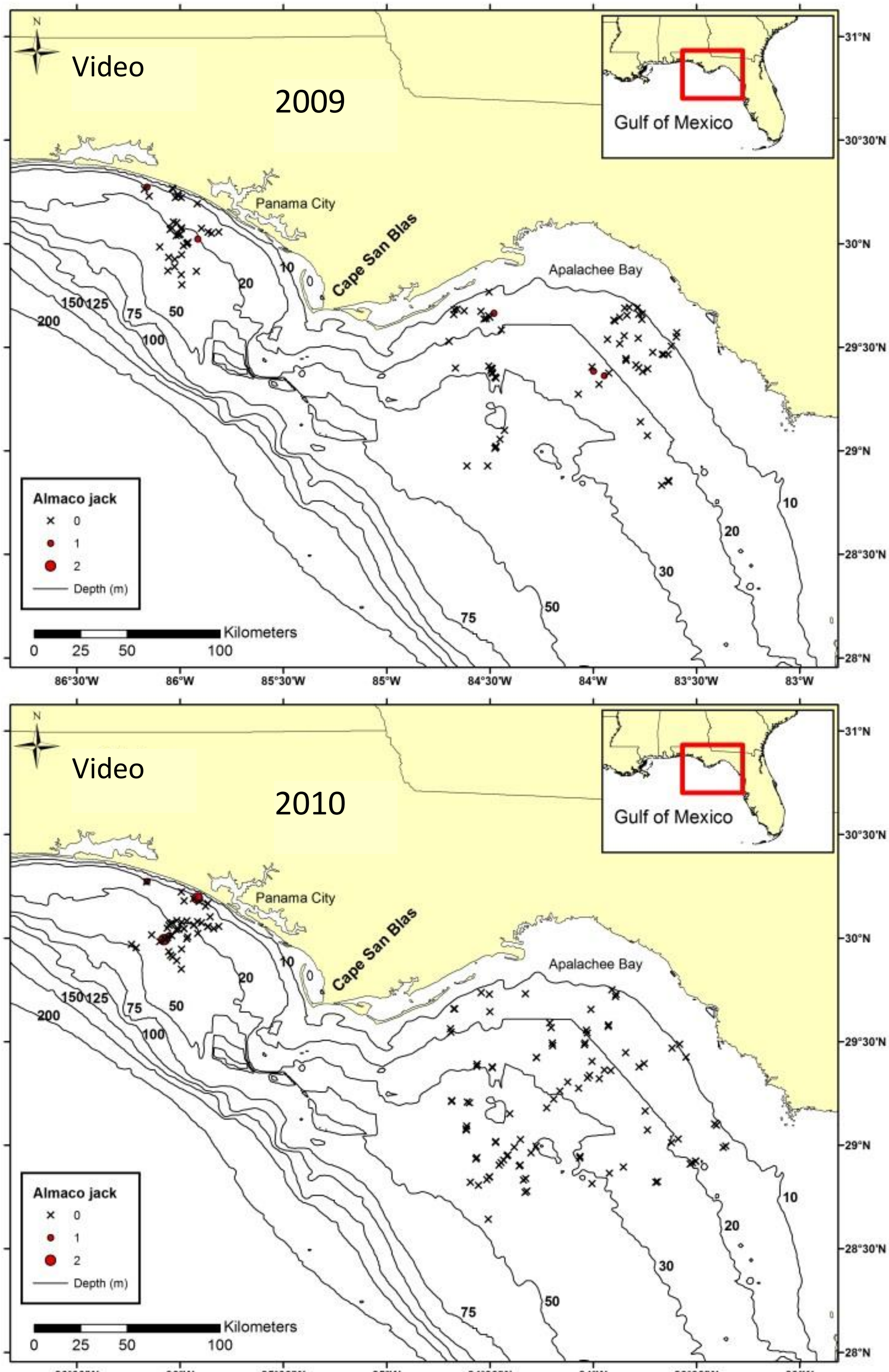
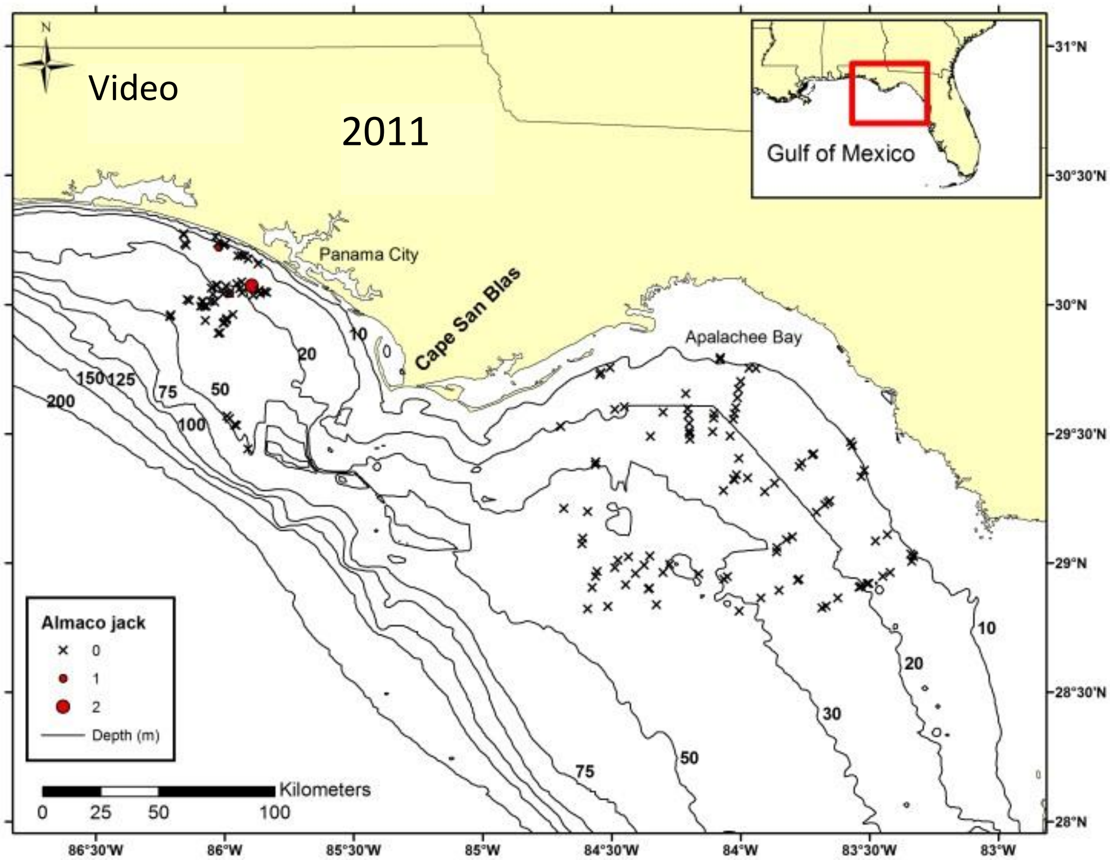


Figure 14 cont. Annual distribution and relative abundance of almaco jack observed with stationary, high definition video or mpeg cameras (min counts) in the Panama City NMFS reef fish survey, 2009 - 2010. Sites sampled where no almaco jack were caught or observed are indicated with an X.



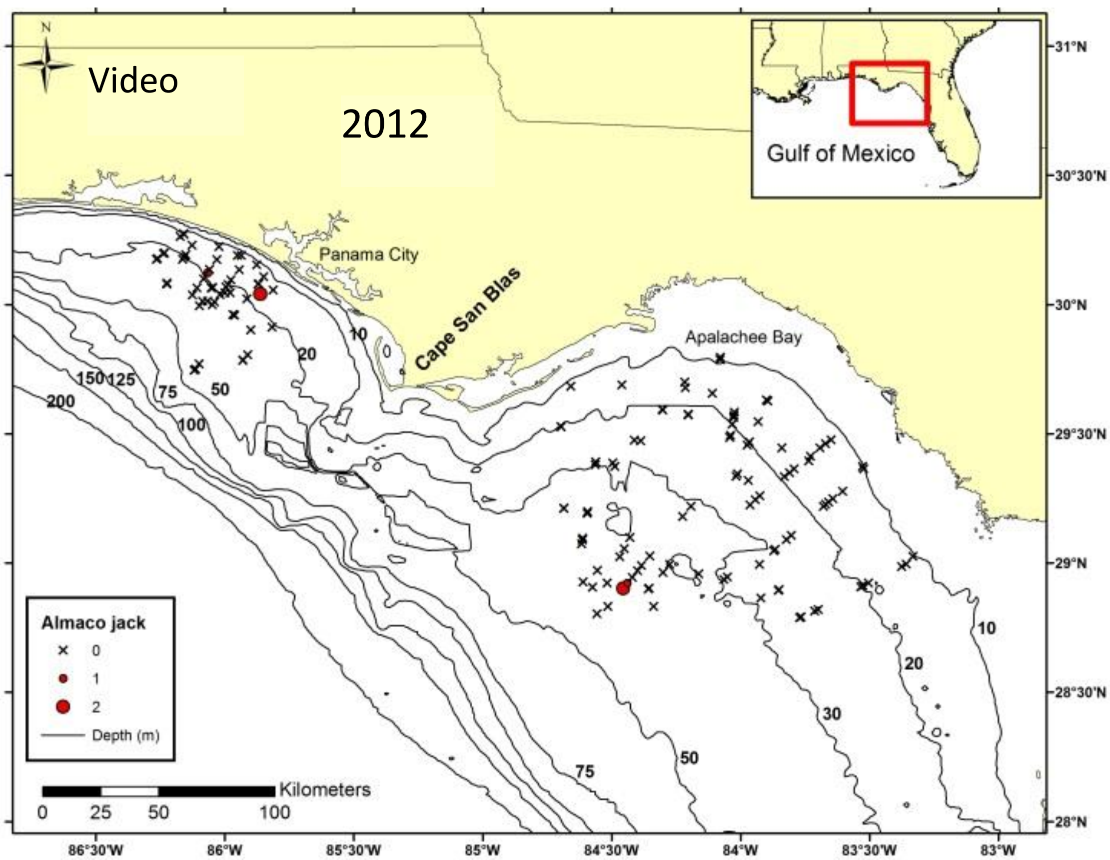
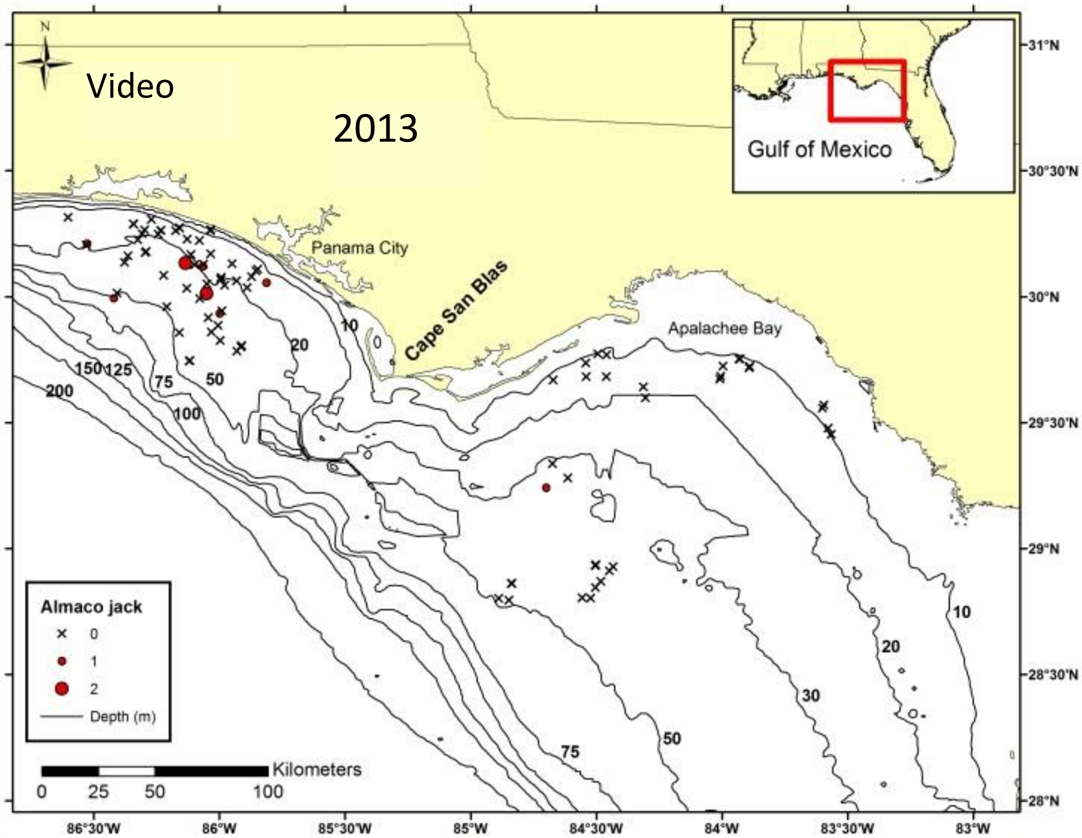


Figure 14 cont. Annual distribution and relative abundance of almaco jack observed with stationary, high definition video or mpeg cameras (min counts) in the Panama City NMFS reef fish survey, 2011-2012. Sites sampled where no almaco jack were caught or observed are indicated with an X.



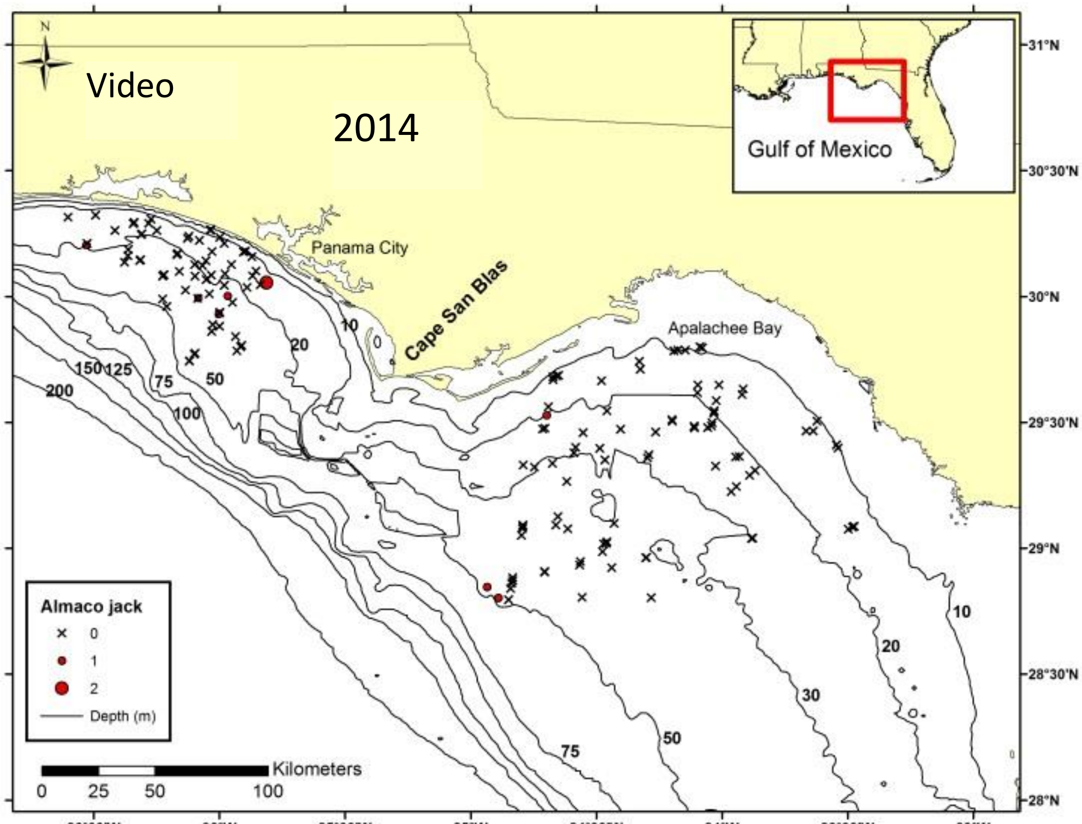


Figure 14 cont. Annual distribution and relative abundance of almaco jack observed with stationary, high definition video or mpeg cameras (min counts) in the Panama City NMFS reef fish survey, 2013-2014. Sites sampled where no almaco jack were caught or observed are indicated with an X.