Standardized video counts of Southeast U.S. Atlantic red snapper (*Lutajanus campechanus*) from the Southeast Reef Fish Survey

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Abstract

Standardized video counts of red snapper were generated from video cameras deployed by the Southeast Reef Fish Survey from 2010 – 2014. Samples between Cape Hatteras, North Carolina, and St. Lucie Inlet, Florida, were included in the analyses. The index is meant to describe population trends for red snapper in the region. To obtain an index of video counts, a zero-inflated negative binomial model was used to standardize video count data by a variety of predictor variables, differences across years in sampling effort (with respect to the predictor variables investigated) were accounted for, and a camera calibration calculation was used to calibrate counts of red snapper between two different cameras that were used during monitoring.

Background

The Marine Resources Monitoring, Assessment, and Prediction (MARMAP) program has conducted most of the historical fishery-independent sampling in the U.S. South Atlantic (North Carolina to Florida). MARMAP has used a variety of gears over time, but chevron traps are one of the primary gears used to monitor reef fish species and have been deployed since the late 1980s. In 2009, MARMAP began receiving additional funding to monitor reef fish from the SEAMAP-SA program. In 2010, the SouthEast Fishery-Independent Survey (SEFIS) was initiated by NMFS to work collaboratively with MARMAP/SEAMAP-SA using identical methods to collect additional fishery-independent samples in the region. Together, these three programs are now called the Southeast Reef Fish Survey (SERFS).

The SERFS survey currently samples between Cape Hatteras, North Carolina, and St. Lucie Inlet, Florida. This survey targets hardbottom habitats between approximately 15 and 100 meters deep. SERFS began affixing high-definition video cameras to chevron traps on a limited basis in 2010 (Georgia and Florida only), but since 2011 has attached cameras to all chevron traps as part of their normal monitoring efforts. All five years of data are included here, as recommended by Bacheler and Carmichael (2014; SEDAR41-RD23).

Hard-bottom sampling stations were selected for sampling in one of three ways. First, most sites were randomly selected from the SERFS sampling frame that consisted of approximately 3,000 sampling stations on or very near hard bottom habitat. Second, some stations in the sampling frame were sampled opportunistically even though they were not randomly selected for sampling in a given year. Third, new hard-bottom stations were added during the study period through the use of information from various sources including fishermen, charts, and historical surveys. These new locations were investigated using a vessel echosounder or drop cameras and sampled if hard bottom was detected. Only those new stations landing on hardbottom habitat were included in the analyses. All sampling for this study occurred during daylight hours between April and October on the R/V *Savannah*, R/V *Palmetto*, NOAA Ship *Nancy Foster*, or the NOAA Ship *Pisces* using identical methodologies as described below.

Samples were intentionally spread out spatially on each cruise (see Figure 2 in Bacheler and Carmichael 2014).

Chevron fish traps with attached video cameras were deployed at each station sampled in our study (Figure 1). Chevron traps were constructed from plastic-coated, galvanized 2-mm diameter wire (mesh size = 3.4 cm2) and measured 1.7 m × 1.5 m × 0.6 m, with a total volume of 0.91 m^3. Trap mouth openings were shaped like a teardrop and measured approximately 18 cm wide and 45 cm high. Each trap was baited with 24 menhaden (*Brevoortia* spp.). Traps were typically deployed in groups of six, and each trap in a set was deployed at least 200 m from all other traps to provide some measure of independence between traps. A soak time of 90 minutes was targeted for each trap deployed.

GoPro Hero (2010) or Canon Vixia HFS-200 high-definition video cameras in Gates underwater housings (2011 – 2014) were attached to chevron traps. A second high-definition GoPro Hero video or Nikon Coolpix S210/S220 still camera was attached over the nose of most traps in an underwater housing, and was used to quantify microhabitat features in the opposite direction. Cameras were turned on and set to record before traps were deployed, and were turned off after trap retrieval. Trap-video samples were excluded from our analysis if videos were unreadable for any reason (e.g., too dark, camera out of focus, files corrupt) or the traps did not fish properly (e.g., bouncing or dragging due to waves or current, trap mouth was obstructed).

For each fish trap deployed with a camera, video reading time was limited to an interval of 20 total minutes, commencing 10 minutes after the trap landed on the bottom to allow time for the trap to settle. One-second snapshots were read every 30 seconds for the 20-minute time interval, totaling 41 snapshots read for each video sample. SERFS employs video readers to count fish on videos. There was an extensive training period for each video reader, and all videos from new readers are re-read by fish video reading experts until they are very high quality. After that point, 10% or 15 videos (whichever is larger) are re-read annually by fish video reading experts. Video readers also quantify microhabitat features (percent of bottom that is hardbottom, maximum substrate relief, substrate size, coverage of attached biota, predominant biotic type, and maximum biotic height), in order to standardize for habitat types sampled over time. Water clarity was also scored for each sample as poor, fair, or good. If bottom substrate could not be seen, then water clarity was considered poor, and if bottom habitat could be seen but the horizon was not visible, water clarity was considered fair. If the horizon could be seen in the distance, water clarity was considered to be good. Including water clarity in index models allowed for a standardization of fish counts based on variable water clarities over time and across the study area. A CTD cast was also taken for each simultaneously deployed group of traps, within 2 m of the bottom, and water temperature from these CTD casts was available for standardization models.

Camera calibration

GoPro cameras were used for fish counts in 2010, while Canon cameras were used in 2011 – 2014. To calibrate fish counts between these two cameras, side-by-side Canon-GoPro videos were taken during the summer of 2013 and read for red snapper. Additionally, a lab experiment was conducted to quantify differences in field of view between the two cameras. Results indicated the Canon cameras saw 51% of the field of video of GoPro cameras, but the quality of GoPro videos was perhaps slightly lower than that of Canon videos. A total of 15 calibration videos were read that included red snapper. Based on a regression analysis applied to the calibration video results, there were 53% (1 minus the regression slope parameter) fewer red snapper seen on Canon cameras compared to GoPro cameras, which is almost exactly what one would predict based on the reduction of field of view on Canon cameras compared to GoPro cameras (see Figures 7-9 in Bacheler and Carmichael 2014). Therefore, it was recommended that the 2010 relative abundance data point be reduced by 53% to account for differences in viewing areas among the cameras.

Data and Treatment

Data subsetting

Overall, there were 4923 survey videos with *L. campechanus* data during the 5 year sampling period (2010-2014). We removed data points in which the survey video was considered unreadable by an analyst, or if the survey point was located at a depth greater than 100 meters, due to very limited samples in waters deeper than 100 m. Additionally, survey video for which less than 41 video frames were read was removed from the full data set. Standardizing the number of readable frames was essential due to our use of *SumCount* as a response variable (see below). We also identified any video sample in which corresponding predictor variables were missing and removed them from the final data set.

Of the total 4923 video samples considered for inclusion in our modeling analysis, 514 were removed based on the data subsetting procedure described above, leaving 4409 samples in the *L. campechanus* analyses for 2010 – 2014 (Figure 2).

Standardization

Response Variable

For the video index of *L. campechanus,* we modeled the *SumCount*, or total number of red snapper observed across all readable video frames for each sample. There are a number of viable candidate response variables applicable for the estimation of abundance from video surveys, the relative merits of which were discussed at length during the video index development workshop (Bacheler and Carmichael 2014). The panel recommended the use of *SumCount* as a response variable suitable for a zero-inflated modeling approach (we employed a zero-inflated model in our analysis). The use of *SumCount* requires that an equal number of video frames be considered for each data point considered in the model estimation. As a result, only samples with 41 readable frames (the maximum number) were included in our analysis (~99% of all samples).

Explanatory Variables

We considered 9 explanatory variables in our model analysis, which included year, season, depth, latitude, water temperature, turbidity, and current direction, all of which were recommended during the video index development workshop (Bacheler and Carmichael 2014). The workshop panel also suggested including habitat variables, for which we included biotic density and substrate composition.

YEAR (*y*) – Year was included because standardized catch rates by year are the objective of this analysis. We modeled data from 2010-2014, data from 2010 was spatially limited due to reduced video deployment during this initial year. Due to the high spatial overlap between the sampled region and the spatial occupancy of *L. campechanus,* data from 2010 were included in this analysis. This decision was supported by recommendations from the video index development panel (Bacheler and Carmichael 2014). Annual summaries of data points considered are outlined in Table 2.

SEASON (*t*) – a temporal parameter based on the Julian day the sample was collected (Figure 3). The season parameter was treated as an octile factor based on the recommendations of the video index development workshop.

DEPTH (*d*) – Water depth is a key component effecting the distribution of *L. campechanus*, we considered all data points in waters shallower than 100m. Data points were excluded from deeper waters generally due to limited samples and rare occurrence (Figure 3). Annual depth distribution for survey data are outlined in Table 2. The depth parameter was treated as a quantile factor based on the recommendations of the video index development workshop.

LATITUDE (*lat*) – The latitude of video samples were included as a spatial parameter in the model (Figure 3). Based on recommendations made by the video index development workshop, latitude was treated as a factor in the model and divided into 8 levels based on octiles.

TEMPERATURE (*temp*) – Bottom water temperature was collected from each group of traps and incorporated as a predictor variable. Bottom water temperature ranged from 12 – 29 degrees Celsius (Figure 3). For the standardization model temperature was treated as a factor with 4 levels based on quantiles.

TURBIDITY (*wc*) – Due to the effect of turbidity on both species distributions and on the ability of an analyst to process video survey samples, we included water clarity (*wc*) in our standardization model. Turbidity information was recorded during video analysis based on the ability of an analyst to perceive the horizon and surrounding habitat and was scored at 3 levels (0 – Horizon visible, 1 – Horizon not visible but habitat is still visible, 2 – Both horizon and habitat are not visible).

CURRENT DIRECTION (*cd*) – A categorical variable estimating current direction based on the video point of view. Current direction data was included to better account for variability in detection due to the current moving fish away or towards the camera. This variable is collected during video processing and scored natively as a 4 level categorical variable (Towards, Away, Left to Right, and Right to Left). It was incorporated into the model as “Towards”, “Away”, and “Sideways”.

BIOTIC DENSITY (*bd*) – An estimation of the percent cover of attached biota visible during any video. The estimation is made based on percentage cover and ranged from 0 – 98%. For our analysis, *bd* was treated as a categorical variable with 4 levels: none (0%), low (1-9%), moderate (10-39%), and high (>40%).

SUBSTRATE COMPOSITION (*sc*) – An estimate of the total percent of substrate that is consolidated sediments. Consolidated sediment is defined as rocks or boulders the size of a fist or larger, or hard pavement habitats. For our analysis, substrate composition was treated as a categorical variable with 4 levels: none (0%), low (1-9%), moderate (10-39%), and high (>40%).

Zero-Inflated Model

The recommendation of the video index workshop was to apply a zero-inflated modeling approach to develop a fishery-independent video index for *L. campechanus* in the South Atlantic. Zero-inflated models are valuable tools for modeling distributions that do not fit a standard error distribution due to an excessive number of zeroes. These data distributions are often referred to as “zero-inflated” and are a common condition of count based ecological data. Zero inflation is considered a special case of over dispersion that is not readily addressed using traditional transformation procedures (Hall 2000). Due to the high proportion of zero counts found in our data set (Figure 4), we used a zero inflated mixed model approach that models the occurrence of zero values using two different processes, a binomial process and a count process (Zuur et al. 2009). The benefit and utility of this approach was discussed at length during the video index workshop (Bacheler and Carmichael 2014) and was the final recommendation of the panel.

Initially, both a zero-inflated Poisson (ZIP) and a zero-inflated negative binomial (ZINB) formulation were considered and each model included all nine of the predictor variables.

(1) $$SumCount = y + wc + cd + sc + bd + d + t + lat + temp \mid y + wc + cd + sc + bd + d + t + lat + temp$$

We compared the variance structure of each model formulation using a likelihood ratio test (Zuur et al 2009), to determine the most appropriate model formulation for the development of a video index for red snapper. A likelihood ratio test (Table 1) showed strong support for application of a ZINB formulation, as did a comparison of model fit for both the ZIP and ZINB formulations (Figure 5), which resulted in the decision to use a ZINB approach. The results concurred with expectations based on the level of zero-inflation and over dispersion within the original red snapper data and with the recommendations of the video index development panel (Bacheler and Carmichael 2014).

A backwards step-wise model selection procedure was used to exclude unnecessary model parameters from the full model (1) formulation. The optimum red snapper model formulation (2) was determined using a combination of AIC and likelihood ratio tests (Zuur et al. 2009). Water clarity (*wc*) was excluded from the negative binomial component of the model and both water clarity (*wc*) and season (*t*) were excluded from the binomial component of the model (Table 3).

(2) $$SumCount = y + cd + sc + bd + d + t + lat + temp \mid y + cd + sc + bd + d + lat + temp$$

Model diagnostics showed no discernable pattern of association between Pearson’s residuals and fitted values or the fitted values and the original data (Figure 6). Additionally, an examination of model residuals for the spatio-temporal (Figure 7) and environmental model parameters (Figure 8) showed no clear patterns of association, indicating correspondence to underlying model assumptions (Zuur et al. 2009). Finally, a comparison of predicted values against the original data distribution (Figure 9) shows how our model fits the original data.

All data manipulation and analysis was conducted using R version 3.1.2 (R Core Team 2014). Modeling was executed using the *zeroinfl* function in the *pscl* package (Jackman 2008), available from the Comprehensive R Archive Network (CRAN).

Results

The relative nominal CPUE for *L. campechanus* was 2.609 in 2010, 0.433 in 2011, 0.567 in 2012, 0.639 in 2013, and 0.752 in 2014 (Table 4). After standardizing the original data set by each of the predictor variables included in the final model, we obtained a CPUE estimate of 2.913 in 2010, 0.379 in 2011, 0.503 in 2012, 0.537 in 2013, and 0.880 in 2014. When also accounting for unequal sampling across years (with respect to the predictor variables included in the final model), we obtained a CPUE estimate of 2.016 in 2010, 0.467 in 2011, 0.831 in 2012, 0.626 in 2013, and 1.060 in 2014. When we applied the camera calibration calculation for 2010 to these standardized annual values, we obtained a CPUE estimate of 1.206 in 2010, 0.592 in 2011, 1.056 in 2012, 0.798 in 2013, and 1.348 in 2014 (Table 4).

Only the 2011 relative nominal value falls within the 2.5% and 97.5% confidence intervals of the standardized index (Figure 10). The nominal value for 2010 was considerably higher than the standardized index value for 2010 while the nominal values for 2011, 2012, 2013, and 2014 were all considerably lower than their standardized index values, which was expected due to the integration of the camera calibration calculation into the standardized index. The standardized video index indicates that while there is a considerable amount of fluctuation in relative annual abundance from year to year, red snapper relative abundance has been generally stable across the survey years (Figure 10). However, due to the short temporal extent of this index (5 years), limited inferences can be made concerning long term patterns of *L. campechanus* relative abundance.

Literature cited

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Table 1: Preliminary model formulation comparison

	df	Likelihood	df	χ^2	*p*-value
ZIP	70	-14517			
ZINB	71	-5257	1	18520	<0.001

Table 2: Annual total number of video samples included in the analysis

Year	Number of video samples	Depth range (m)	Latitude range	Date range
2010	166	23-64	28.71-31.74	209-300
2011	575	15-93	27.23-34.54	139-298
2012	1075	15-98	27.23-35.02	115-284
2013	1219	15-92	27.33-35.02	114-277
2014	1374	15-99	27.23-35.02	113-294

Table 3: Model selection results for Zero-Inflated Negative Binomial model for red snapper observed during SERFS video surveys, 2010-2014

	Removed Term						
Step	**Binomial Process**	**Count Process**	***df***	**AIC**	**χ2**	***df***	***p-value***
null	<none>	<none>	71	10655.88			
1	*wc*	<none>	69	10652.49	0.62	2	0.735
2	*wc*	*t*	62	10648.46	9.97	2	0.191
3	*wc*	*t*, *wc*	60	10645.86	1.40	2	0.497

Table 4: The relative nominal *SumCount*, number of stations sampled, proportion positive, standardized index, and CV for the SERFS red snapper video index

Year	Relative nominal *SumCount*	N	Proportion positive	Standardized index	CV
2010	2.61	166	0.355	1.21	0.22
2011	0.43	575	0.233	0.59	0.17
2012	0.57	1075	0.241	1.06	0.14
2013	0.64	1219	0.267	0.80	0.12
2014	0.75	1374	0.218	1.35	0.14

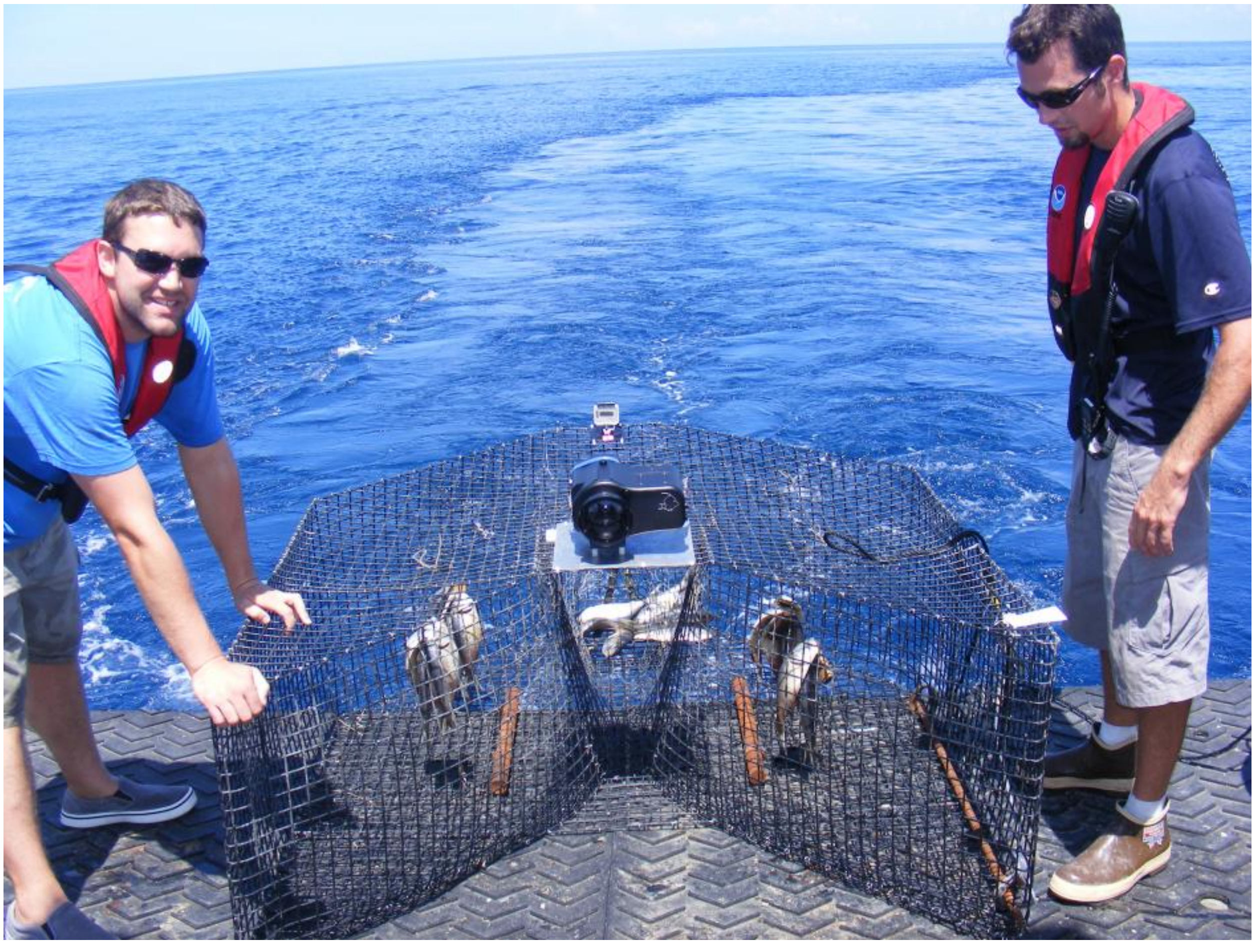

Figure 1: Chevron trap used by SERFS showing the attached underwater video cameras.

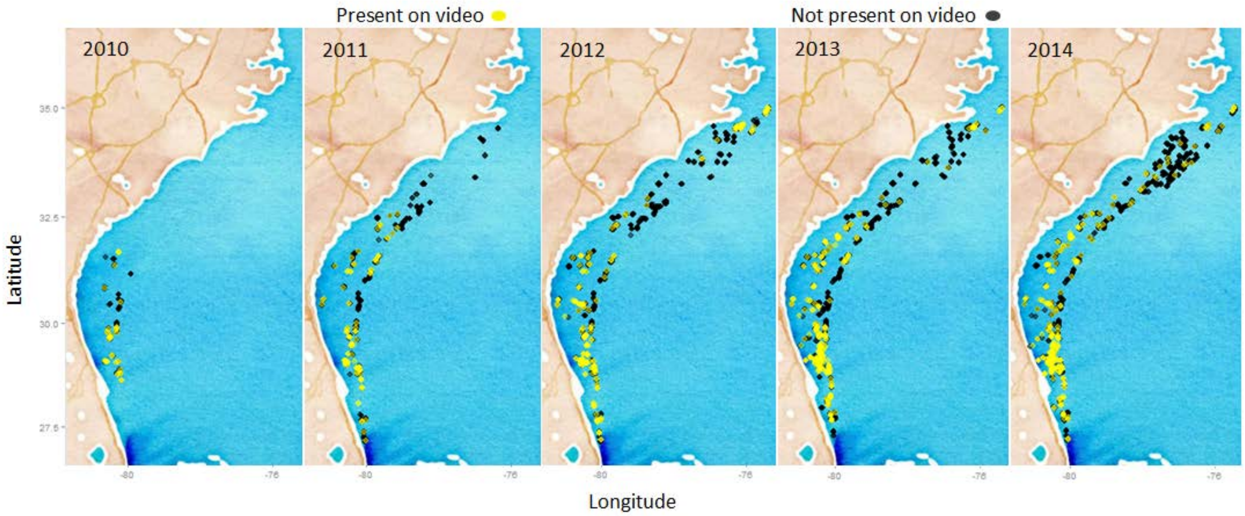

Figure 2: Annual spatial distribution of underwater video samples collected by SERFS in 2010 – 2014. Dark gray points indicate no red snapper were seen on video and yellow points indicate red snapper were seen on video. Note that yellow points were overlaid on top of gray points, and points may overlap. As a result, points were made slightly transparent.

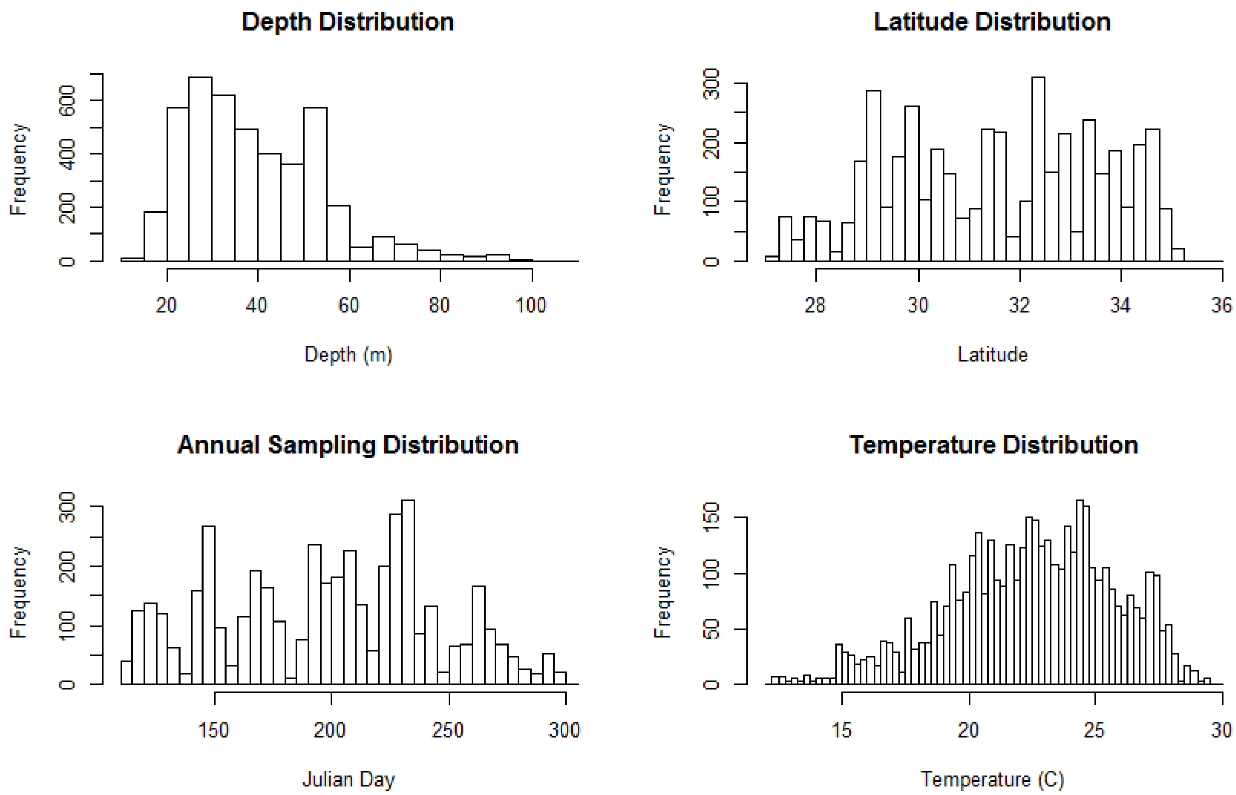

Figure 3: Sample distribution for original data continuous variables

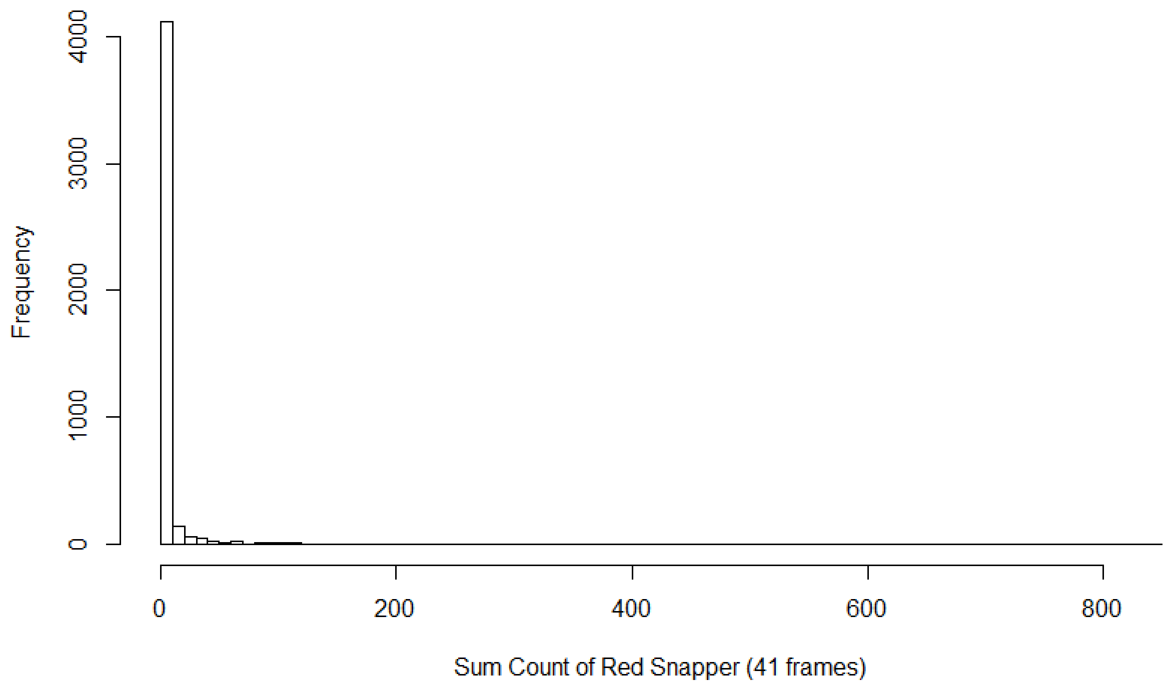

Figure 4: *SumCount* distribution for red snapper video observations in the South Atlantic.

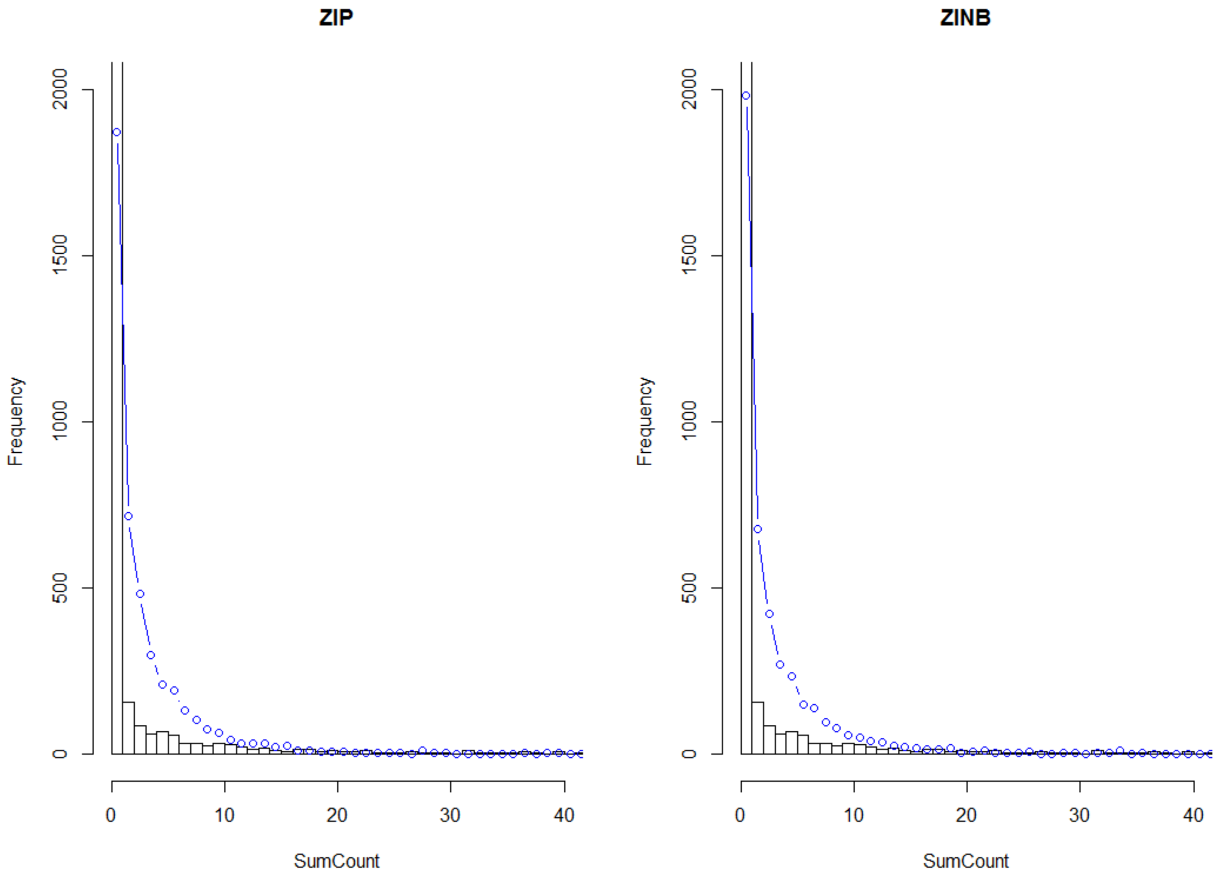

Figure 5: Model formulation comparison, with ZIP (left) and ZINB (right) fitted values plotted against the original data distribution

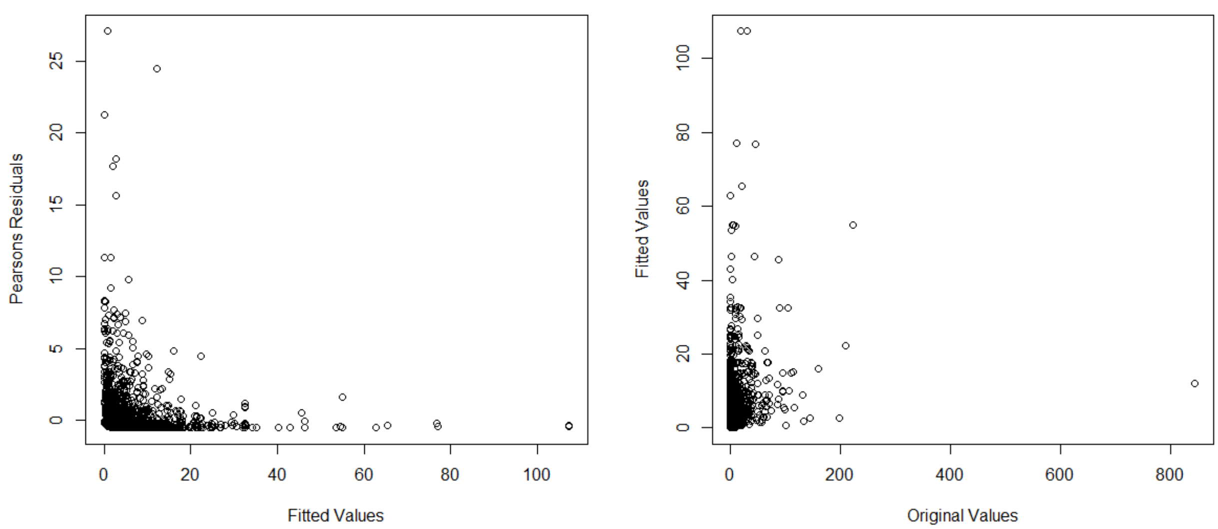

Figure 6: Model diagnostic plots showing fitted model values against Pearson's residuals (left) and fitted values plotted against original data values (right)

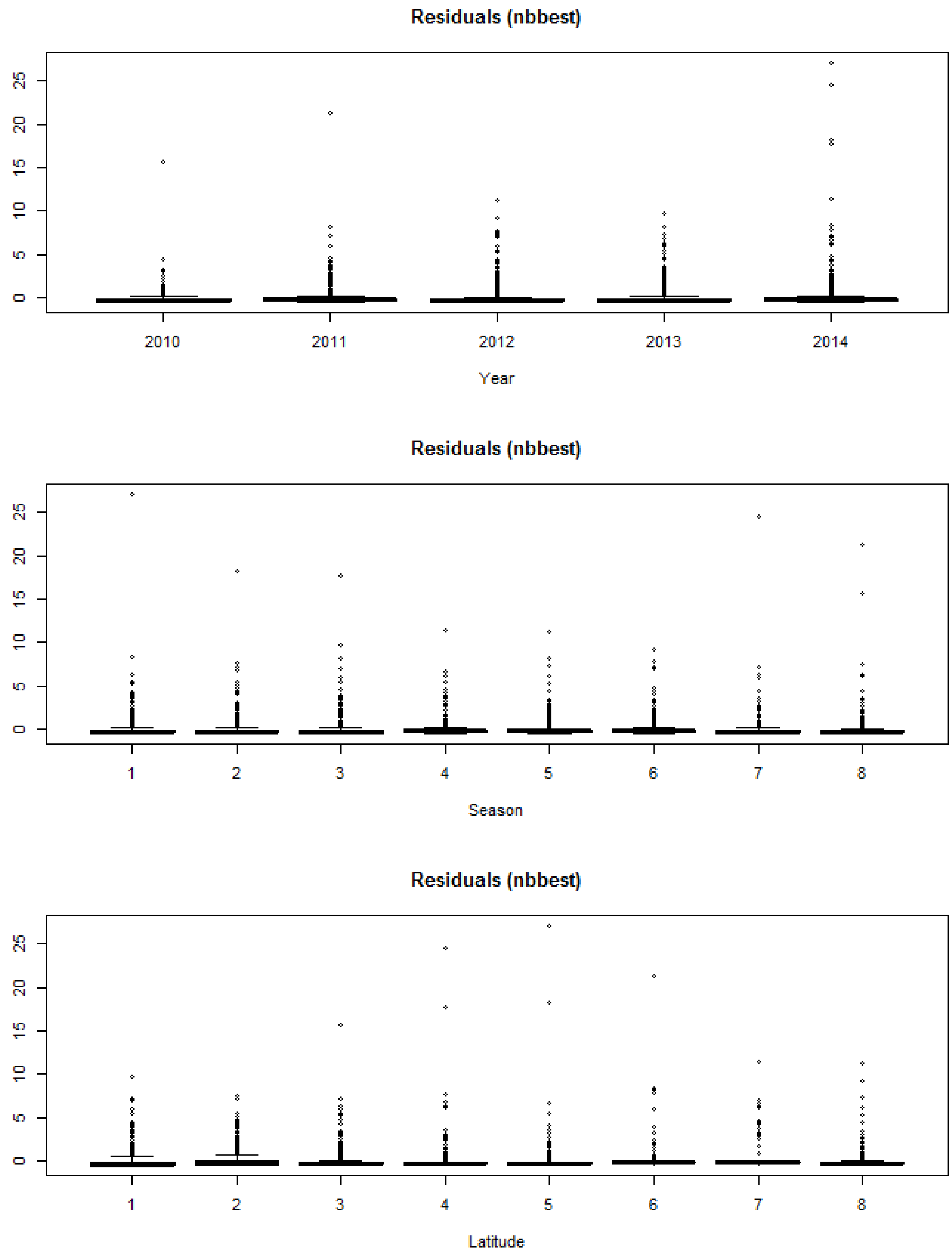

Figure 7: Model diagnostic plots showing Pearson's residuals from the final model plotted against both the temporal and spatial model variables

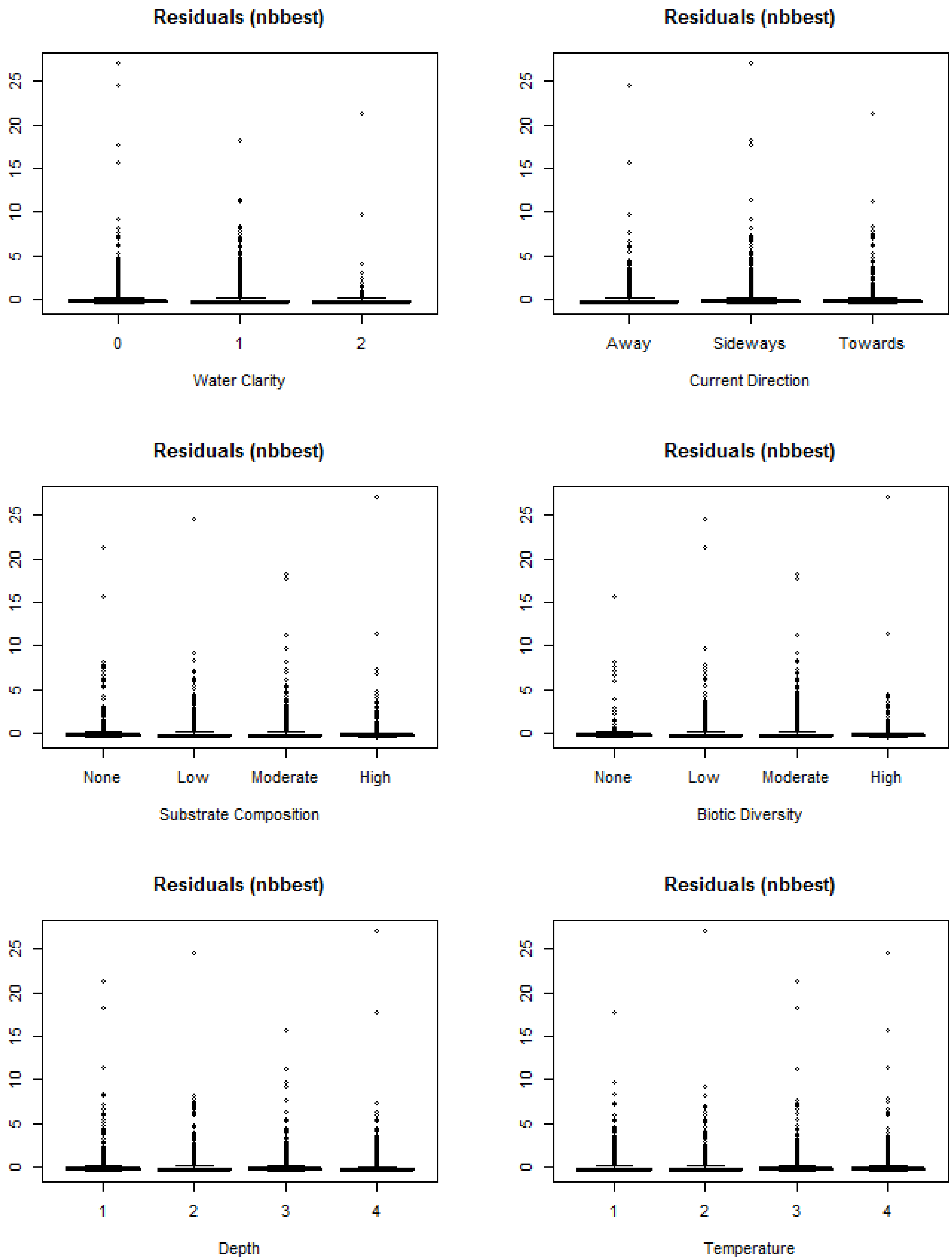

Figure 8: Model diagnostic plots showing Pearson's residuals for the final model plotted against environmental model parameters

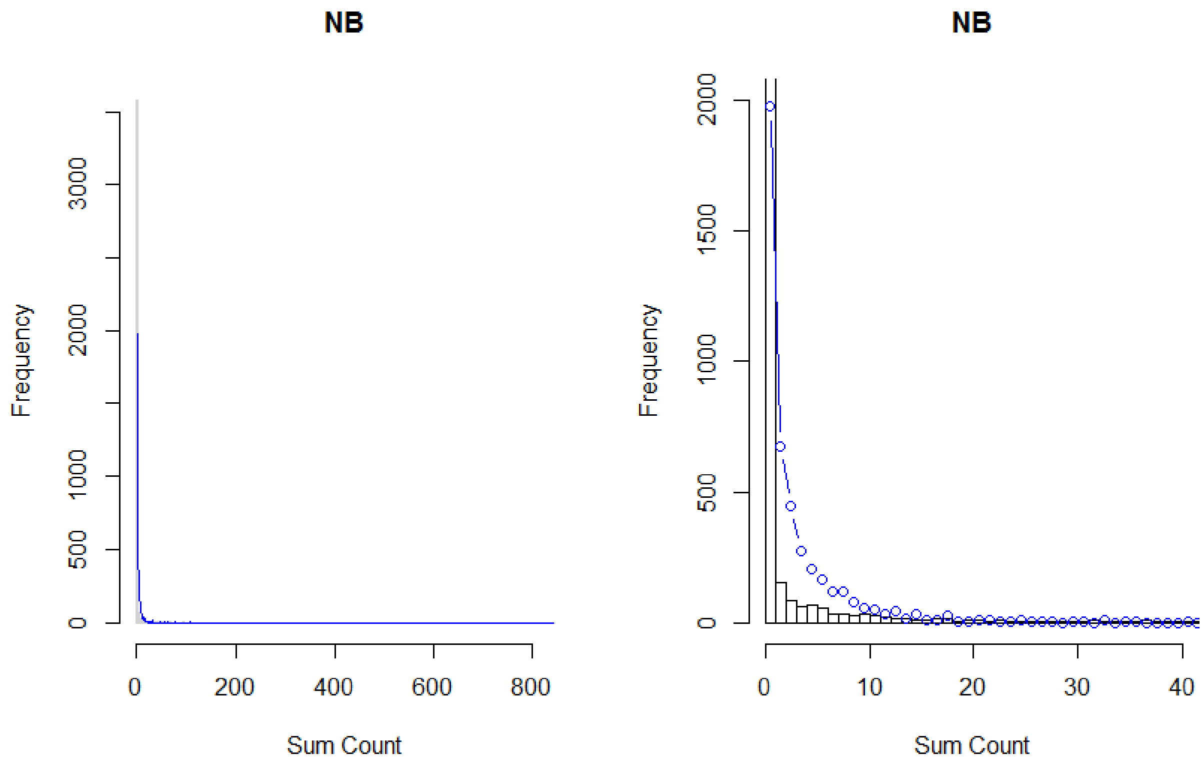

Figure 9: Model diagnostic plots of fitted model values (blue line) against the original data distribution. Full distribution view (left) and limited axis view (right)

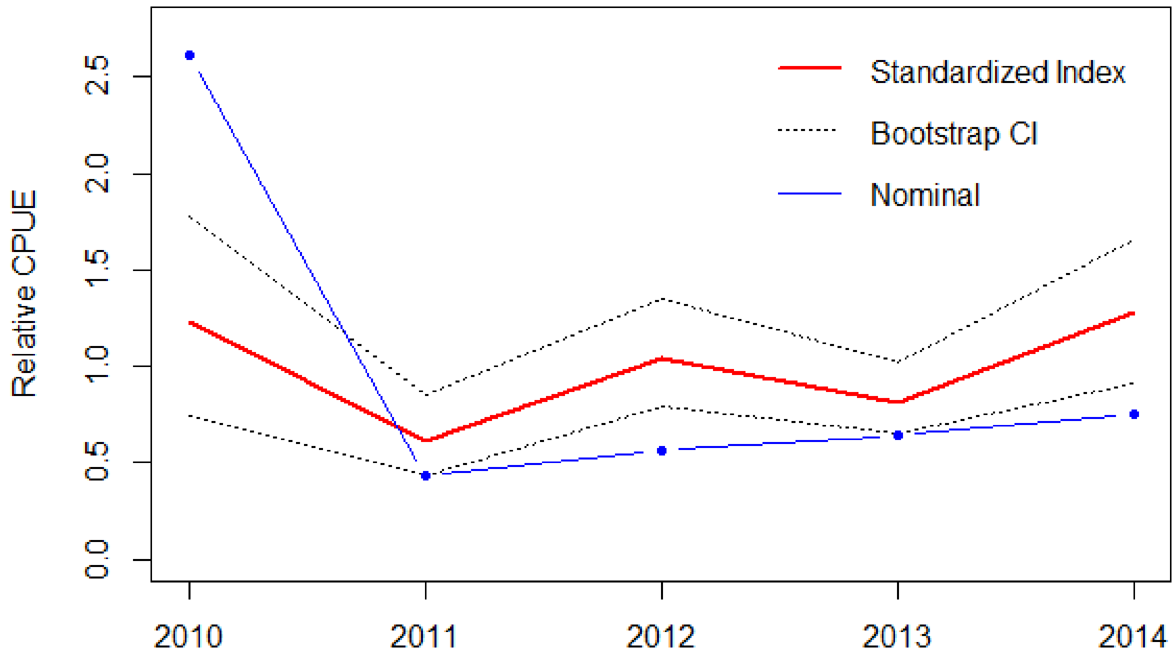

Figure 10: Relative standardized index (solid line) with 2.5% and 97.5% confidence intervals (dashed lines) and the relative nominal index (blue) for red snapper CPUE in the SERFS video survey