Galveston Bay Freshwater Inflow Re-Study

An investigation of productivity-inflow relationships

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Acknowledgements

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Executive Summary

This report addresses two tasks assigned to the Texas Water Development Board, which are related to the development of productivity-inflow regression equations for commercial fisheries harvest data and for Texas Parks and Wildlife Department fisheries-independent data. The first task was to extend the existing statistical analyses for the Galveston Bay Freshwater Inflow Study to include the most recent harvest and inflow data. It was also part of this task to relate freshwater inflows to Texas Parks and Wildlife's fisheries-independent dataset for Galveston Bay using existing methodologies. These analyses were conducted, compared, and evaluated. The second task was to explore new methodologies in relating freshwater inflows to the health of estuarine ecosystems along the Texas Coast. Several different approaches at multiple temporal and spatial scales were explored and evaluated. The results from these analyses can be used to determine which data sets and which approaches are helpful for making recommendations for freshwater inflows into Galveston Bay.

Background

In response to legislative directives beginning in 1975, the Texas Water Development Board (TWDB), the Texas Parks and Wildlife Department (TPWD), and the Texas Commission on Environmental Quality (TCEQ) jointly established and currently maintain a data collection and analytical study program focused on determining the effects of and needs for freshwater inflows into the state's bays and estuaries. To do this, Texas has developed a State Methodology whereby optimum freshwater inflow needs are determined through use of statistical regression models, which are based on bi-monthly freshwater inflows, bay salinities and commercial fisheries harvest (or biomass as total pounds harvested per year). Using this methodology, state agencies have developed inflow recommendations for each of the major estuaries and are now working on modifications to the methodology to provide recommendations for the five minor estuaries. These recommendations provide the basis for the Texas Commission on Environmental Quality's (TCEQ) consideration of water diversion permit applications within 200 river miles of the coast. Since the State Methodology was first developed and applied to the Trinity-San Jacinto Estuary, many additional years of hydrologic (flow), salinity, commercial fisheries harvest, and fisheries-independent data have been collected. Also, a review by Ward (1999) of the State Methodology as applied to Galveston Bay raised concerns regarding the scientific merits of the flow recommendations from the original study. In response, the agencies agreed to use an open, facilitated process to identify and clarify issues about the State Methodology as well as to obtain stakeholder input. The resulting stakeholder group met three times in 2004 and 2005 and concluded that a technical subcommittee needed to be appointed to address the technical concerns raised about the State Methodology.

In February 2005, the technical subcommittee met and identified 17 short- and long-term tasks to be completed by the TWDB, TPWD or other consulting agencies. Several high-priority tasks subsequently were selected to be addressed immediately by TPWD and TWDB. From these, two tasks assigned to the TWDB were supported with financial assistance from the U.S. Army Corps of Engineers' Texas Water Allocation Assessment (TWAA) funds. Results for these two tasks are reported herein.

Objectives of Study

Task 1: Extend and evaluate existing statistical analyses for the Galveston Bay Freshwater Inflow Study

a) Assist TWDB staff with the re-analysis of data relating freshwater inflows to commercial fisheries harvest in Galveston Bay using existing methodologies (i.e. multivariate regression techniques) on an extended data set (1959-2004). This task includes conducting regression analyses for eight commercially important species, along with indentifying outliers, testing for assumptions, and identifying limitations of the data and methodology. The methodology and related discussions are documented.

b) Assist with the analysis of data relating freshwater inflows to Texas Parks and Wildlife's fisheries-independent dataset for Galveston Bay using existing methodologies, such as multivariate regression. This task includes conducting regression analyses for key species (such as the previously identified eight commercially harvested species), along with identifying outliers, testing for assumptions, and identifying limitations of the data and methodologies. The methodology and discussions are documented.

Task 2: Explore new methods to relate freshwater inflows to the health of estuarine ecosystems along the Texas Coast

a) Assist in the development of new approaches, which will improve and extend the current methodology for relating bay health to freshwater inflows. Ecological data is often highly variable and complex and thus requires the use of modern, but complex statistical analyses which can account for stochastic influences. This includes indentifying other possible analyses and indentifying additional factors, other than inflow and species data, which may influence the relationship between inflow and bay health. The methodology and discussions are documented.

b) Assist in the investigation of suitable alternatives to bi-monthly inflows currently used in the regression methodology. Alternatives include investigating different temporal resolutions and exploring the importance of notable hydrological events, such as floods and droughts, on species abundance. The methodology and discussions are documented.

Task 1

(Text for this section was modified from a technical memo prepared for the Galveston Bay Technical Subcommittee, Guthrie 2008)

Task 1a: Assist TWDB staff with the re-analysis of data relating freshwater inflows to commercial fisheries harvest in Galveston Bay using existing methodologies (i.e. multivariate regression techniques) on an extended data set (1959-2004)

Task 1a includes a re-analysis of the original productivity-inflow regression equations along with an extension of the data set to include data from years 1988-2004. Substantially different hydrological regimes after 1988, which are discussed in a later section, resulted in the need to evaluate the fisheries harvest data as two data sets, *Early* (1959-1987) versus *Later* period (1988-2004, Task 5c).

Harvest Data

The original productivity-inflow regression equations were developed using commercial fisheries harvest data from 1959-1987. (Information about data sources are provided in Lee *et al.* 2001.) Whereas, the current analysis uses commercial fisheries harvest data from 1959-2004 (Fig. 1), thus adding 17 years of data. Annual harvest data for brown and white shrimp were obtained from the National Marine Fisheries Service (NMFS). Although reported values were provided as "heads off" weights, a conversion factor was used to calculate "heads on" weight for use in analysis (F. Patella, NMFS, *pers. comm.*). Annual harvest data for the remaining six species were obtained from Texas Parks and Wildlife Coastal Fisheries Division. In 1981, commercial harvest of red drum and spotted seatrout ended as these species were reclassified as game species. Without additional harvest data for red drum and spotted seatrout, it was not possible to include these species in the current, extended analysis, and so further development of harvest productivity-inflow equations included only six of the original species (Table 1).

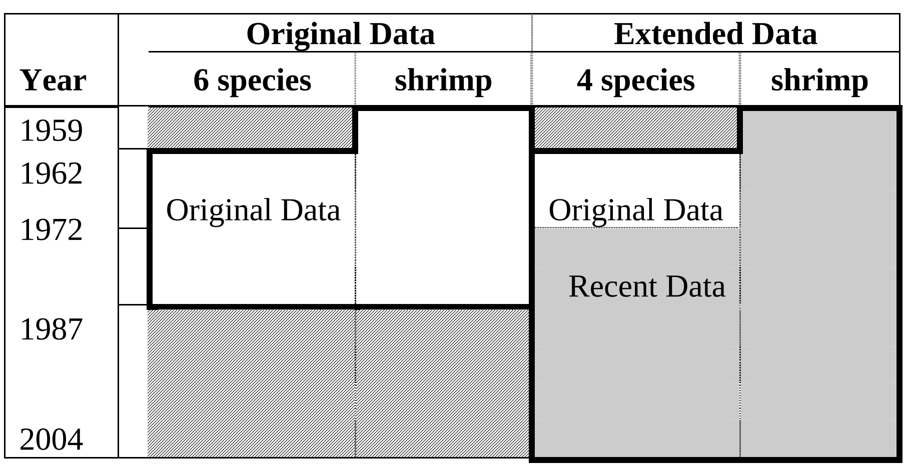

Figure 1. Schematic of original versus extended harvest datasets used to develop productivity-inflow relationships. The extended dataset uses the most recent data available from TPWD and NMFS. A portion of the original dataset was used for fisheries harvest data prior to 1972, because this data was not available electronically.

Table 1. Eight commercially important fisheries species included in the original harvest-inflow regression analyses for years 1959-1987 (Lee *et al.* 2001). After 1980, *S. ocellatus* and *C. nebulosus* were classified as game species and no longer commercially harvested. As indicated, the remaining six fisheries species were included in the extended analyses.

Species	**Common Name**	**Notes**	**Period of Record**	**Included in Extended Analysis**
Farfantepenaeus aztecus	Brown Shrimp	Heads on weight	1959 - 2004	✓
Litopenaeus setiferus	White Shrimp	Heads on weight	1959 - 2004	✓
Cynoscion nebulosus	Spotted Seatrout		1962 - 1980	∅
Sciaenops ocellatus	Red Drum		1962 - 1981	∅
Crassostrea virginica	Eastern Oyster	Meat only	1962 - 2004	✓
Callinectes sapidus	Blue Crab	Meat only	1962 - 2004	✓
Pogonias cromis	Black Drum		1962 - 2004	✓
Paralichthys lethostigma	Southern Flounder		1962 - 2004	✓

We obtained all harvest data from the same sources as the 2001 freshwater inflow study; however, electronic versions of harvest data for non-shrimp species from 1962-1971 were not available. Instead, we used the original harvest data for 1962-1971, obtained from published records, for the six non-shrimp species (Figure 1). This utilizes the most complete and recent data from TPWD and NMFS, although some differences exist between the original dataset and current dataset for years 1959-1987. Differences are attributed to corrections applied by TPWD and current reports should be considered the most accurate data (Page Campbell, *pers. comm*.). Specific differences include:

> <u>Brown Shrimp</u> annual harvest as provided by NMFS differs from previously recorded values for years: 1962, 1973, 1974, 1982, and 1987. When previously recorded values are used, mean harvest is 1,262.3 thousand pounds. The 2001 Freshwater Inflow Analysis report gives an annual mean harvest of 1,317.3 thousand pounds.
>
> <u>White Shrimp</u> annual harvest as provided by NMFS differs from previously recorded values for years: 1961, 1967, 1971-1975, 1982, 1985, and 1987. When the previously recorded values are used, mean harvest is 2,897.4 thousand pounds, the same as reported in the 2001 Freshwater Inflow Analysis.
>
> <u>Easter Oyster</u> annual harvests as calculated from txgal72-04.xls data set differed from previously recorded values for the years 1972-1982.

Black Drum maximum harvest occurred in 1984. TPWD Landings Report lists the maximum as 269.0 thousand pounds. However, the recent data file sent by TPWD (txgal72-04.xls) calculates maximum harvest in 1984 at 346.3 thousand pounds and mean harvest at 70.2 th.lbs.

Flounder annual harvest in 1984 as calculated from txgal72-04.xls yielded a value of 182.3 thousand pounds. This differed from the previously recorded harvest for this year (of 127.7 thousand pounds).

Statistical Analyses

The original analysis of productivity-inflow relationships used commercial fisheries harvest data from the period 1959-1987 regressed against bi-monthly freshwater inflows grouped according to either water-year or calendar-year and lagged from one to three years relative to the harvest year of the target species. The analysis was based on ordinary least squares regression methods using all possible subsets regression (Program 9R) in BMDP to identify the best model. To ensure consistency in comparison of results of the original and extended datasets, we completed an initial re-analysis of the original dataset. Model selection for both the re-analysis and extended dataset analysis was conducted using best subsets regression in the Minitab (Version 15) statistical package as well as all subsets regression in SAS (Version 9.1). Best subsets regression computes regression equations for all possible combinations of predictor terms (in this case all possible subsets of the six bi-monthly inflow terms) and shows the results for the best regression models for each possible number of predictor terms. We verified that the best subsets regression in Minitab and the all subsets regression in SAS produce the same results as the 9R program in BMDP. Model selection was based on selecting the most parsimonious model which maximized adjusted r^2 and minimized Mallow's statistic (C_p). Mallow's C_p indicates estimated model bias. Once a model was selected based on the criteria listed above, it was investigated for outliers and for violations of assumptions of linear regression.

The protocol for assessment of outliers in the original study was not clear, but seemed to result in an inconsistent declaration of outliers among species. In this re-analysis, we applied a conservative approach to selecting outliers which was based on a visual analysis of the residuals, values of the standardize residuals, and Cook's Distance, a measure of the influence of a single observation on the regression model. As a result, only two data points were considered to be outliers among the eight species datasets; they were: harvest year 1983 for eastern oyster (*Crassostrea virginica*) and harvest year 1984 for black drum (*Pogonias cromis)*. All other harvest years for all species were considered to fall within the expected natural range of variation and therefore included in the re-analysis. Additionally, the original analysis of harvest data for red drum (*Sciaenops ocellatus*) and spotted seatrout (*Cynoscion nebulosus*) was log transformed. Log transformations often are applied in an attempt to correct problems associated with skewed data, outliers, and/or unequal variation among the residuals (heteroscedasticity). There was no evidence to support transforming the harvest data for either of these species, and none were used in the re-analysis of the original harvest data (*Early Period*) or in analysis of the *Extended* data set.

Interpretation of statistical results must include consideration of ecological significance in addition to statistical significance. Moreover, expectations for inferring ecological relationships are necessarily different than those in other disciplines. In other words, models with r^2 values explaining less than 50% of the variation should not automatically be disregarded. Such equations still may be biologically meaningful in that they explain much of the observed variation, though other unaccounted factors may be equally or more important. A typical rule-of-thumb used in interpreting ecological analyses is to accept relationships which explain more than 30% of the variation as being biologically meaningful.

Another more rigorous method is to use a null expectation, where model $r^2 >$ null r^2. All results are then compared to a null model which assumes that no relationship exists between inflows and harvest. This null expectation is based on a regression equation computed from a set of data where the response variable is independent of all predictor terms. Such an equation has an expected r^2 value of $[(k-1)/(n-1)]$, where k is the number of *a priori* predictor terms, including the intercept, and n is the number of observations (Cook and Weisberg 1999). For the harvest-inflow regression equations – even if there were *no* relationship between annual harvest and bi-monthly inflows – we can expect r^2_{null} values between 0.13 to 0.38 due to chance alone (depending on the number of predictor terms (in our case, $k=7$) and the number of years (n) being evaluated in the data set). In using this approach, we are being realistic about our expectations for what the data can yield. Only statistically significant ($p<0.05$) and biologically meaningful ($r^2 > r^2_{null}$) models were considered as potentially meaningful regression equations.

Task 1a: Re-analysis of Original Harvest Data (Early Period, 1959-1987)

Action: We completed a re-analysis of harvest-inflow regression equations based on an original harvest dataset of eight commercially important species for years 1962-1987 (*Early Period*) with corresponding freshwater inflows and the adoption of a conservative approach for selecting outlier data points (Table 2).

Results: Re-analysis of the original dataset resulted in significant regression equations (for all but black drum) with similar bi-monthly inflow variables (Figure 2, Table 2) as originally reported in the 2001 study (Lee *et al.* 2001). In general, this conservative approach to outlier selection resulted in lower r^2 values (ranging from 0.09 to 0.58). However with the exception of black drum, the results still suggest that freshwater inflows are biologically important to these species. Most of the observed differences between the two analyses result from changes in the method of outlier selection. However, minor changes to the commercial harvest and hydrologic datasets contribute to some of the observed differences.

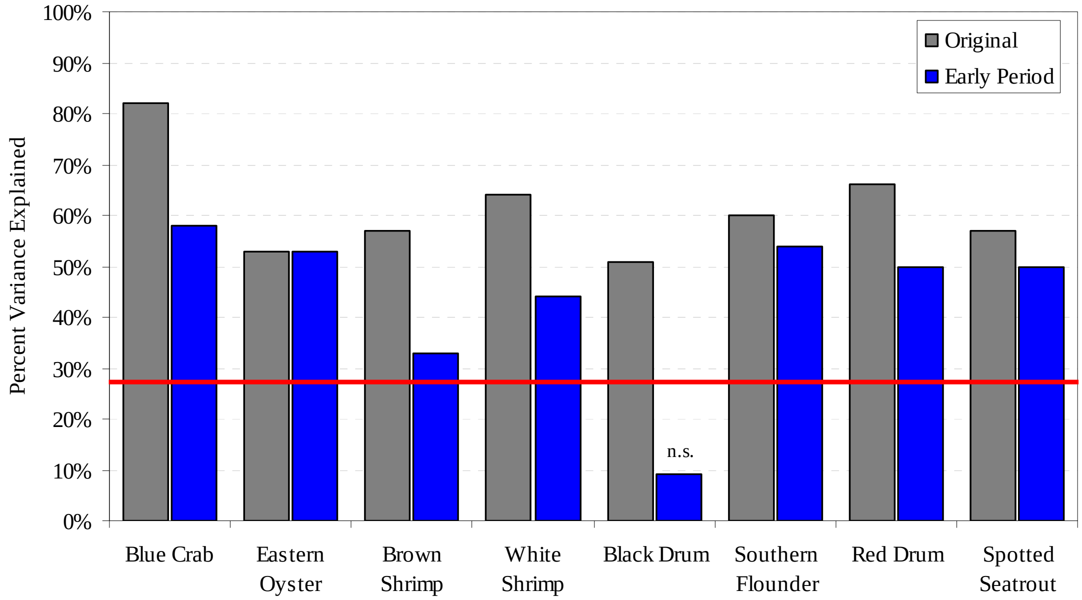

Figure 2. Percent of variance (r^2) explained by bi-monthly inflows on commercial harvest. All models are statistically significant, except the new black drum model, indicated by (n.s.). *Original* refers to analyses reported in the 2001 report; *Early Period* is the re-analysis of the original dataset (1962-1987) with a conservative selection of outliers. The bold red line represents a general *rule-of-thumb* threshold based on a calculated r^2_{null}, above which an analysis may be considered biologically relevant. Species-specific values are provided in Table 1.

Table 2. Regression equations for the harvest-inflow analysis of eight commercially important species. For each species, results of the *Original* analysis (as reported in Lee *et al.* 2001) are summarized along with three additional analyses. *Early Period* (1959-1987*) refers to a re-analysis of the original dataset. *Later Period* examines harvest and inflow data after 1987, 1988-2004*. *Extended* refers to an analysis of the full period of record, 1959-2004. Significant bi-monthly inflow predictor variables are highlighted to indicate negative (darker shading) or positive (lighter shading) responses to inflow. *Note: In general, the *Original* analysis included harvest years 1959-1987. For all species, except white and brown shrimp, harvest data begins in 1962. After 1981, red drum and spotted seatrout were reclassified as game species and are not included in the Later period or Extended analyes.

Harvest	r^2_{null}	r^2	$r^2_{adj.}$	n	y-intercept	Bi-Monthly Inflow Periods						Outlier Years	*p*
						JF	MA	MJ	JA	SO	ND		
Blue Crab													
Original	0.27	0.82	0.76	23	751	-0.276	0.846	-0.184		-0.474	0.600	1969 1974 1987	0.0001
Early Period	0.24	0.58	0.47	26	1070	-0.259	0.617	-0.172		-0.364	0.537	na	0.003
Later Period	0.38	0.54	0.39	17	3252	0.126			-0.680	0.551	-1.013	na	0.041
Extended	0.14	0.10	0.03	46	1782	-0.166	0.196			-0.142		na	0.26
Eastern Oyster													
Original	0.25	0.53	0.46	25	4170	-0.940		0.284	-0.945			1983	0.001
Early Period	0.25	0.53	0.46	25	4039	-0.944		0.299	-0.902			1983	0.001
Later Period	0.38	0.67	0.63	17	3403				-1.358	1.190		na	0.0004
Extended	0.14	0.21	0.15	43	1770	-0.558	0.376				0.854	na	0.027
Brown Shrimp													
Original	0.23	0.57	0.49	27	1019.8	-0.578			0.419	0.406	0.353	1959 1975	0.0005
Early Period	0.21	0.33	0.21	29	1223	-0.468	0.089			0.261	0.259	na	0.043
Later Period	0.38	0.43	0.35	17	3690			-0.255		-0.122		na	0.019
Extended	0.13	0.11	0.05	46	1559		0.141	-0.132			0.151	na	0.164
White Shrimp													
Original	0.23	0.64	0.57	27	3212	-0.691	0.273		-0.325		0.505	1965 1968	0.0001
Early Period	0.21	0.44	0.35	29	3112	-0.610	0.283		-0.362		0.472	na	0.006
Later Period	0.38	0.53	0.37	17	3391	-0.156	0.237	-0.243		0.215		na	0.045
Extended	0.13	0.32	0.26	46	3023	-0.306	0.208		-0.353		0.327	na	0.003

Harvest	r^2_{null}	r^2	$r^2_{adj.}$	n	y-intercept	Bi-Monthly Inflow Periods						Outlier Years	*p*
						JF	MA	MJ	JA	SO	ND		
Black Drum													
Original	0.25	0.51	0.41	25	50.2	-0.030			0.104	-0.064	0.039	1962	0.0053
Early Period	0.25	0.09	0.003	25	67				0.022	-0.023		1984	0.371
Later Period	0.38	0.44	0.25	17	114	-0.026	0.025		-0.063	-0.013		na	0.11
Extended	0.15	0.13	0.06	42	89	-0.008		-0.005		-0.008		1984	0.15
Southern Flounder													
Original	0.25	0.60	0.54	25	-12	-0.031			0.054		0.049	1974	0.0002
Early Period	0.24	0.54	0.45	25	2.34	-0.030			0.079	-0.030	0.045	na	0.002
Later Period	0.38	0.31	0.21	17	43				-0.011		-0.003	na	0.077
Extended	0.14	0.27	0.19	42	16	-0.017			0.039	-0.029	0.035	na	0.017
Red Drum													
Original (1962-1981, ln harvest)	0.33	0.66	0.59	19	3			0.0004	0.002	0.0007		1963	0.0009
Early Period (1962-1981, no transformation)	0.32	0.50	0.41	20	7.5			0.012	-0.039	0.025		na	0.010
Spotted Seatrout													
Original (1962-1981, ln harvest)	0.32	0.57	0.52	20	8	-1.824					1.425	na	0.0007
Early Period (1962-1981, no transformation)	0.32	0.50	0.45	20	204	-0.103					0.103	na	0.003

Task 1a: Analysis of Extended Harvest Data (1959-2004)

Action: We conducted an analysis of the productivity-inflow relationship based on an extended harvest dataset for eight commercially important species for years 1962-2004, using the same protocol for determining model selection and outlier designation as described previously.

Results: Freshwater inflows (as bi-monthly inflows) explain very little of the variation in the extended dataset (Table 2, *Extended*). In all but a few species, r^2 failed to exceed r^2_{null}. Upon further investigation, we found that the hydrologic regime post-1988 differed from that prior to 1988 (Table 3, Figure 3). For example, average annual inflow for the 29 years comprising the *Early Period* (1959-1987) was 9.91 million acre feet per year. Average inflow for the 17 years of the *Later Period* (1988 to 2004) was 14.88 million acre feet per year. We therefore conducted harvest regression analyses to compare freshwater inflow relationships during the *Early* (pre-1988) and the *Later* periods (post-1987).

Table 3. Comparison of hydrologies for the Trinity-San Jacinto Estuary for several periods from 1941 to 2004.

Period	Average Inflow (million acre ft/year)	Time Category	Number of Years
1941 - 2005	11.27	Full period of record	65
1959 - 1987	9.91	Early Years (shrimp)	29
1962 - 1987	9.66	Early Years (other species)	26
1959 - 2004	11.74	Extended (shrimp)	46
1962 - 2004	11.72	Extended (other species)	43
1988 - 2004	14.88	Later Years	17

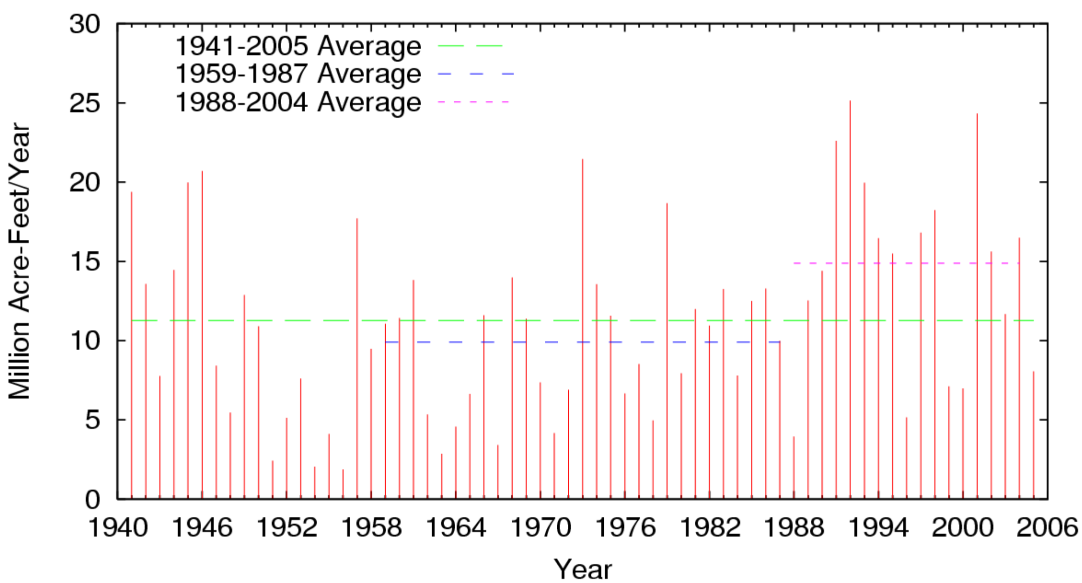

Figure 3. Total annual freshwater inflow (gaged and ungaged) into the Trinity-San Jacinto Estuary for 1941-2005. Long-term average annual inflow (green line) is 11.27 million acre ft/year. *Early* period (1959-1987) average annual inflow (purple line) is 9.91 million acre ft/year, and *Later* period (1988-2004, pink line) is 14.88 million acre ft/year.

Task 1a: Analysis of Later Years Harvest Data (1988-2004)

Action: We conducted a regression analysis considering only harvest years after 1987, thus 1988-2004.

Results: As compared to the *Extended* analysis, freshwater inflows (as bi-monthly inflows) explain a greater percent of variation in the *Later Period* dataset (Table 2, *Later,* 1988-2004) and is comparable to results obtained in the *Early Period* (Figure 4 1962-1987) analysis. This suggests that either the hydrologic regime post-1988 differed from that prior to 1988, or some other factors are contributing to differences between the two time periods. In this analysis, significant models ($p<0.05$ and $r^2 > r^2_{null}$) were obtained for: blue crab (*Callinectes sapidus*), eastern oyster, brown shrimp (*Farfantepenaeus aztecus*), and white shrimp (*Litopenaeus setiferus*). Black drum and southern flounder (*Paralichthys lethostigma*) were not significant. Red drum and spotted seatrout were not included in this exercise, because commercial harvest of these species ceased after 1981.

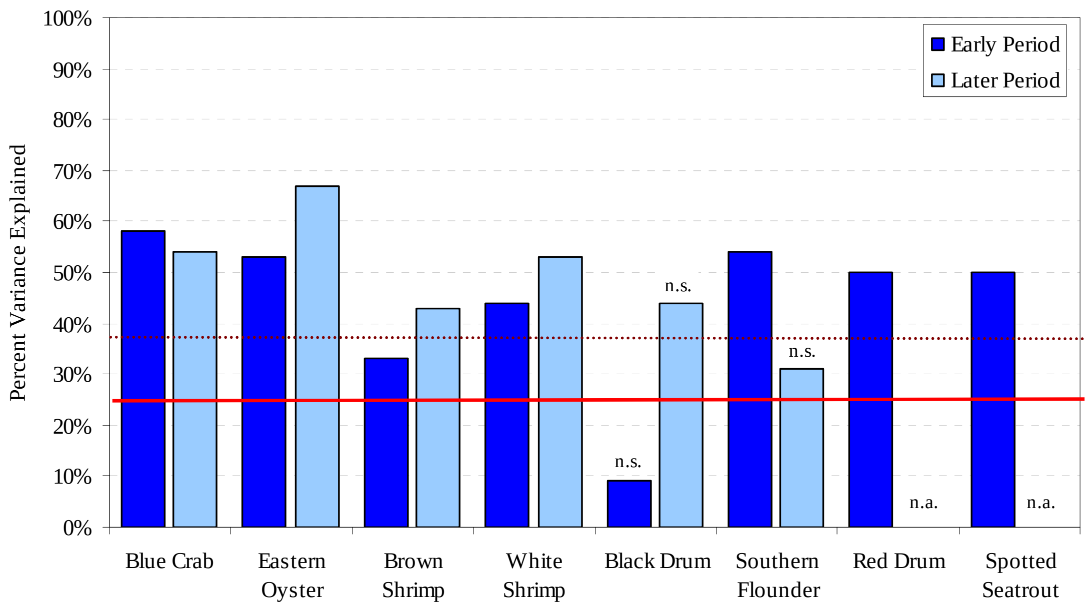

Figure 4. Percent of variance (r^2) explained by bi-monthly inflows on commercial harvest for two time periods: *Early* (1962-1987) and *Later* (1988-2004). The horizontal red lines represent general *rule-of-thumb* thresholds above which an analysis may be considered biologically relevant. The lines are based on a calculated r^2_{null} for the *Early Period* (solid red) and *Later Period* (dashed dark red). All models are statistically significant, except for black drum both periods and southern flounder later years, as indicated by n.s. Species-specific values are provided in Table 1. Red drum and spotted seatrout were reclassified as game species prior to the *Later* period and so not included in the *Later* analysis, as indicated by n.a.

Task 1b: Develop productivity-inflow regression equations using fisheries-independent (TPWD Coastal Fisheries) data and apply to TxEMP analysis

Action: Task 1b included analyses of productivity-inflow regression equations for the eight commercially important species used in the 2001 freshwater inflow study (Lee *et al.* 2001). This analysis utilized TPWD fisheries-independent bag seine data for seven species (1978-2004) and oyster dredge data for eastern oyster (1986-2004). Since individuals in bag seine samples represent mostly juveniles (generally <1 year old), corresponding freshwater inflows were lagged to reflect the relatively short-term influences on juvenile stages (typically <1 water year). Species-specific lags were assigned based on previous studies published for the Laguna Madre (TPWD 2004) and Sabine (TPWD 2005) estuaries. Eastern oyster data was obtained from oyster dredge samples (1986-2004) and so is not necessarily representative of juvenile stages. Therefore, a two-water year freshwater inflow lag was applied to the oyster analysis. Species abundances were expressed as average annual catch per-unit effort (CPUE) for bag seine (normalized to 0.03ha) and oyster dredge samples.

Because of the hydrological complexity and large spatial scale of the estuary, analyses were conducted for the whole estuary (TPWD segment 0.0) and for three of the largest sub-bay segments: Trinity Bay (segment 0.330), East Bay (segment 0.150), and the middle stretch of Galveston Bay extending from Baytown to Bolivar Roads (segment 0.180, Figure 5). We recognize that many smaller segments, such as those in the lower portion of the San Jacinto River and in the sub-bays along the western edge of Galveston Bay, were left out of the sub-bay analyses. Also in this first attempt at a sub-bay analysis, we did not geographically partition freshwater inflows by bay location, but rather compared regional species abundance data to total estuarine inflow. We would like to include both of these refinements in any future work.

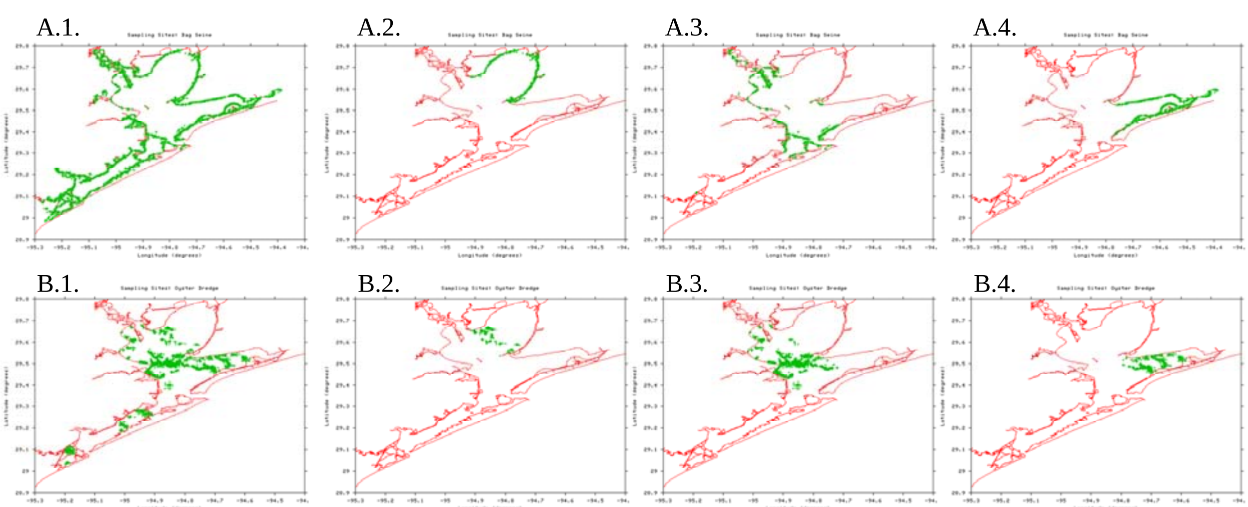

Figure 5. Whole and sub-bay designations based on TPWD sampling locations in the Trinity-San Jacinto estuary. Series *A* represents bag seine; series *B* represents oyster dredge sample locations by 1) whole bay, 2) Trinity Bay (TPWD segment 0.330), 3) mid bay (segment 0.180), and 4) East Bay (segment 0.150).

Sampling effort increased in later years; therefore, we used a weighted least-squares regression analysis. This differs from the previous approaches outlined in the Laguna Madre (TPWD 2004) and Sabine (TPWD 2005) freshwater inflow studies where standard ordinary least squares regression analysis was conducted. We used the same protocol for model selection and outlier designation as in the harvest regression analyses.

Results:

- This analysis mimicked the harvest analyses using TPWD fisheries-independent (CPUE) data, but differed by focusing on juvenile life stages for all species except eastern oyster. Inflows were again aggregated into six bi-monthly predictor variables, but the lagging periods (typically one water year) were shorter than those used in the harvest analysis. While analysis of the commercial harvest dataset is informative, it appears that upon comparison, the TPWD fisheries-independent dataset offers more robust productivity-inflow regression equations and greater flexibility for further refinements to the analysis.

- For seven of the eight species, freshwater inflows, in at least one bay area, explained a significant source of variation in annual abundance (Table 4). Of the 32 species x bay analyses, 16 were statistically significant. Eastern oyster and black drum were the only species to have a significant relationship in *all* bay regions, whereas red drum had no significant relationship to freshwater inflow in *any* region.

- Only three species (blue crab, eastern oyster, and black drum) demonstrated a significant response to freshwater inflow when analyzed across Whole Bay. It is not surprising that these were the only significant whole bay models, because most of the data analyzed was for juveniles, which may demonstrate spatially-distinct affinities for particular regions of the estuary. For those species with a significant Whole Bay model, this was selected as the representative, or most important, model for these species (see below).

- Of the 16 significant models, most produced r^2 values that were appreciably higher than the value of r^2_{null} (Figure 6 and Table 4). From these, the best (most important) model for each species was selected and is listed below. For these, model r^2 ranged 0.45 - 0.67, except for blue crab which had $r^2 = 0.27$ (which is only slightly higher than its $r^2_{null} = 0.23$). While eastern oyster had the best relationship ($p = 0.002$) in East Bay (Table 4), the whole bay model was selected to represent the productivity-inflow relationship for this species.

Summary of best model selected for each species:

Whole Bay	Blue Crab Eastern Oyster Black Drum
Trinity Bay	Southern Flounder Spotted Seatrout
Mid Bay	Brown Shrimp White Shrimp
(No acceptable model)	Red Drum

Table 4. Summary results for weighted least squares regression analyses of TPWD fisheries-independent data. Resulting equations represent whole and sub-bay annual catch per unit effort (CPUE) data for bag seine samples for all species (1978-2004), except eastern oyster (1986-2004) which is based on dredge samples. Explanatory variables include six bi-monthly inflow aggregations. Flows were lagged by either one water year (1WY) or one calendar year (1 CY) relative to *catch year* as indicated next to the species name. Bay segments include: whole bay (all of the Trinity-San Jacinto Estuary); Trinity Bay (TPWD segment 0.330); East Bay (segment 0.150); and, the middle portion of Galveston Bay (segment 0.180, see Figure 4). Significant bi-monthly inflow predictor variables are highlighted to indicate negative (darker shading) or positive (lighter shading) responses to inflow. The most important model for a given species is highlighted in yellow. LN(bay) denotes when a natural log transformation of the abundance data was necessary to meet the assumptions of linear regression.

TPWD - CPUE	r^2_{null}	r^2	$r^2_{adj.}$	n	y-intercept	Bi-Monthly Inflow Periods						Outlier Years	*p*
						JF	MA	MJ	JA	SO	ND		
Blue Crab (1WY)													
Whole Bay	**0.23**	**0.27**	**0.21**	**27**	**3.83**			**-0.00009**		**-0.00034**			**0.022**
Trinity	0.24	0.29	0.20	26	4.37		0.00018	-0.00019		-0.00047		1987	0.051
Mid	0.23	0.29	0.20	27	4.36			-0.00028		-0.00024	-0.00024		0.045
East	0.23	0.29	0.19	27	4.06	0.00042	-0.00024			-0.00047			0.047
Eastern Oyster – Dredge (2WY)													
Whole Bay	**0.33**	**0.57**	**0.48**	**19**	**45.95**		**0.00533**		**-0.01507**		**-0.00339**		**0.005**
Trinity	0.33	0.47	0.36	19	85.36				-0.02987	0.00962	-0.00981		0.022
Mid	0.33	0.53	0.44	19	45.76		0.00542		-0.01288		-0.00368		0.009
East	0.33	0.56	0.50	19	29.94		0.00873		-0.01906				0.002
Brown Shrimp (1WY)													
Whole Bay	0.23	0.06	-0.02	27	14.40					-0.0077	0.00094		0.495
Trinity	0.24	0.09	0.02	26	12.18			-0.0014		0.00065		1982	0.319
Mid	**0.23**	**0.51**	**0.46**	**27**	**4.87**		**-0.00048**				**0.0021**		**0.0002**
East	0.23	0.10	0.03	27	14.17		-0.0015	0.0017					0.265
White Shrimp (1CY)													
Whole Bay	0.24	0.25	0.11	26	13.48	0.0030		0.0029		0.0032	-0.0033	1982	0.180
Trinity	0.23	0.16	0.09	27	16.40		0.0042	0.0034					0.129
Mid	**0.24**	**0.67**	**0.65**	**26**	**1.01**					**0.0041**	**-0.0011**	**1984**	**<0.0001**

TPWD - CPUE	r^2_{null}	r^2	$r^2_{adj.}$	n	y-intercept	Bi-Monthly Inflow Periods						Outlier Years	*p*
						JF	MA	MJ	JA	SO	ND		
East	0.23	0.34	0.23	27	17.30	0.0077		-0.0067	0.0070		-0.0052		0.046
Black Drum (1WY)													
LN (Whole)	**0.23**	**0.52**	**0.48**	**27**	**-0.98**	**-0.00019**					**-0.00037**		**0.0002**
LN (Trinity)	0.23	0.29	0.23	27	-1.36			-0.00052			-0.00158		0.017
Mid	0.23	0.30	0.21	27	0.21		-0.00002	-0.00002		-0.00003			0.037
East	0.23	0.30	0.24	27	0.55	-0.00005					-0.00008		0.013
Southern Flounder (1WY)													
Whole Bay	0.23	0.07	-0.003	27	0.33	-0.00001			-0.00005				0.395
Trinity	**0.23**	**0.47**	**0.40**	**27**	**0.26**	**-0.00003**			**-0.00008**		**-0.00003**		**0.002**
Mid	0.24	0.18	0.11	26	0.12			-0.00002			0.00002		0.099
East	0.23	0.05	-0.03	27	0.38				-0.00005	-0.00003			0.554
Red Drum (1CY)													
Whole Bay	0.23	0.06	-0.02	27	0.56					-0.00005	-0.00004		0.504
Trinity	0.23	0.04	-0.04	27	0.64		-0.00010			-0.00004			0.631
Mid	0.23	0.11	0.04	27	0.03		0.00009			0.00002			0.237
East	0.23	0.15	0.08	27	0.26					-0.00005	0.00011		0.149
Spotted Seatrout (1WY)													
Whole Bay	0.23	0.08	0.01	27	0.27			-0.00002	0.00010				0.355
Trinity	**0.23**	**0.45**	**0.38**	**27**	**0.34**		**0.00011**	**-0.00007**			**-0.00008**		**0.003**
Mid	0.23	0.07	-0.01	27	0.13			-0.00001			-0.00001		0.410
East	0.24	0.13	0.06	26	0.31			-0.00002			-0.00003	1978	0.198

- Within a species, the significant bi-monthly inflow predictor variables change depending on bay or sub-bay region. This result is not necessarily surprising given the size of the bay, individual species life-history characteristics, and the longitudinal gradient created by freshwater inflows entering the upper estuary from the Trinity and San Jacinto rivers. Each river contributes approximately 55% and 24%, respectively, of the freshwater entering the system. In general however, the species show a fair degree of consistency in inflow predictor variables among bay areas, which is not only consistent with our knowledge of these species but also suggest avenues for further refinement of this approach.

- With the exception of eastern oyster, it is difficult to compare the results of analyses from the TPWD fisheries-independent data (whole bay) with that of the commercial harvest data (*Extended* data) since bag seine data focuses on juvenile life-stages while harvest data focuses on larger, mature size classes. Overall though, models developed using the harvest data required more freshwater inflow predictor variables than models developed with the TPWD data (Table 2 and 4). Also, the TPWD analysis showed that species (juvenile) abundances more often had a negative relationship, than a positive relationship, with increasing freshwater inflows.

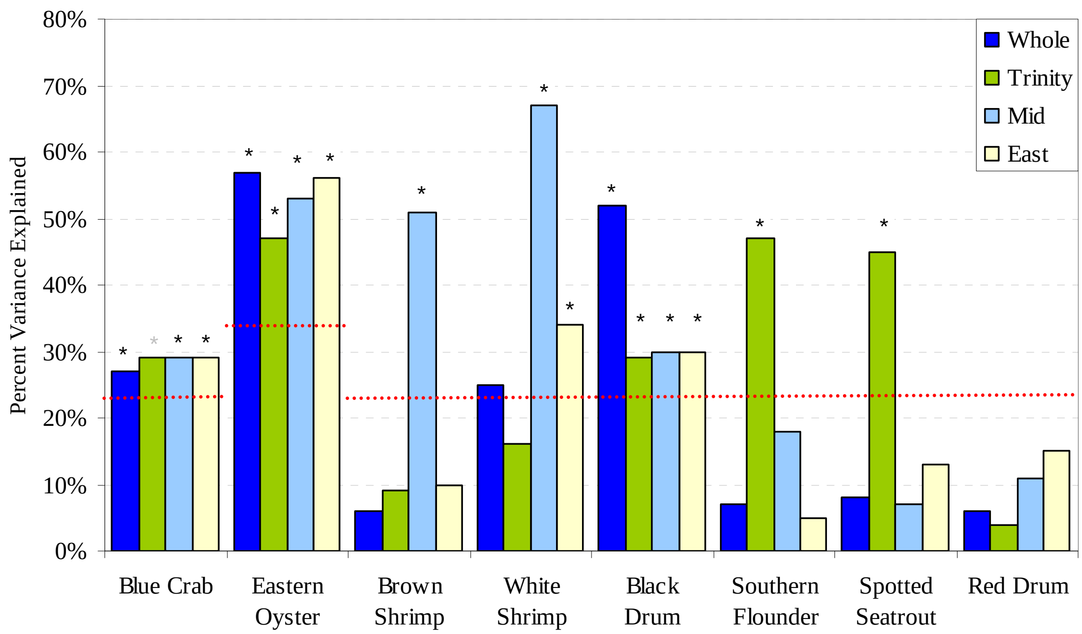

Figure 6: Percent of variance (r^2) explained by bi-monthly inflows on TPWD fisheries-independent data (CPUE) for eight commercially important species in the Trinity-San Jacinto Estuary for whole bay, Trinity Bay, middle portion of Galveston Bay, and East Bay. Statistically significant models are indicated by (*). The red line represents a general *rule-of-thumb* threshold, above which an analysis may be considered biologically relevant. Species-specific values of r^2_{null} are provided in Table 3.

- Analysis of the fisheries-independent data included the period 1978-2004, which includes data from the *Early Period*, pre-1988. We considered the nine years from 1978-1987 to not be a sufficiently long time period to run an Early Period analysis of the fisheries independent data. Since we noted a change in total annual freshwater inflows to the estuary after 1987, we did, however, conduct a *Later Period* analysis on the TPWD fisheries-independent data, but only for the model selected as the most important for each species and excluding red drum as no models were significant for this species. Lending support to the idea that changing hydrologies have influenced the ecosystem, all species, except black drum showed improvements in model r^2 when the analysis excluded pre-1988 years (Table 5). For blue crab, brown shrimp, and spotted seatrout, the *Later Period* analysis explained between 20 – 30% more variation than the full period analysis.

Table 5. Summary results for weighted least squares regression analyses of TPWD fisheries-independent data for the *Later Period*, 1988-2004. Resulting equations represent whole or sub-bay annual catch per unit effort (CPUE) data for bag seine samples for all species, except eastern oyster which is based on dredge samples. Explanatory variables include six bi-monthly inflow aggregations. Flows were lagged relative to *catch year* as indicated next to the species name. Bay segments include: whole bay (all of the Trinity-San Jacinto Estuary); Trinity Bay (TPWD segment 0.330); and, the middle portion of Galveston Bay (segment 0.180, see Figure 4). Significant bi-monthly inflow predictor variables are highlighted to indicate negative (darker shading) or positive (lighter shading) responses to inflow. Only the bay area corresponding to the previously selected most important model for a given species was used in this analysis. Therefore, there is no model for red drum.

TPWD - CPUE	r^2_{null}	r^2	$r^2_{adj.}$	n	Bi-Monthly Inflow Periods						Outlier Years	*p*
					JF	MA	MJ	JA	SO	ND		
Blue Crab (1WY, Whole)	0.38	0.52	0.40	17				-	-	-		0.021
Eastern Oyster (Dredge, 2WY, Whole)	0.38	0.58	0.48	17		+		-		-		0.009
Brown Shrimp (1WY, Mid)	0.38	0.81	0.77	17		-	+			+		<0.0001
White Shrimp (1CY, Mid)	0.38	0.71	0.67	17			+	-				<0.0001
Black Drum (1WY, Whole)	0.40	0.42	0.33	16	-					-	2000	0.031
Southern Flounder (1WY, Trinity)	0.38	0.55	0.45	17	-			-		+		0.013
Spotted Seatrout (1WY, Trinity)	0.38	0.71	0.61	17		+	-		+	-		0.003

Task 2

Task 2a: Assist in the development of new approaches, which will improve and extend the current methodology for relating bay health to freshwater inflows

Ultimately, the sub-bay analysis of the TPWD fisheries-independent data set (presented previously as Task 1b) was determined to be the most useful. This analysis was the most analogous to the analysis of the fisheries-dependent data that still provided potentially biologically significant relationships. Several exploratory analyses were conducted, however, before this conclusion was reached.

Initial analysis of average annual abundance and bi-monthly inflow

- *Action:* The first attempt to analyze the TPWD fisheries-independent data was a linear regression of annual CPUE vs. bi-monthly flows for each gear type of brown and white shrimp. Model and outlier selection was based on the same criteria as described in Task 1a.

- *Results:* These regressions generally did not produce significant results and had low r-squared values. Thus, we determined that different temporal and spatial scales as well as additional explanatory variables should be explored for analysis of this data set.

Initial analysis of annual average abundance and average annual salinity

- *Action:* A regression of annual whole-bay CPUE for brown and white shrimp vs. annual salinity was performed for each gear type. Model and outlier selection was based on the same criteria as described in Task 1a.

- *Results:* This in general produced stronger relationships than inflows, so we concluded that approaches that explore the direct relationships between abundance and environmental variables should be investigated.

Analysis of effect of environmental variables on raw abundance data

- *Action:* A logistic regression of individual observation CPUE (i.e the raw data observations from TPWD) for brown shrimp during the peak abundance months (March to July) versus seasonal (month) and environmental variables (salinity, temperature, dissolved oxygen). The raw data had many zero values, so a logistic regression was first performed to determine which environmental factors might predict the presence/absence of shrimp in the bay.

- *Results:* Month and salinity were significant predictors of presence and absence, but it was concluded that the resolution of the individual data observations was too fine for meaningful analysis and thus it was decided to aggregate the data.

Additional analyses on the effect of environmental variables on abundance data

- *Action:* Using TPWD fisheries-independent data, we continued to explore the direct relationship of inflows, salinity, and temperature on average abundance. For the

A plot of four predictor variables (month, average salinity, average dissolved oxygen, and average temperature) versus the dependent variable (ln(avg.catch)) was conducted to examine the data for the violation of the assumptions of regression. In almost all cases there was a strong non-linear relationship apparent between month and the dependent variable. This was by far the strongest non-linear trend. For some sub-bays there were slight non-linear trends observed for the other predictor variables. Based on this observation and a preliminary exploratory analysis of several non-linear models, a quadratic term ($month^2$) was added to the set of predictor variables.

A best subsets approach was used (in Minitab) to examine which combination of predictor variables could maximize the adjusted r-squared and Cp (Mallow's index). In most cases month and $month^2$ seem to produce the best regression model. In a few cases (e.g. brown shrimp in East Bay) a regression model with month, $month^2$, and an environmental predictor would slightly improve the model (an adjusted r-squared of 66% versus 65%).

The findings from the best-subsets analysis led to a two-step approach to search for further relationships between the environmental predictors and the ln(avg.catch). The quadratic seasonal regression was first performed for each species for each bay (ln(avg.catch) versus month and $month^2$). Residuals for each observation were then calculated. A second regression was then performed using all three of the environmental predictor variables on the residuals. *This two-step process recognizes that the seasonal trend is the strongest predictor of average shrimp abundance in these sub-bays, but explores for a linear relationship between the environmental predictors and average abundance once the seasonal effect has been removed.*

- *Results:* In all cases there appeared to be no significant linear relationship (Table 6) between the residuals of the monthly quadratic regression and the environmental variables (white shrimp in West Bay produced the relationship closest to significant, but it also had a very low r-squared value). This approach attempts to find a linear relationship between the environmental predictors and average abundance (CPUE) once the seasonal effect has been removed. It is important to recognize that many of these environmental variables may fluctuate seasonally. Thus fish abundance may vary seasonally due to biologically hard-wired migration patterns in the fish or because of a biological response to environmental variables that fluctuate seasonally. This two-stepped approach is a rather aggressive way to examine the relationship between the environmental variables and shrimp abundance. The fact that no such relationship was found suggests that if a relationship between the average abundance and the environmental variables beyond the fluctuations that occur monthly is present, then it is too weak to detect at this temporal and spatial resolution. To rule out the possibility of a non-linear relationship between the residuals and the environmental variables, a scatterplot of the residuals versus each of the environmental variables was produced (see Figure 7 for an example). Again, if such a relationship does exist it is weak. These results were true for both shrimp species for all three sub-bays.

Table 6. Summary of the two step analysis of TPWD data. For each species, a separate analysis was performed for each of three sub-bays (East, West, and Trinity). The first analysis shown was an analysis of log transformed monthly average catch for the sub-bay using a quadratic combination of month. The second analysis used the residuals of the first analysis as the dependent variable and then used the three environmental variables (salinity, temperature, and dissolved oxygen) as predictors.

Species/ Version	r^2	adj. r^2	Avg Sal	Avg Temp	Avg DO	p
Brown Shrimp East						
Quadratic for Month	65.7	65.1	**	**	**	0.0001
Analysis on residuals	1.3	0.0	0.219	0.970	0.895	0.671
Brown Shrimp West						
Quadratic for Month	63.1	62.5	**	**	**	0.0001
Analysis on residuals	0.8	0.0	0.961	0.464	0.720	0.804
Brown Shrimp Trinity						
Quadratic for Month	37.2	36.0	**	**	**	0.0001
Analysis on residuals	0.41	0.31	0.616	0.689	0.351	0.602
White Shrimp East						
Quadratic for Month	39.4	38.9	**	**	**	0.0001
Analysis on residuals	0.1	0.00	0.718	0.962	0.876	0.984
White Shrimp West						
Quadratic for Month	31.2	30.4	**	**	**	0.0001
Analysis on residuals	3.8	2.2	0.645	0.073	0.012	0.072
White Shrimp Trinity						
Quadratic for Month	39.8	39.0	**	**	**	0.0001
Analysis on residuals	0.2	0.0	0.779	0.836	0.647	0.955

** r-squared values reported as a percentage

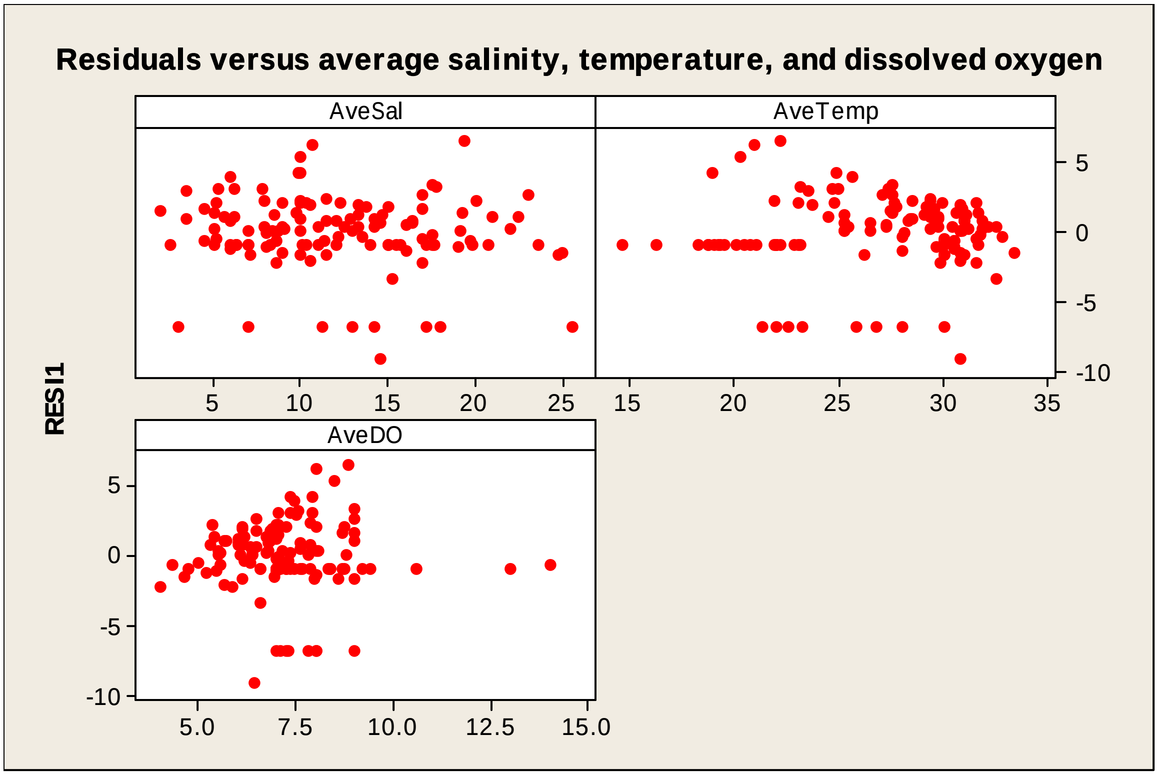

Figure 7. A quadratic function of month was used to predict average brown shrimp bag seine abundance during peak months. Those predicted values were then used to calculate a residual value (observed value - predicted value) for each observation in the data set. This plot shows a graph of these residuals versus average salinity, average temperature and average dissolved oxygen for brown shrimp in east bays. There appears to be no linear or non-linear pattern to these data, suggesting that any relationship that does exists between abundance and these environmental variables was captured in the monthly trend quadratic equation.

<u>Salinity response curves for each species across coast</u>

- *Action:* Using TPWD fisheries-independent data, an analysis was completed of annual CPUE for each major bay (Sabine Lake, Galveston Bay, Matagorda Bay, etc.) versus annual salinity. This was done for each gear type for all eight species using the TPWD data. This produced a "salinity response curve" for each species across all bays. Regression analysis was then applied to look for a relationship between annual CPUE and annual salinity. Model and outlier selection was based on the same criteria as described in Task 1a.

- *Results:* **Brown Shrimp**

  <u>Seine</u> - The seine analysis produced a significant (p-value <0.001), but very weak (adj. r-squared value 7%) negatively sloped linear relationship between annual CPUE and annual salinity. Inclusion of the Sabine Lake data seems to make the relationship less linear.

  <u>Bay trawl</u> - The bay trawl analysis did not have a significant relationship when the Sabine Lake data was included. A non-linear relationship would likely fit, but was not explored. When the Sabine Lake data was removed the analysis did find a significant (p-value of 0.001), but weak (adj. r-squared 7%) negatively sloped linear relationship.

  <u>Gulf trawl</u>- There was no significant linear relationship found for the gulf trawl data.

- *Results:* **White Shrimp**

  For all three gear types, when you remove the Sabine Lake data, the relationship between CPUE and salinity changes from a non-linear, unimodal relationship (with Sabine included) to a negatively sloped linear relationship (without Sabine)

  <u>Seine</u>- The seine analysis produced a significant (p-value <0.001) but weak (adj. r-squared 6%) relationship. Removal of the Sabine Lake data did not drastically improve the strength of the relationship (adj. r-squared 75).

  <u>Bay trawl</u>- The bay trawl produced a significant (p-value <0.001) and somewhat stronger (adj. r-squared 35%) negatively sloped linear relationship even with the Sabine Lake data included. Without the Sabine Lake data the relationship became even stronger (adj. r-squared 52%).

  <u>Gulf trawl</u>- For the gulf trawl analysis, again it appears the Sabine Lake data tends to make the relationship look less linear. Nevertheless, there was a significant (p-value <0.001) and somewhat strong (adj. r-squared 29%) negatively-sloped linear relationship for the gulf trawl data.

- *Results:* **Blackdrum**

  <u>Seine</u>- Sabine Lake stands out in the Blackdrum data set. Notice for Blackdrum average salinity is lowest of the gradient and so is the annual CPUE seine. So, it could just be that Blackdrum do not like low salinity waters. A non-linear regression shows this relationship. It is significant (p-value = .003, but not very strong adj. r-squared of 5%).

  <u>Bay trawl</u>- If, however, you look at the Bay trawl data, which shows the highest CPUE's of the gradient in Sabine Lake, then it appears that Blackdrum adults, anyway, prefer the low salinity waters. This relationship is significant (p-value of

0.02) and stronger (adj. r-squared of 34%). If you remove the Sabine data, the relationship disappears. The scales of the salinity gradient are the same for the two different gear types so it does not appear that the differences in the trend are do to reaching a physiological limit in one area.
Gulf trawl- The gulf trawl has much higher salinities (as one would expect), but unfortunately it just further complicates the story. The relationship is significant, but much weaker (adj. r-squared of 14%).

- *Results:* **Bluecrab**
Both of the bay data sources (seine and bay trawl) depend heavily on the Sabine Lake data for their relationships between CPUE and salinity.
Seine- The seine data show a weak (adj. r-squared 5%) but significant (p-value 0.003) non-linear relationship.
Bay trawl- The bay trawl data also show a significant (p-value 0.03) but weak (adj. r-squared 3%) non-linear relationship that is once again dependent on the Sabine Lake data. Without Sabine Lake, the relationship remains significant (p-value 0.001), is still weak (adj. r-squared of 7%), but switches to a negatively-sloped linear relationship.
Gulf trawl- A fairly strong (adj. r-squared 48%, p-value <0.0001) non-linear, but negative relationship appears to exists between CPUE and salinity for bluecrab in the gulf trawl data.

- *Results:* **Flounder**
Seine- The relationship for the seine data was not significant. Removal of the Sabine data from seine analysis just further weakens the relationship.
Bay trawl- A significant (p-value <0.001, but weak (12% adj. r-squared) relationship exists for the bay trawl data. This relationship is linear, negatively sloped and depends on the inclusion of the Sabine data (relationship is weaker without the Sabine data).
Gulf trawl- The gulf trawl data is remarkable similar to the bay trawl data. The significant but weak (11% adj. r-squared) is linear and negatively sloped. Again without the Sabine data, the relationship would be much weaker and would probably become non-linear.

- *Results:* **Oyster**
Once again, Sabine stands out as a critical source of data for the analysis.
Seine- The seine data do not show a significant relationship (although it is close with a p-value 0.09) and it is very weak (adj. r-squared 2%). With the Sabine data included, the relationship is non-linear. With it removed the relationship changes to a weak and non-significant negatively-sloped linear relationship.
Bay trawl- The bay trawl analysis also depends critically on the Sabine Lake data. With Sabine Lake included in the analysis, there is a weak (adj. r-squared 3%) but significant non-linear relationship (p-value 0.03). We cannot determine with these data whether Sabine Lake represents the end of a salinity gradient, or whether CPUE trends upwards for lower salinities in a different bay area.
Gulf trawl- There essentially were no oysters caught in gulf trawls.

- *Results:* **Reddrum**

  Seine- No significant relationship exists between CPUE and avg. salinity for the seine data. If one includes the Sabine data, then the relationship looks unimodal, non-linear. If one removes the Sabine data, then the relationship changes to a negative-sloped linear relationship (in both cases these relationships are very weak and non-significant) Again one must ask whether this just because Sabine represents the end of a salinity gradient, or would the CPUE trend upwards for lower salinities in a different bay area?

  Bay trawl - There does not appear to be any trend with bay trawl data.

  Gulf trawl- Too few fish were caught in gulf trawls to do an analysis.

Summary of species response curves

The data from Sabine Lake seem to constantly provoke the question of whether fish there are reacting negatively to lower salinities or if there is something else about that area that causes low CPUE for the species examined. All the analyses above have serious problems with the assumptions of regression and therefore would not be well suited for prediction.

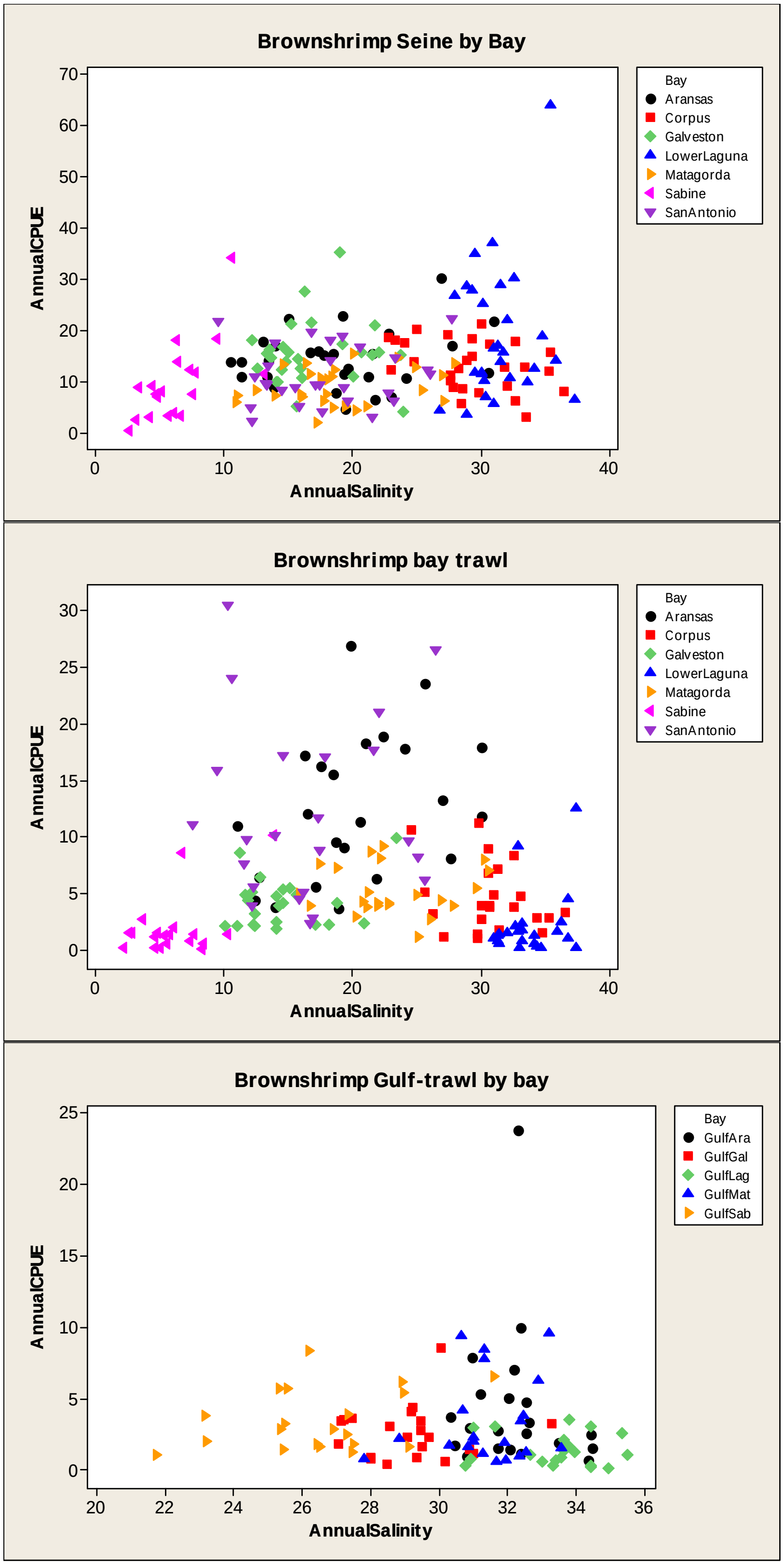

Figure 7. Annual CPUE of brown shrimp vs. annual salinity by bay and gear type.

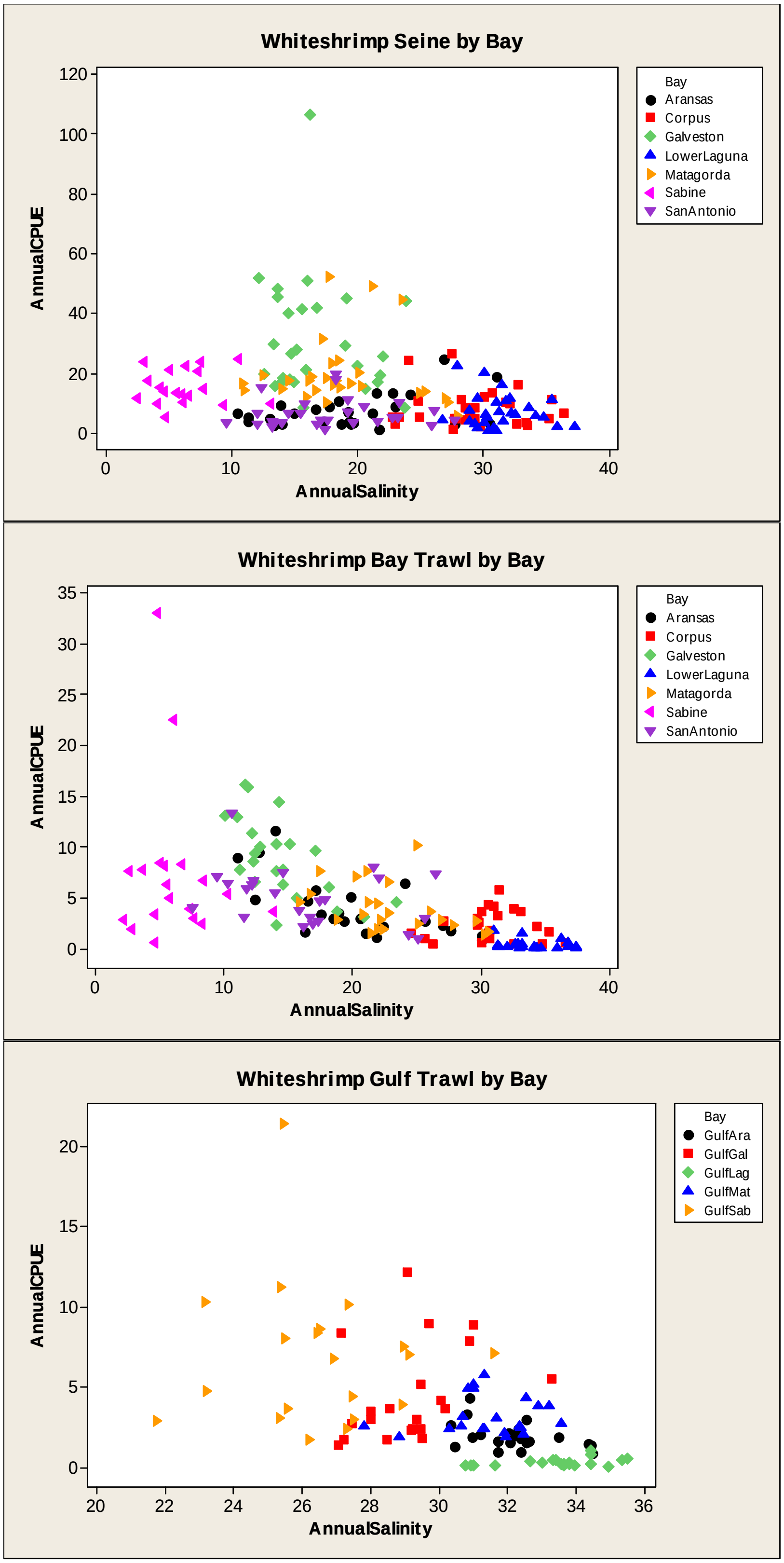

Figure 8. Annual CPUE of white shrimp vs. annual salinity by bay and gear type.

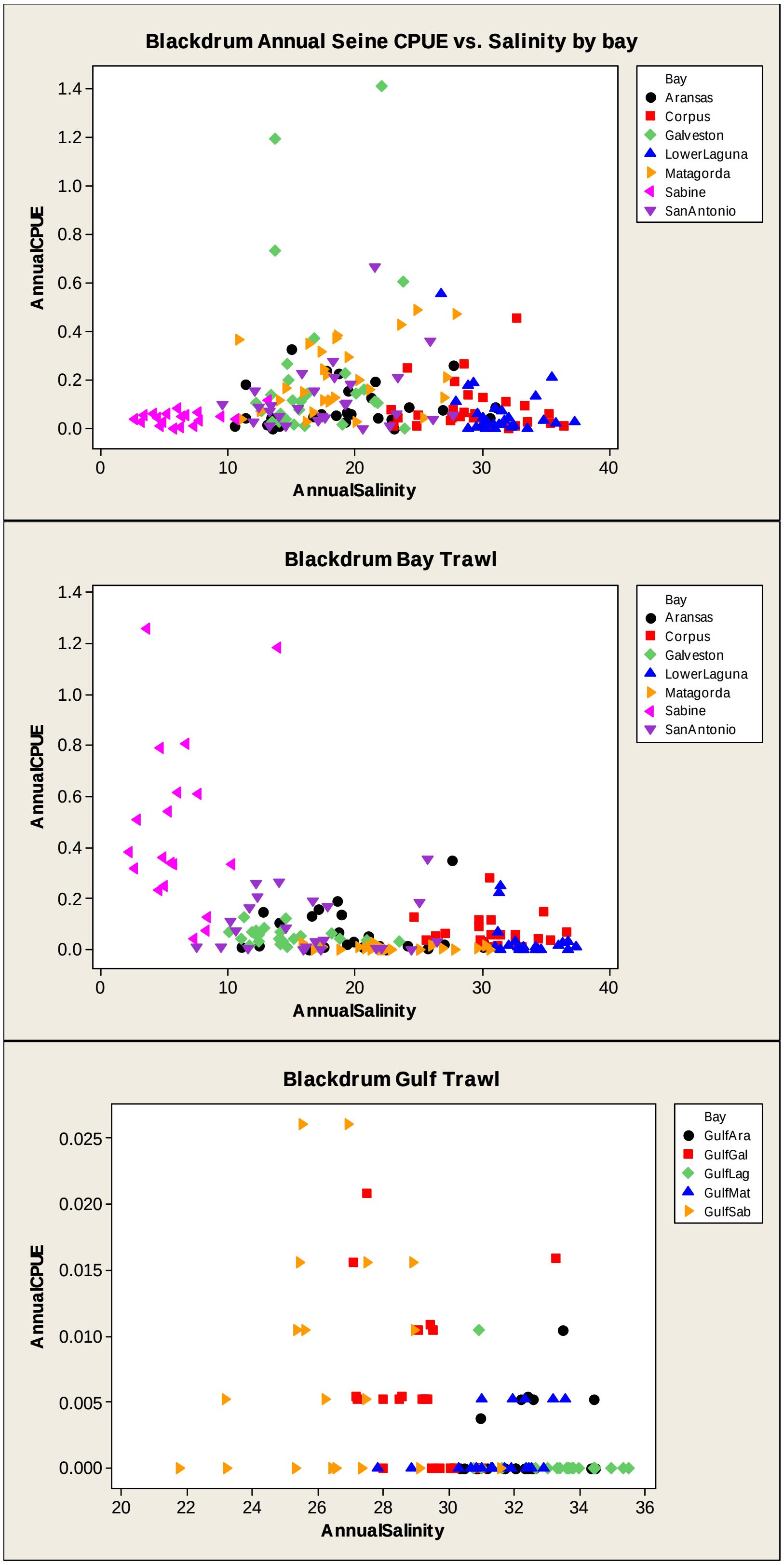

Figure 9. Annual CPUE of blackdrum vs. annual salinity by bay and gear type.

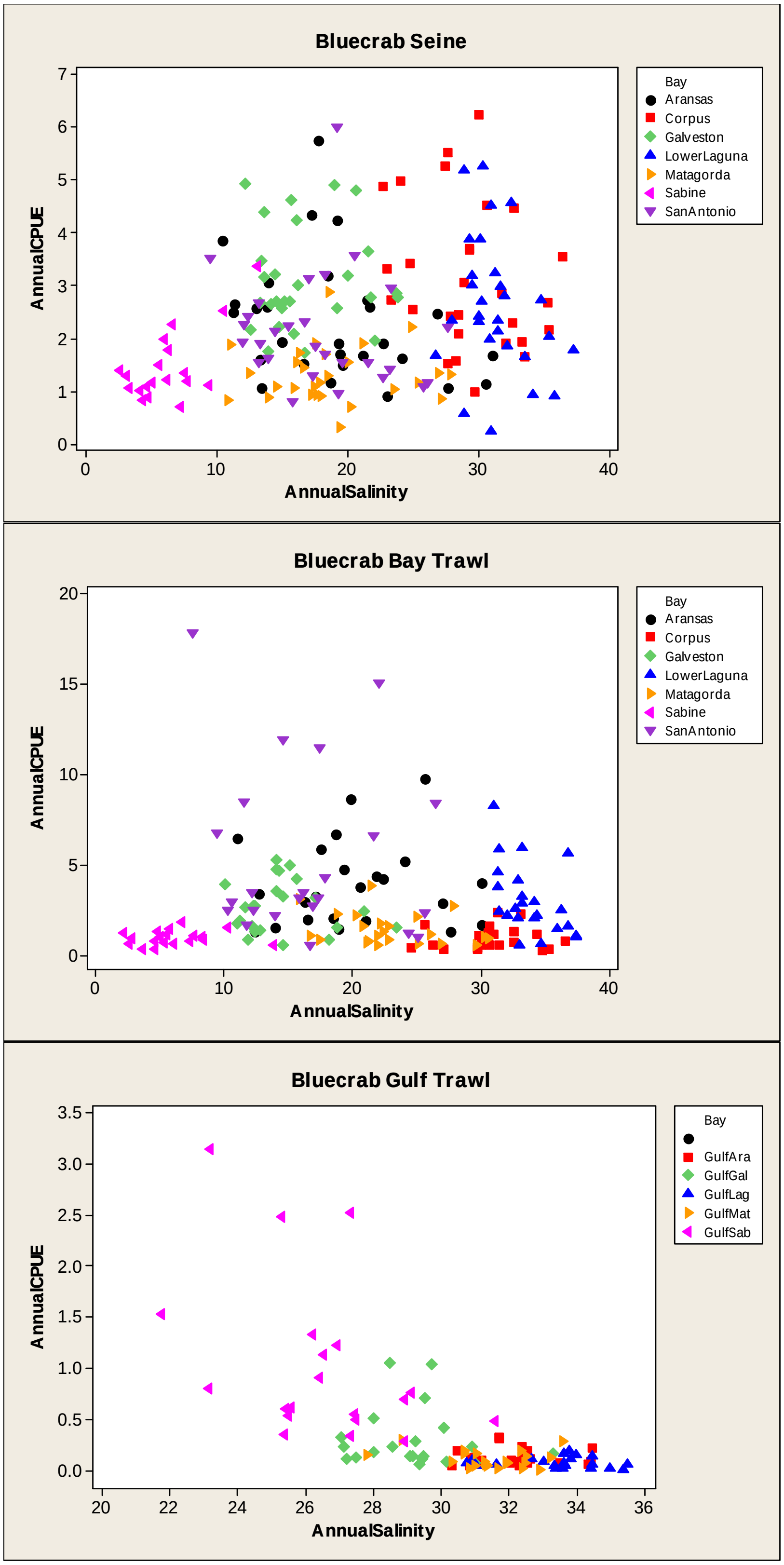

Figure 10. Annual CPUE of bluecrab vs. annual salinity by bay and gear type.

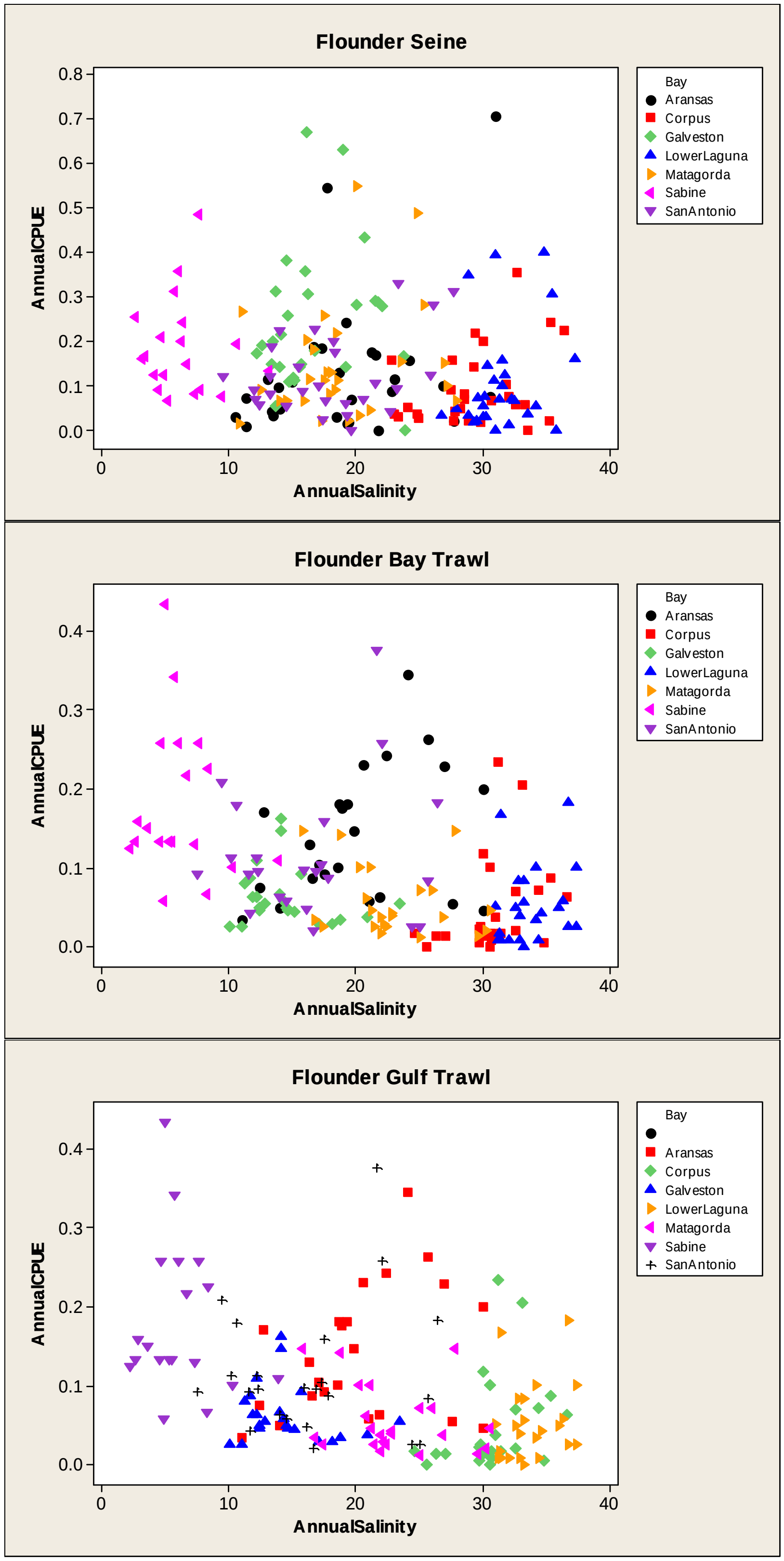

Figure 11. Annual CPUE of flounder vs. annual salinity by bay and gear type.

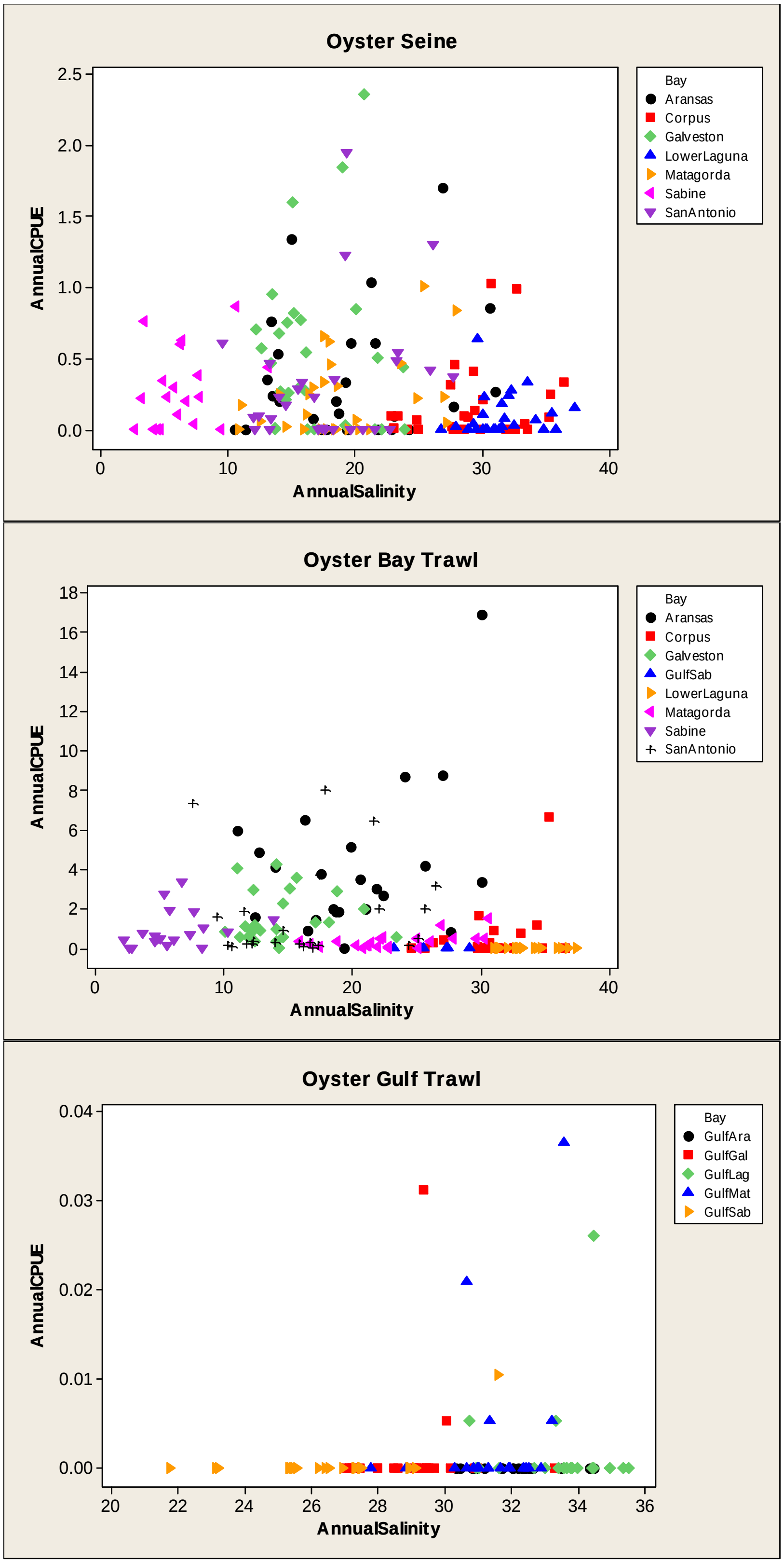

Figure 12. Annual CPUE of oyster vs. annual salinity by bay and gear type.

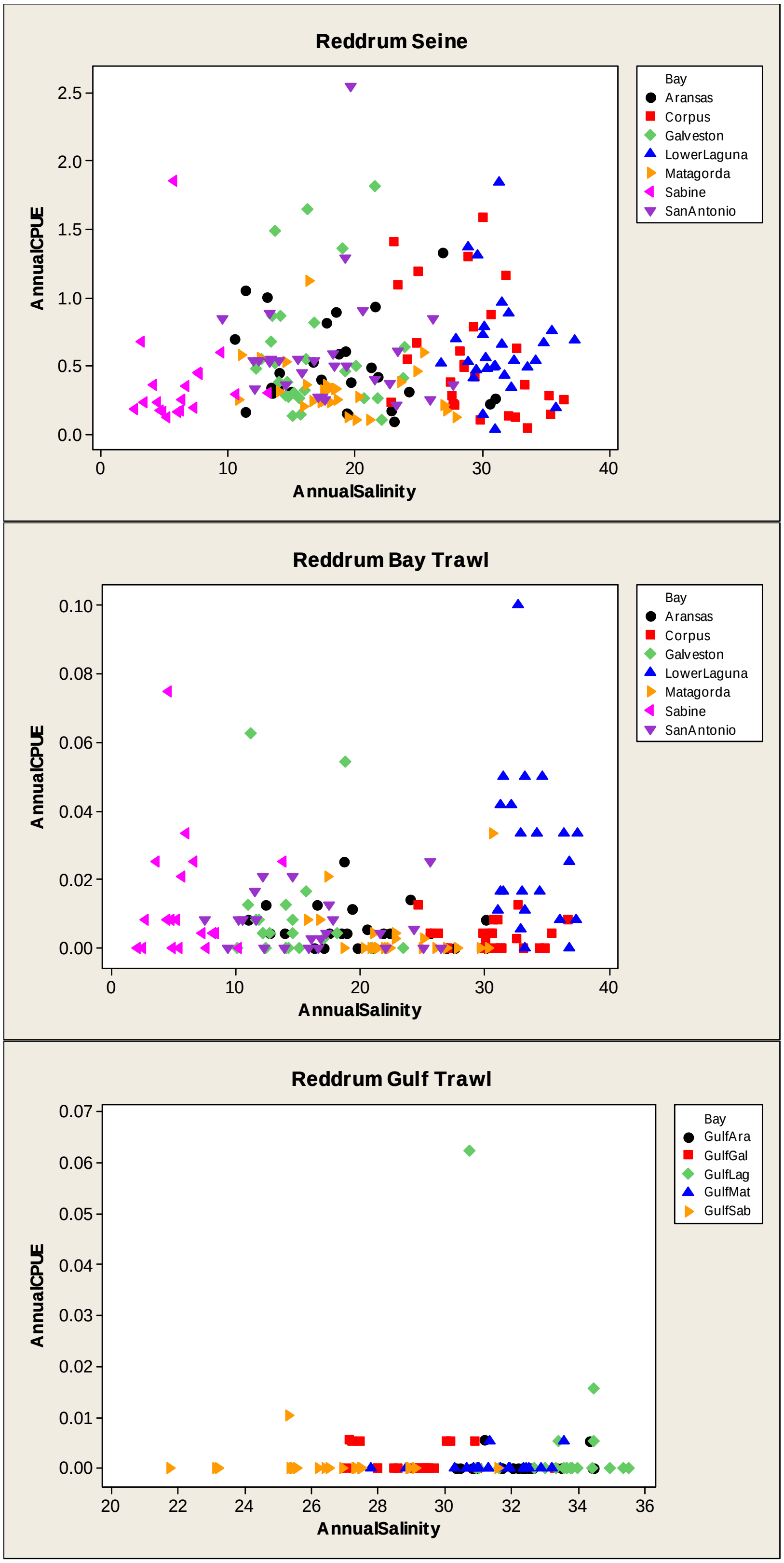

Figure 13. Annual CPUE of redrum vs. annual salinity by bay and gear type.

Task 2b: Assist in the investigation of suitable alternatives to the bi-monthly inflows currently used in the regression methodology.

Analysis of harvest vs.“species”-flow year

- *Action:* Conducted a regression analysis of monthly inflows, defined by biological “species year”, and harvest for six commercially important species: brown shrimp, white shrimp, blue crab, oyster, flounder, and blackdrum. Model and outlier selection was based on the same criteria as described in Task 1a.

  For brown shrimp, white shrimp, oyster, bluecrab, and blackdrum, the biologically relevant inflows were determined to be February-January. This conclusion came from examining peak abundances for the species using the TPWD data and from research on the life histories of the individual species. A February “species:-flow year would mean flows from February 1990-January1991 were used to predict 1990 harvest. Flounder was determined to have an August-July species year. In this case the flows from the previous year’s August-December were combined with the antecedent year’s January-July flows to produce the flow data set for a single harvest year. This was done for all years for which there were harvest data.

  For each species, we performed the analysis on three sets of data: all the years in the data set or “entire data set” (1959-2004), the original or early years subset (1959-1987), and the later years subset (1988-2004) because of the vastly different water regimes between the early subset of the data and the later years subset as previously discussed in Task 1a. For each species we also manipulated the water/flow year to see if the significant predictors were robust to slight adjustments to the definition of flow year. If the flow year was February-January, then changing it to January-December shouldn’t dramatically change the regression if the relationships are meaningful and robust. These multiple analyses were conducted to test whether the significance of different inflow months was robust to changing subsets of data.

- *Results:* **White Shrimp**
  - Percent variation explained: The adj. r-squared for the entire data set was only 19%. Breaking the data set up in to the early and late years did improve the adj. r-squared (as one might predict, given the very different hyrdologies). The early years analysis produced an adj. r-squared of 27% and the late years analysis produced an adj. r-squared of 23%.
  - Significance: The entire data set and early years regressions were significant (p-values of 0.013 and 0.028 respectively) the later years analysis was not (p-value of 0.367).
  - Robustness: Interestingly February and March turned up as important in all three analyses. July was important for the complete and early years analyses and May was important for the later years analysis. This is comforting knowing that the shrimp are most likely migrating into the estuaries during the February/March time frame and therefore would likely respond to inflows during this time. Unfortunately, changing the definition of the flow year from Feb. to Jan. did change the regression. It did not change drastically, but enough to make us

question the robustness of the findings. For a February flow year, significant months were February, March, July, and January. For a January flow year, January, March, May, and July were significant.

- *Results:* **Brown Shrimp-** We used the same biological inflow year definition based on the life history reported in the United States Department of Commerce Brown Shrimp Forecast for 2007. This does not quite coincide with what one might define based on the peak abundance months, but it is not in great conflict with the seasonal distribution.
  - Percent variation explained: The adj. r-squared for the entire data set was only 22%. Breaking the data set up in to the early and late years did improve the adj. r-squared (as one might predict, given the very different hyrdologies). The early years analysis produced an adj. r-squared of 22% and the late years analysis produced an adj. r-squared of 58.%.
  - Significance: The analyses for all three data sets produced significant relationships (entire: p-value of 0.006, early years: 0.038, later years 0.009).
  - Robustness: Interestingly February and March again turned up as important in all three analyses. April was important for the complete and early years analyses; May was important for the later years analyses. This is comforting knowing that the shrimp are most likely migrating into the estuaries during the February-March time frame and therefore would likely respond to inflows during this time. Fortunately, the changing of the flow year from Feb. to Jan. did not change the regression model (i.e. same predictors, same r-squared).

- *Results:* **Blue Crab-** The blue crab analyses yielded no detectable relationship between inflows defined by its species year and harvest.

- *Results:* **Oyster-** There did not appear to be any seasonal trend in the abundance data (with the exception of a small peak in June). Thus because of the seasonality that seems to govern other species, the February flow year was used for oyster as well.
  - Percent variation explained: The adj. r-squared for the entire data set was only 19%. Breaking the data set up in to the early and late years had mixed effects on the adj. r-squared (as one might predict, given the very different hyrdologies). The early years analysis produced an adj. r-squared of 43% and the late years analysis produced an adj. r-squared of 18.%.
  - Significance: The analyses for all three data sets produced significant relationships (entire: p-value of 0.016, early years: 0.003, later years 0.05).
  - Robustness: These regressions were highly sensitive to the data set used. All three data sets had different sets of significant predictors and changing the flow year from February to January altered all the significant predictors except July and August.

- *Results:* **Blackdrum**- Once again February was found to be the best flow year for blackdrum both because of the abundance data and because of the species' life history. Spawning occurs in February, March, and April, thus February seemed like a logical beginning of the flow year. As with previous analyses the blackdrum harvest data set has one year that stands out as a clear outlier (1984). We performed the analysis with the outlier on the entire data set and then removed it for the standard three analyses performed for all the other species (all, early, late). The results reported here are for the analyses with the outlier (harvest year 1984) removed.
    - Percent variation explained: The adj. r-squared for the entire data set was only 17%. Breaking the data set up in to the early and late years did improve the adj. r-squared (as one might predict, given the very different hyrdologies). The early years analysis produced an adj. r-squared of 19% and the late years analysis produced an adj. r-squared of 67.%.
    - Significance: entire: p-value of 0.03, early years: 0.07, later years 0.003.
    - Robustness: No consistency in the significant predictor terms between the early and late year analysis. These regression models were not robust to changes in the definition of flow year.

- *Results:* **Flounder**- Flounder was defined to have an August species year primarily based on the abundance data. TPWD reports that flounder like low salinity waters in the spring, but return to higher salinity waters off-shore in the winter. Thus, August (right before the fall migration) was considered to be the beginning of the species year.
    - Percent variation explained: The adj. r-squared for the entire data set was only 8%. Breaking the data set up in to the early and late years did improve the adj. r-squared (as one might predict, given the very different hyrdologies). The early years analysis produced an adj. r-squared of 40% and the late years analysis produced an adj. r-squared of 42.%.
    - Significance: The later and early years analyses produced significant relationships, but the enitre data set did not (all:p-value of 0.125, early years: 0.005, later years 0.017).
    - Robustness: The analyses were not robust to changes in the flow year or changes in the data set.

Assumptions and limitations of the harvest data set for freshwater inflow analyses

Fish populations are notoriously hard to sample accurately (Rozas and Minello 1997). For this reason, many analyses attempting to estimate fish populations are based on harvest data sets. These data sets have the advantages of usually being quite large (many decades of records), relatively easily available, involving economically important species, and in some ways are less noisy than randomly collected data because fisherman generally attempt to fish where they believe they will catch their desired species (so in some ways are like are like data from a stratified sampling scheme). These data sets, like any data set, have several inherent problems. Harvest data are not only a reflection of population abundances but also a reflection of confounding factors such as market price of the fish being harvested, fuels costs, and new fishing technologies. In addition, the data are not based on a probabilistic sampling scheme and therefore are vulnerable to many sources of bias. The very act of fishing may artificially increase the variability of fish abundances (Stenseth and Rouyer 2008). Thus, it is hard to know whether changes in harvest are due to changes in population size or changes in fishing effort or another source of bias. The analyses presented in this report have the additional problem of lack of robustness when the data set was extended beyond the years of the original analysis. We presented the hypothesis that a significant change in hydrology could affect the bay in a different way as a plausible explanation for why the inflow-harvest regression equations would change. Without an experiment, however, it is difficult to be certain that the change in the hydrology is the reason for the change in the regression equations. Finding a similar pattern in other bay systems along the Texas coast may lend support to this hypothesis and should be investigated.

Assumptions and limitations of the TPWD fisheries-independent data set for freshwater inflow analyses

(*Text for this section was modified from two technical memoranda sent to Galveston Bay Technical Subcommittee: Guthrie and Batchelor 2008 and Guthrie et al. 2008)*

An alternative to using fish harvest data as a measure of abundance is the use of the Texas Parks and Wildlife (TPWD) fisheries-independent data set. This dataset, which describes the abundance of key juvenile species, was insufficient when the original work on Galveston Bay was being conducted. However, for estuaries that were studied later, such as Sabine Lake, the TPWD data was used exclusively. Both datasets have their advantages and disadvantages, but until this study the two datasets have never been directly compared for any one estuary. The TPWD data set was collected using an independent simple random sampling approach. In general, such sampling schemes are preferred by statisticians because they are less likely to produced biased estimates of population parameters. There are some potential limitations and cautions to using this data set, which are outlined below.

1. Bag-seine data, is it an index or an estimate of density?

The bag seine method provides an estimate of population density at 0.03ha and therefore is more than a simple catch per unit effort (CPUE). Since this method does not provide a surrogate measure of density, it is therefore not an index. The bag seine method may not provide an accurate or consistent estimate of population density and thus the use of this data and interpretations of results should be made with caution. Nonetheless, the data available are unique and important to the task.

2. Spatial and temporal heterogeneity – the natural variability in both the spatial and temporal distribution of species.

Estuarine populations are inherently patchy as a result of spatial and temporal heterogeneity. Individuals are not distributed uniformly across available habitat, but rather tend to be clumped. For some species (e.g., brown/white shrimp), the populations are highly seasonal resulting in temporal heterogeneity. Because populations are patchy, we expect that even during peak season some bag seine samples may show zero individuals for particular species. Likewise, other samples may contain numerous individuals. This is OK. Sampling variability will occur in any situation where one is sampling from a population. The fact that these populations are spatially and temporally heterogeneous in their distribution just adds to the variance of the distribution of population densities observed over space and time. A large sample size (e.g., n=240 bag seines/year) allows us to overcome such heterogeneity when estimating mean population density. (In fact, increased precision of an estimate of a mean as sample size increases is the essence of the law of large numbers and the central limit theorem.) We maintain that this source of variability in our data is conceptually different than measurement error (see below) and can be overcome with sufficiently large sample size.

3. Probability of detection, how can this be determined for the bag-seine data?

Two additional sources of variability, in addition to the inherent and real temporal and spatial heterogeneity in abundance that exist for these species, are likely to contribute to the large amount of "noise" seen in this data set. The first source is the idea that the sampling techniques have a probability of detection less than one. In other words, the organisms may be present but are failed to be captured by the sampling technique. This would result in some of the absences recorded in the data set for a particular species for a particular sampling event not being "true" absences. Other ecological studies with similar problems try to estimate a species-specific probability of detection in each sampling event. Software packages are available to assist in this effort (MacKenzie et al. 2002, MacKenzie et al. 2003). Unfortunately, upon inspection we discovered several major obstacles to using the commonly available software package, PRESENCE, for the bag seine data. First, the estimation methods embedded within the software assume that a site is closed to changes in occupancy rate during the period of sampling. We know from the basic life-history of many of these species that abundance and therefore occupancy rates are likely to change dramatically within a year and likely even within a season. Secondly, the software requires at least two repeated visits to the same site within the "closed sampling season". Ideally, for the best estimates, sites would be revisited seven times during a "closed sampling season". TPWD uses a random sampling program to obtain bag seine data. Given the size of Galveston Bay, the probability of revisiting a given site within the same year is extremely low. Thus, the TPWD bag seine data cannot be used with the PRESENCE software to estimate the probability of detection.

MacKenzie *et al.* (2002) also acknowledge that PRESENCE currently does not allow for heterogeneity in detection probabilities across sites. Thus, any correction that could be applied would be a constant factor correction to all of the densities that were used for our analysis. This type of correction in a linear regression analysis is analogous to changing the units of measurement. For example, anthropologists have long used femur length of human skeletal remains to estimate the height of a specimen. This is based on a strong linear regression between femur length and height of humans. If we change the units we measure height from centimeters to meters (i.e., changing all the values of the dependent variable by a constant factor of 100), we have not changed the underlying relationship between femur length and height (Demendonca 2000). Similarly, adding a standard correction factor to estimates of CPUE would not change the underlying freshwater inflow-abundance relationship reported.

Some have recommended that we correct the samples using instead a species-specific *naive probability of detection* which is based on the ratio of the number of samples with $species_i$ to the total number of samples collected. We remain hesitant to adopt this approach, because as described in #2 above, a sample of zero is expected in some cases and is not necessarily due to failure of detection. The use of the *naive probability of detection* confounds the probability of detection with true sampling variability and thus introduces unquantifiable error that would make further statistical inference unreliable.

4. Probability of detection versus Catch Efficiency using bag-seines?

In addressing the probability of detection issue, we realized that a potentially greater issue when estimating population size using bag seine data is that of *catch efficiency*, the second source of variability in the data set in addition to the natural heterogeneity. Bag seines (unlike other sampling methods more familiar to us) tend to lose a fair portion of individuals during the sampling process and therefore can lead to a biased estimate (Steele et al. 2006 & Rozas and Minello 1997). We feel that the low, variable catch efficiency of bag seines may present a greater problem than the issue of detection. However, like detection, neither is easily quantified. Since we are not attempting to estimate absolute population size for the various species, the bag seine data may still be sufficient for the purpose of estimating freshwater inflow needs to Galveston Bay. This requires two key assumptions: 1) catch efficiency within bag seine samples, while unknown, has a well-defined mean and 2) is not subject to systematic bias. In addition, for the purpose of developing freshwater inflow relationships using TxEMP, the model relies on ratios of species abundances, rather than absolute numbers. In the end, we assume there to be a one-to-one correspondence between the measured ratios found within the bag seine samples and population ratios.

5. Inconsistent population responses as measured by bag seines, bay trawl, gulf trawl, and commercial harvest, as well as oyster dredge and oyster harvest.

There are several factors that may prevent any tight correlation from being observed when comparing these gear types. First, these populations (whole and at various life stages) have a wide range of natural variability. Second, the gear types are sampling different habitats and different life-stages of very open populations. Some gear types sample by a fixed area, others by a fixed time (a catch per unit effort). In the case of harvest, the "sampling effort" is targeted for maximum results as opposed to the random sampling used by TPWD. These in combination with differing probabilities of detection and catch efficiencies make it unreasonable to expect a perfect correlation among the sampling methods. In fact, even in closed, closely monitored populations where nearly all individuals can be accounted for, comparisons in population trends among life stages can yield unexpected surprises. Now, in contrast to the bag seine and trawl, the oyster dredge data seems to have a more "perfect" correlation than the other gear types. This may be in part due to the fact that the methodology between TPWD sampling and commercial harvesting is similar, both in gear type and in habitat sampled.

Conclusions and Recommendations

- One should interpret the results of the harvest analyses with caution for the reasons listed above. Nevertheless, we believe that the harvest-inflow regression equations could still be used in the TxEMP process for flow recommendations. This is especially true where or when no other information on fish populations is available, as was the case when the first inflow recommendation study for Galveston Bay was conducted. We feel the constraints employed in TxEMP, particularly the constraints to maintain the historic ratio of the target species and to maintain realistic salinity gradients, guard against the negative consequences of potential inaccuracies or lack of precision in the regression equations.

- While analysis of the commercial harvest dataset is informative, it appears that upon comparison, the TPWD fisheries-independent dataset offers more robust productivity-inflow regression equations and greater flexibility for further refinements to the analysis.

- Given the indirect nature of the relationship between freshwater inflows and the abundance or harvest of the species discussed in this report and the many sources for variation present in the two data sets used, it is our conclusion that it is unreasonable to expect to find strong relationships between freshwater inflows and harvest or abundance. This does NOT imply that freshwater inflows are unimportant in this system. Nor does this mean that these relationships cannot be useful for management decisions. It does mean, however, that these relationships should not be used for predicting individual observations of harvest or abundance. Although weak relationships cannot precisely predict observed abundances in fish, they can be used for model optimization. Models attempt to optimize a mean trend. As long as the mean response in abundance is statistically (and biologically) significant, then it is reasonable to use such a relationship in an optimization model.

- Several directions for further work with the TPWD fisheries-independent data might improve the relationships, although we think it is unlikely that a dramatic improvement will be observed. These directions include:
    1. Aggregate estimates of species abundance from minor bay segments into the larger bay regions for mid-Galveston, Trinity, East and West.
    2. Use *Seasonal* estimates of species abundance for those species which are found only seasonally within the bay, *e.g.* white shrimp.
    3. Use Regional freshwater inflows, as opposed to total estuarine inflow, and their effect on species throughout the estuary at the local (sub-bay) and estuary scale.

- Environmental variables were not helpful in explaining the variation in abundance of individual populations at the spatial scales of Galveston Bay and smaller. Again, this does not mean smaller scale changes in environmental variables are unimportant. It has been observed that environmental variables are useful in explaining variation at this scale at the community level (J. Lester, pers com). Environmental variables at small scales are likely important even for individual populations, but they were not detectable with this data set. The limitations of the data set only allow us to see the relationship between environmental variables and abundance at larger spatial scales. Incorporating environmental variables into a larger scale model, such as a coast-wide analysis, that is then applied to Galveston Bay may be useful.

- Bi-monthly inflows have properties of independence that are statistically convenient. Their designation is, however, somewhat biologically arbitrary. Reducing inflow to a single inflow predictor term (or perhaps two) would have the advantage of potentially being more biologically or hydrologically meaningful. It would also be more statistically flexible; one can explore non-linear combinations of fewer predictor terms more easily. Determining what inflow predictor term is most meaningful biologically, however, is a daunting task that could vary among species and therefore should not be undertaken lightly.

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