WILL SMALL DIVERSIONS OF FRESHWATER INFLOW AFFECT WATER QUALITY?

A Thesis

by

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This thesis meets the standards for scope and quality of
Texas A&M University-Corpus Christi and is hereby approved.

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ABSTRACT

Freshwater inflow is a vital component of an estuary, as several ecological relationships exist between the level of inflow and aspects of estuary function. For future management, it is necessary to know how diversions of freshwater inflow may affect both water quality and ecosystem function within estuaries. The purpose of this study was to examine the effects of variation in freshwater inflow to make inference about water quality variables and estuary function during low inflow periods. This study focused on three bays: Carancahua Bay, San Antonio Bay (including Guadalupe Bay), and Tres Palacios Bay. Data was collected monthly via water quality sampling, and with continuous and discrete multiparameter sondes. Acoustic doppler current profilers (ADCPs) collected current speed and direction daily. Hourly precipitation and wind data was collected from the National Climatic Data Center (NCDC). Daily discharge was collected from two USGS flow gages. Freshwater inflow is responsible for driving nutrient concentrations and salinity ranges, as demonstrated by a principal component analysis. Based on results of this study, San Antonio Bay requires a large amount of freshwater inflow change (above 10,000 ac-ft/mo) to yield changes in water quality response, because it typically receives large volumes of inflow. Conversely, Carancahua Bay and Tres-Palacios Bay both require smaller volumes of freshwater inflow (less than 10,000 ac-ft/mo) to have large (i.e., 30% change) effects on water quality response because these bays receive lesser amounts of inflow. Freshwater inflow also alters net ecosystem metabolism (NEM) of an estuary. Freshwater inflow and salinity both had a significant but weak correlation to NEM when lagged, due to a time lag experienced between drivers and estuary response. The flow-to-waterquality concept created in this study provides a generic framework that can be applied by managers and policy-makers to analyze how specific amounts of flow diverted from, or added to, specific bays may alter water quality conditions.

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INTRODUCTION

Freshwater inflow is often a key factor that influences habitats, communities, and biological productivity in estuarine ecosystems (Montagna et. al., 2013). Inflow is an important factor in the maintenance of estuaries: freshwater inflows are known to maintain salinity gradients, sediment amounts, and loading of water quality variables, such as inorganic nitrogen and phosphorous. The variability of freshwater inflow alters the condition of estuaries, which in turn affects the integrity, function and sustainability of those ecosystems (Alber, 2002; Palmer et. al., 2011).

A potential problem occurs due to anthropogenic changes of the landscape because streams and rivers have been channelized, dammed, and altered for reservoirs. Few estuarine systems in the world are unaffected by upstream manipulation and diversion of natural freshwater inflow. 77% of the total water discharged by 139 large river systems in the northern hemisphere was found to be moderately to strongly affected by alterations such as dams, withdrawals, and other diversions (Dynesius and Nilsson, 1994). As humans continue to develop water resource projects for diverting and capturing freshwater, reductions to inflow may alter the functioning of estuary ecosystems (Montagna et. al., 2013).

Ecological relationships exist between the level of inflow and aspects of estuary function. Nutrients and organic matter delivered by freshwater inflow have been linked to overall estuary productivity, health and function (Russell et. al., 2007). Estuary function can be described by the processing of organic and inorganic materials through various subcomponents of the ecosystem (Montagna et. al., 2011). The volume of flows is related to the rate of nutrient input, and also the rate of nutrient cycling that occurs (Powell et. al., 2002).

Although nutrient delivery is crucial to estuarine production, there is a limit to the level of nutrients necessary to sustain balanced production within these systems (Olsen et. al., 2006). Loss of freshwater inflow can result in less nutrient input and create oligotrophic conditions with low nutrient levels leading to low productivity **(**Kim and Montagna, 2012; Palmer et. al., 2011; Wetz et. al., 2011). In some estuaries, an increase in nutrients can cause dense algal blooms to occur, which may block sunlight to submerged aquatic vegetation (Valiela et. al., 1992; Paerl et. al., 1998; Wetz et. al., 2011). Decaying algae from these blooms may also take up oxygen that would have otherwise been available to the system, and lead to hypoxic conditions (Valiela et. al., 1997).

Discharge-associated changes in nutrient and organic matter delivery can have implications for estuarine productivity (Alber, 2002). Both nutrient and organic loading have been previously linked to estuary metabolic rates (Russell et. al., 2006). Net ecosystem metabolism (NEM), proposed by Odum in 1956, can be used to measure estuarine ecosystem metabolic rates. NEM may be used as an indicator of how alterations to freshwater inflow affects one aspect of an estuary's ecological functioning and how estuarine ecosystems respond to changing environmental conditions (Tang et. al., 2015). NEM is calculated by subtracting aerobic respiration rates from photosynthetic rates for the biological components contained in a body of water (Russell et. al., 2007).

NEM is the net effect of production and respiration and can be used to evaluate if an ecosystem is a source or sink of carbon (Caffrey, 2004). A negative NEM indicates heterotrophic ecosystems where respiration exceeds photosynthesis and where external organic matter, or allochthonous inputs, is dominant over internal sources of organic matter. A positive NEM indicates autotrophic ecosystems that may rely on internal nutrient sources (Russell et. al., 2006; Tang et. al., 2015). The balance between a heterotrophic and autotrophic system may have implications for the system's balance between organic nutrient assimilation and release (Russell et. al., 2007).

Salinity is a key determinant in the habitat characteristics of an estuary. Over seasonal time scales, freshwater inflow serves as the main control of estuarine salinity gradients (Schmidt and Luther, 2002). Decreased freshwater inflows allow for salt water to intrude further upstream. In most estuaries, as freshwater inflows decrease, salinity may increase along the estuarine gradient (Alber, 2002). Coupled with other factors, such as evaporative semi-arid conditions (i.e. low rainfall and high temperatures), this can result in a hypersaline estuary. Changes in the salinity structure of an estuary, in turn, can affect the distribution of estuarine organisms (Gunter, 1961; Underwood et. al., 1998; Telesh and Khlebovich, 2010).

Some estuaries are divided into primary and secondary bays, with different freshwater inflow effects due to the river-to-sea gradient that occurs. Many bays within Texas receive freshwater inflow primarily through rivers draining watersheds into the secondary bays of the estuaries (Sharp et. al., 1986; Montagna et. al., 2002; Palmer et. al., 2011). Freshwater inflow has greater influence on secondary bays than primary bays due to the physical connection between secondary bays and riverine influences (Montagna et. al., 2013) and due to their close proximity. Thus, more study should be directed towards secondary bays because they have a more direct connection to inflow and reduced inflow effects may be more predominant in the upper reaches of secondary bays.

Complicated interconnections exist between the quality, quantity and timing of freshwater inflows and the health of estuaries. A small change in inflow may affect the fundamental functioning of an estuary, which in turn can have ramifications for the biota and humans dependent on the estuary (Olsen et. al., 2006). Given the importance of freshwater inflow, further assessment and study is necessary for future management of estuarine ecosystems. While much attention has previously been focused on estuary structure, much less study has focused on estuary function in relation to inflow (Montagna et. al., 2013). This is another relationship that needs to be further defined.

In recent years, legislative changes have brought hydrological restoration to the forefront of policy. Senate Bill 3 was passed in Texas in 2007. Senate Bill 3 encourages voluntary water and land stewardship to benefit the water in the state. It was also unique in that it adds a permit use for instream flows dedicated to environmental needs or bay and estuary inflows. In 2012, the president signed into action the Resources and Ecosystems Sustainability, Tourist Opportunities, and Revived Economics of the Gulf Coast States Act (RESTORE Act). This ultimately made funding available for hydrological restoration in the Gulf. These legislative changes concerning the effects of altered inflow regimes on downstream bays and estuaries have caused managers and policy makers to put effort towards hydrological restoration.

The Texas Environmental Flows Initiative (consisting of the National Fish and Wildlife Foundation, Harte Research Institute, Meadows Center, National Wildlife Federation, the Nature Conservancy, and Ducks Unlimited) seeks to take advantage of both Senate Bill 3 and the RESTORE Act. The Texas Environmental Flows Initiative recently proposed the creation of an estuarine protection and enhancement plan for the bays located within the Texas Gulf coast. This plan consists of using water transactions as a tool for benefiting environmental flows for Texas bays and estuaries. Part of this plan involves obtaining water rights in certain areas to enhance freshwater flow in order to enhance or restore the ecological benefits of returning freshwater into these. However, the amount of freshwater rights available may be limited, thus only the upper reaches of the estuary (within the secondary bay) may be affected, so it is necessary to know if small changes, such as 10,000 ac-ft/mo., to inflow can have impacts to the bays.

The purpose of the present study was to examine effects of variation in freshwater inflow to make inference about the relationships between water quality variables, estuary function, and freshwater inflow. This was done by measuring changes in water quality variables and calculating NEM over time in comparison to freshwater inflow and salinity ranges. The hypothesis assumed in this study was that small changes to inflow will change water quality and ecosystem functioning within the estuary, most notably in areas that are closer to the river mouth or source of freshwater inflow. The goal of this project is to create a baseline model that can be applied to other systems and contribute to an improved management of freshwater resources. Studying freshwater inflow in relation to estuary water quality and function will help to create improved methods for restoration in areas which have experienced reduced inflow.

METHODS

Study Sites

This study focused on three bays: Carancahua Bay, San Antonio Bay (including Guadalupe Bay), and Tres Palacios Bay because these are areas where freshwater inflow rights might be available. Carancahua Bay and Tres Palacios Bay are a part of the Lavaca-Tres Palacios estuary. The Lavaca-Colorado estuary has two major rivers feeding into it: the Colorado River and the Navidad River. The estuary covers approximately 910 km^2 as a whole. Similar to other Texas bay systems, it is shallow with water depths less than 4 meters (Bao et. al., 1994). San Antonio Bay and Guadalupe Bay are a part of the Guadalupe estuary. This estuary is fed by freshwater inflow primarily from the Guadalupe River and the San Antonio River that merge above the head of the estuary. The estuary is relatively small with an area of 551 km^2. It is shallow, where mean approximate depth is one meter (Slack et. al., 2009).

Five sampling stations were chosen for this study: N1-N5 (see Figure 1). Stations N1 (28.71369 °N latitude, -96.19079 °W longitude) and N2 (28.67166 °N latitude, -96.23936 °W longitude) were located within Tres Palacios Bay. Station N3 (28.728143 °N latitude, -

96.428696 °W longitude) was located within Carancahua Bay. Stations N4 (28.43492 °N latitude, -96.77047 °W longitude) and N5 (28.39352 °N latitude, -96.7724 °W longitude) were located within San Antonio Bay, with N4 being stationed in Guadalupe Bay. All stations were chosen specifically because they are areas that could potentially benefit from freshwater inflows

Two USGS stream gaging stations were located near the sampling locations and recorded daily inflow amounts. USGS Station 08188810 (28.478052 °N latitude, -96.861812 °W longitude) was located above San Antonio Bay. USGS Station 08162600 (28.927778 °N latitude, -96.170832 °W longitude) was located above Tres Palacios Bay. The National Climatic Data Center (NCDC) station at the Palacios Municipal Airport was located between Carancahua Bay and Tres Palacios Bay (28.72472 °N latitude, -96.25361 °W longitude) and recorded daily precipitation and wind amounts.

Hydrographic Data

Hydrographic data was collected using multiple YSI Sondes at surface (directly below the water line) and bottom (roughly 15 cm from the benthos) depths. The sondes collected specific conductivity, salinity, temperature, dissolved oxygen and depth readings continuously every 15 minutes. The sondes were removed and replaced every month with a new sonde. These sondes are calibrated before they are placed in the field and post-calibrated when they returned. They were cleaned to ensure no bio-fouling. Sampling occurred on a monthly basis for water quality variables to assess how concentrations change because of seasonality, and the differences between times of low and high inflow. Continuous sampling occurred over the course of eight months to ensure that all inflow events, such as those caused by rainfall, were captured. Continuous sampling of these stations was compared to discharge readings from nearby U.S. Geological Survey (USGS) stream gaging stations for the same time periods to assess freshwater inflow quantities in relation to water quality variables recorded by the sondes.

On the day of sampling, a sonde was used to measure current conductivity, temperature, salinity, pH, dissolved oxygen, and depth from surface, bottom, and mid depths. Water samples were taken monthly at each station for total suspended solids (TSS) and chlorophyll-*a* from surface, bottom, and mid depths with 2 replicates per depth.

Nortek AquaPro Acoustic Doppler Current Profilers (ADCPs) were deployed at each sample station between 20 October, 2016 to 10 March, 2016. The ADCP units were calibrated to measure water current speed and direction within 0.5 m cells at 15 minute increments. Two ADCP units were horizontal mount style allowing for shallow water deployment and stationed at N3 and N4. The remaining units were vertical mount styles with a minimum deployment water depth of 1.0 m, and were stationed at N1, N2, and N5.

Chlorophyll-*a*, Nutrients, and Turbidity

Two replicate water samples were collected at the surface and at roughly 15 cm from the bottom using a Van Dorn sampler. There was a total of four chlorophyll cartridges per station. Two 14-ml sub-samples were collected for nutrient analysis from a 1-L Van Dorn bottle and filtered on site using a hand syringe and 0.7-µm glass fiber filters. Water samples for chlorophyll-*a* were filtered through 0.7-µm glass fiber filters (Welschmeyer, 1994).

Nutrients and chlorophyll samples were placed in a freezer immediately upon return from field work. Chlorophyll filters were thawed at a later date and placed into individual vials. 5-mL of methanol was added to each vial as a solvent to dissolve the chlorophyll into solution. Chlorophyll vials was then placed into the freezer again for 12-18 hours before being thawed and analyzed using a Turner Designs Trilology fluorometer (Welschmeyer, 1994).

Nutrient samples were filtered to remove biological activity (0.45-μm polycarbonate filters) and placed on ice .Water samples were analyzed using an O.I. Analytical Flow Solution IV autoanalyzer that combines both segmented flow analysis and flow injection analysis techniques with computer controlled sample selection and peak processing. Chemistries are as specified by the manufacturer, with minor modifications to reflect sample matrix differences. Matrix matching between the carrier, standards and the sample matrix minimizes refractive index effects on absorbance mainly due to salinity. Matrix matching is particularly important for nitrogen chemistries and requires the use of low nutrient seawater (LNSW) to accurately detect low (μM) levels of N in samples. For both orthophosphate and silicate chemistries, artificial seawater is adequate for analysis. The Minimum Detection Limit (MDL) values were as follows: nitrate+nitrate (0.007 μM; O.I. Analytical method 15040908, OIA 2008), silicate (0.071 μM; O.I. Analytical method 15061001, OAI 2001a), ammonium (<0.03 μM; O.I. Analytical method 15031107, OIA 2007) and orthophosphate (0.009 μM; Perstorp Analytical method 000589, OIA 2001b, Alpkem 1993).

Water samples for turbidity were filtered onto 0.7 μm, 47-mm diameter glass fiber filters. The filtered sediment samples were dried and weighed to determine total suspended solids (TSS). The carbon and carbon free dry weights were determined by burning pan with dried filter paper at 550 ˚C in a muffle furnace for 3 hours.

Net Ecosystem Metabolism

Net ecosystem metabolism was calculated at only station (N3) in Carancahua Bay. Only one dissolved oxygen probe was available for the duration of the study, limiting NEM to one bay. The location of this station was chosen out of convenience as the dissolved oxygen probe had to be calibrated bi-weekly. Net ecosystem metabolism was calculated using open-water diurnal methods as first proposed by Odum (1956). Dissolved oxygen (DO) concentrations taken every 15 minutes, between December 2015 and May 2016, were converted to a rate of change in dissolved oxygen concentration. Rates of change were adjusted to control for the diffusion of oxygen between the atmosphere and water column by using percent saturation in DO in the water column, the wind dependent diffusion coefficient K (g O_2/L/Hr), and wind speed data using the equations:

$$R_{dc} = R - ((1 - ((S_1 + S_2)/200)) K/4 \text{ where}$$

R_{dc} (mg O_2/L/15min) = diffusion corrected DO rate of change

R (mg O_2/L/15min) = observed DO rate of change

S_1 and S_2 = DO percent saturation at time one and two respectively

K (g O_2/L/Hr) = diffusion rate at 0% DO saturation

The present study uses D'Avanzo et al. (1996) diffusion coefficient equations to calculate NEM because Lavaca Bay is a shallow-water system similar to Waquoit Bay. D'Avanzo et al.'s (1996) diffusion coefficients allowed for diffusion corrected calculations of dissolved oxygen concentration change that vary over short temporal scales. To calculate daily NEM, the 15 minute wind-diffusion corrected rates of DO were summed over a 24 hour period (starting and ending both at 8:00). Hourly wind and precipitation data was obtained from December 2015 to May 2016 from the National Climatic Data Center (NCDC) for the station located at the Palacios Municipal Airport, TX. Hourly data was used as a proxy for 15 minute difference estimates. This data was obtained from the NCDC website: http://www.ncdc.noaa.gov/cdo-web/datasets/GHCND/stations/GHCND:USW00012935.

Analytics

Hydrographic Data

Data was downloaded from the sondes and ADCP units into individual text files, which was formatted in Microsoft Excel. These files were uploaded into a SAS database. In addition, hydrological flow rate data was obtained from August 2015 to May 2016 from the USGS website: http://nwis.waterdata.usgs.gov.

SAS 9.4 software was used to compile the downloaded datasets, including the sonde and ADCP data. The PROC MEANS procedure was used to calculate the mean, standard deviation, minimum, and maximum values for all the hydrology data.

Water Quality

Mean water quality parameters (salinity, temperature, dissolved oxygen, nutrients, chlorophyll a, and pH) and water depth were calculated for each date and station. Time series were created using PROC SGPLOT to analyze changes in water quality variables over time as compared to salinity per station. PROC GRADAR was used to plot radar charts for each ADCP unit showing the velocity and direction of flow over time. PROC CORR was used to calculate Pearson's Correlations on discharge and water quality variables.

Principal Component Analysis (PCA) was used to classify water quality variables. PCA is a variable reduction technique that reduces a multivariate dataset, and then creates new variables by extracting variance in order of importance. Results of the analysis are a new set of PC variable loads and sample scores. The PC loads represent the underlying structure of the dataset, and the scores represent the contribution of each sample. Results are presented in plots of the vectors of the PC loads to aid interpretation of the underlying structure, and sample scores to visualize spatial and temporal comparisons. A second output is a matrix of sample scores that represents sample contributions. This allows for spatial and temporal comparisons among the different loading variables, stations, and sampling periods (Pollock, 2009; Clarke and Warwick, 2001). PCA was performed using the PROC FACTOR in SAS 9.4.

Water Quality Response to Inflow

PROC REG was used to create a linear regression on flow index (PC1 sample scores) and logged discharge (ac-ft/mo). PROC NLIN procedure was used to plot the predicted regression trend using discrete sonde data and water quality data collected at stations N1-N5 as well as USGS gage inflow into the Lavaca-Colorado Estuary and Guadalupe Estuary. In lieu of available inflow data specific to Carancahua Bay, inflow was substituted using USGS flow data above the Lavaca-Colorado Estuary. Empirically, we can predict the flows needed to provide specific ranges by regressing the data and using an exponential decrease model. In a previous study, an exponential decrease model was used to convert salinity to flow values and was modified in this study to convert flow index values to flow discharge values (Palmer et. al., 2015). The exponential decrease model was as follows:

$$\log(Q+1) = ae^{-bS}$$

Where Q is equal to discharge and S is equal to a created flow index (using PC1 scores). Flow and 90% confidence intervals were estimated for flow index percent changes between 0 - 30% across all bays using SAS 9.4 software. PROC SGPLOT was used to plot the relationship between the flow index and corresponding flow.

Net Ecosystem Metabolism

Salinity recorded monthly was correlated to NEM values. Discharge readings from USGS Gage 0816200, located above Tres-Palacios Bay, were used in lieu of direct inflow readings above Carancahua Bay. Freshwater inflow (discharge) values were summed over a 10 day period prior to sampling, as this was the scale needed to see larger total changes in inflow, and the total volume was labeled as “10 Day Cumulative Discharge.” NEM values were plotted using PROC SGPLOT to create a time series for Carancahua Bay. PROC CORR was used to run Pearson’s Correlations on NEM, cumulative discharge and salinity values. PROC REG was used to create a linear regression between NEM and salinity, as well as NEM and discharge.

RESULTS

Hydrography

Within the Lavaca-Colorado Estuary, freshwater inflow rates for USGS Gage 0816200 ranged from 308.03 ac-ft/mo to 48,741.8000 ac-ft/mo with a mean value of 11,563.45 ac-ft/mo (Figure 2). Discrete salinity measured at station N1 and N2 was similar throughout the study period. N1 had lower salinity values as compared to N2. Both stations experienced decreases in salinity during peaks of inflow throughout November and again throughout June. Within the Guadalupe Estuary, freshwater inflow rates for USGS Gage 08188810 ranged from 30,156.69 ac-ft/mo to 269,871.08 ac-ft/mo with a mean value of 144,095.69 ac-ft/mo (Figure 3). Discrete salinity measured at station N4 and N5 was similar from September to November, then variable throughout the remaining study period. N4 had lower salinity values as compared to N5. Both stations remained at relatively low salinity values as inflow values were consistently high.

Salinity recorded from in-situ sondes ranged from freshwater conditions (S <0.50) to brackish (S >26.00) during the study period from September 2015 to September 2016. Average continuous salinities ranged from S = 3.01 to S = 20.01 among all stations (Table 1), whereas average discrete sample salinities ranged from S = 2.70 to S = 18.43 (Table 2).

Current Flow

All instruments except the N4 station (Guadalupe Bay) passed quality control standards (Figures 4-8). Station N1 flow was predominantly between NNE and SSW, and ranged from 0.13 m/s to 0.25 m/s. (Figure 4). Station N2 flow was predominantly directed North to SSE, and ranged from 0.07 m/s to 0.11 m/s (Figure 5). All stations except Carancahua Bay (N3) had an even distribution of flow in and out of the bay system. Carancahua Bay (N3) flow was directed SSW, primarily exiting the bay and consistent with freshwater inflow (Figure 6). Station N3 flow ranged between 0.06 m/s and 0.14 m/s.

Station N5 experienced a difference in flow between flow closer to the surface and flow closer to the benthos. Station N5 bottom flow (0.70m to 1.40 m from surface) cycled between flow entering and exiting the Hynes Bay Mission Lake complex (West and East flow) (Figure 7). Flow in this direction ranged between 0.06 m/s and 0.12 m/s. However, closer to the surface (1.40 m to 1.90 m from the bottom) the flow was predominantly toward Hynes Bay to the West and flow ranged between 0.09 m/s and 0.20 m/s.

Water Quality

Nitrate and Nitrite had the lowest average at Station N2 (0.63 µmol/L) and highest at Station N4 (79.63 µmol/L) throughout the duration of the study (Figure 8D, Table 3). N1 and N2 both had consistently low concentrations of nitrate and nitrite, as it remained below 25.00 µmol/L for the duration of the study. N4 and N5 had noticeable differences in nitrate and nitrite concentrations. N4 remained over 75.00 µmol/L from January through May. N5 peaked at 50.00 µmol/L in December but remained largely below 50.00 µmol/L for the remainder of the study (Figure 8D). Phosphate had the lowest average at Station N2 (0.91 µmol/L) and highest at Station N4 (4.11 µmol/L) except during wet periods (Figure 8B). N4 consistently had the highest amount of phosphate among all stations from winter through summer. N4 had much higher phosphate concentrations than N5, which remained below 4.00 µmol/L excluding a peak to 5.00 µmol/L in June. N1 and N2 had similar phosphate concentrations throughout the winter, trending below 2.00 µmol/L. N1 phosphate concentrations increased above 2.00 µmol/L in June and remained higher than N2 phosphate concentrations for the remainder of the study. Ammonium had the lowest average at Station N4 (1.10 µmol/L) and highest at Station N5 (3.86 µmol/L) (Table 3). Ammonium concentrations were largely similar between N1 and N2 for the duration of the study, remaining below 4.00 µmol/L except for a peak above 8.00 µmol/L at both stations in June (Figure 8C, Table 3). N4 and N5 had noticeably different ammonium concentrations. N4 remained below 4.00 µmol/L for the entirety of the study. N5 had variable ammonia concentration, and increased above 8.00 µmol/L three separate occasions.

TSS had the lowest monthly average at Station N4 (47.76 mg/L) and the highest monthly average at Station N3 (78.29 mg/L). Overall TSS values declined sharply for all stations throughout October – December excluding Station N3 (Figure 9B, Table 3). TSS values rose for all stations throughout the spring and began to decline in the fall (August-September). Silicate had the lowest average at Station N2 (57.37 µmol/L) and highest at Station N4 (176.51 µmol/L). Silicate concentrations increased during the spring and summer across all stations (Figure 9C, Table 3). N4 and N5 had similar silicate concentrations throughout this increase, which were consistently high, above 150.00 µmol/L. N1 and N2 also had similar silicate concentrations, increasing over the spring and summer to just below 150.00 µmol/L.

Chlorophyll-*a* had the lowest monthly average at Station N2 (6.16 µg/L) and the highest monthly average at Station N3 (18.60 µg/L (Figure 10B, Table 3). N1 and N2 had similar chlorophyll-*a* concentrations throughout the study period (Figure 10B). N4 and N5 had differences in chlorophyll-a concentrations. N4 started at a much higher chlorophyll-*a* concentration than N5 at a peak of nearly 60.00 µg/L before dropping to below 10.00 µg/L in December and remaining lower than 30.00 µg/L for the duration of the study. N5 chlorophyll-a concentrations increased after December and peaked at 50.00 µg/L in February, then above 30.00 µg/L in April. The pH had the lowest monthly average at Station N1 (7.98) and the highest monthly average at Station N3 (8.15) (Table 3). The pH values trended along with salinity, sharply declining in December and June, from above 8.00 to below 6.50, across all stations (Figure 10C). Dissolved Oxygen had the lowest monthly average at Station N2 (7.50 mg/L) and the highest monthly average at Station N5 (8.56 mg/L) (Table 3). Dissolved oxygen was largely similar between N1 and N2, excluding a small increase in N1 dissolved oxygen concentration between April and May. N4 and N5 were also largely similar, excluding December where N4 concentrations were nearly 15.00 mg/L and in February where N5 concentrations were nearly 20.00 mg/L (Figure 10D).

Relationships among Water Quality Variables

The first two principal components (PC1 and PC2) explained 30% and 20% respectively for a total of 50% of the variation in hydrographic variables. PC1 variable loads for the hydrographic data had the highest positive values for nutrients (silicate and phosphate) and the highest negative values for salinity (Figure 11). The PC1 axis represents an inflow index, where a decrease in salinity (or increase in freshwater inflow) is associated with increased nutrient concentrations. PC1 also is correlated to seasonal effects: dissolved oxygen (DO) and temperature show an inverse relationship. PC2 loads for the hydrographic data had the highest positive values for particulate organic matter (POM), total suspended solids (TSS) and chlorophyll-a (Chl-a). The PC2 axis has variables that are correlated to particles in the water column. The PC2 axis had no negative values.

Salinity ranges were used as labels for PC1 and PC2 sample scores (Figure 12). Ranges varied from S = 0-5, S = 5-10, S = 10-15, S = 15-20, and S = 20-25. Low salinity ranges (0-5, 5-10) were associated with positive values on the PC1 axis, where high salinity ranges (15-2, 20-25) were associated with negative values on the PC1 axis (Figure 10). Low salinity ranges are more closely associated with higher concentrations of nutrients. Conversely, high salinity ranges are more closely associated with lesser concentrations of nutrients. There was no difference in salinity ranges based on the PC1 axis because the samples were spread along the entire range.

Station sample scores were distributed into spatial patterns along the inflow gradient (Figure 13). Stations N1 and N2 generally exhibited a more positive relationship with salinity.

Lowest salinity values occurred at stations N1 and N2. N2 was largely a fresher station. Stations N1 and N3 were scattered across the plot. Stations N4 and N5 had the strongest positive relationship with nutrient amounts, and a negative relationship with salinity. Station N5 experienced the highest amounts of particulate organic matter and total suspended solids.

Stations scores were also analyzed by periods, where the first period began September 2015 and the last period ended in September 2016 (Figure 14). Stations were spread widely from September through December 2015. During January and February 2016, stations shifted to lower salinity ranges. During June, stations shifted to higher salinity ranges and lower nutrient concentrations. Stations tended towards a positive relationship with nutrients from June through September 2016.

PC1 sample scores were plotted against the sampling date, creating principal response curves for each station (Figure 15). Stations N1 and N2 had similar PC1 scores throughout the study. Both stations trended downwards in February (from -0.50 to -2.00), and peaked in-between May and July (at and above -0.50). Station N3 was not similar to any of the other stations. Station N3 PC1 scores rose above 0.50 in December, then dropped below -1.00 in February, then rose above 1.00 in June before lowering under 1.0 again. Stations N4 and N5 were largely dissimilar. Station N4 PC1 scores rose to 1.50 in December, then fluctuated back and forth between being above and below 1.00 for the remainder of the study. Station N5 PC1 scores dropped from 0.50 in December to -0.50 in February, then rose above 1.50 in July before dropping again below 0.50.

Water Quality Response to Discharge

Pearson correlations for water quality variables and discharge showed some significant relationships that were demonstrated in the PCA (Table 4). Salinity exhibited a significant relationship with discharge ($p < 0.0001$), as did phosphate ($p = 0.0002$), silicate ($p < 0.0001$), and nitrate and nitrite ($p = 0.0002$). Other water quality variables (dissolved oxygen, pH, total suspended solids, particulate organic matter, ammonium, and chlorophyll-a) did not exhibit a significant relationship with discharge ($p > 0.05$).

Flow Index (PC1 sample scores) was plotted against discharge (ac-ft/mo) across stations N1-N5 from September 2015 to September 2016 (Figure 16). As discharge increased, flow index scores increased as well. Stations N1 and N2 both tended to have lower amounts of discharge as compared to other stations. Both of these stations received typically less than 10,000 ac-ft/mo (though N2 reached up to 15,000 ac-ft/mo) and their flow index values ranged from slightly below -2.00 to slightly above 0.00. Station N3 was spread across the graph, experiencing low discharge values (below 1000 ac-ft/mo) and high discharge values (over 15,000 ac-ft/mo). Flow index scores for N3 were lower (below 0.00) with low levels of discharge (0-1000 ac-ft/mo) and highest (above 1.00) at higher levels of discharge (between 10,000 and 15,000 ac-ft/mo). Stations N4 and N5 both had the highest amount of discharge during the study, ranging from above 15,000 ac-ft/mo to over 100,000 ac-ft/mo. Flow index values for station N4 ranged between 0.50 and 1.50, while flow index values for station N5 ranged from under -0.50 to above 1.50.

A linear regression was created for flow index (PC1 sample scores) and logged discharge (ac-ft/mo) (Figure 17). A significant correlation ($p < 0.0001$, $R^2 = 0.44$) was found between the flow index (PC1 scores) and discharge (log ac-ft/mo). As discharge increases, the flow index values increase as well.

Flow Required to Maintain Water Quality

Predicted discharge using the flow index was created using an exponential decrease model at Tres-Palacios Bay, Carancahua Bay, and San Antonio Bay (Figures 18-20). The equations modeled in these graphs were used to create percent change estimate in flow index values and corresponding estimates for flow discharge (ac-ft/mo) (Figure 21, Table 7). San Antonio Bay received a much higher level of flow discharge throughout the course of the study, with a mean flow of 144,095.69 ac-ft/mo (Figure 3). Due to the high level of flow that moved through this bay, a much larger amount of corresponding predicted flow, anywhere between nearly 6,000 up to 35,000 ac-ft/mo, was required to increase the percent change of the flow index created for the bays. Carancahua Bay and Tres-Palacios Bay received similar amounts of discharge, with a mean of 11,563.45 ac-ft/mo (Figure 2), which were much less in volume than San Antonio Bay. Both Carancahua and Tres-Palacios Bay required a lesser amount of corresponding predicted flow, under 6,000 ac-ft/mo in Tres-Palacios and under 3,000 ac-ft/mo in Carancahua, to see between 0-30% change in the flow index.

Net Ecosystem Metabolism

Salinity and net ecosystem metabolism were measured at only station N3 in Carancahua Bay from December 2015 through March 2016 (Table 5). Salinity recorded from discrete sampling ranged from S = 1.09 to S = 15.91, with a mean of S = 10.60. Net ecosystem metabolism ranged from -2.56 (mg O_2/L/day) to 8.06 (mg O_2/L/day) with a mean of 1.00 (mg O_2/L/day). Freshwater inflow was summed over a 10 day sampling period prior to sampling using discharge values from Gage 0816200 above Tres-Palacios Bay in order to see larger fluctuations in inflow. This cumulative flow ranged from 176.93 ac-ft to 1114.71 ac-ft.

Net ecosystem metabolism varied throughout the winter (Figure 22A). NEM values remained between -2.50 and 2.50 from December to January (Figure 22A). NEM then rose to a peak above 5.00 O_2/L/day before dropping down below 0.00 O_2/L/day again. The strongest difference in NEM occurred from a low peak below 0.00 mg O_2/L/day in mid-January to a high peak at 8.06 mg O_2/L/day. NEM values then dropped over time for the rest of the study period, excluding a peak above 5.00 mg O_2/L/day in late-February. Salinity values ranged between S = 5.00 and S = 10.00 for most of December and January (Figure 22B). A large drop in salinity, nearly to S = 1.00, occurred in the middle of January before rising to around S = 15.00. Salinity ranged between S = 10.00 and S = 15.00 for the rest of the study period, excluding a drop to S = 5.00 in late-February. Cumulative discharge peaked three times during the study: once to 800.00 ac-ft in December, another over 1000.00 ac-ft in mid-January, and another above 800.00 ac-ft between late-February and March (Figure 22C).

Net ecosystem metabolism averaged at 1.00 mg O_2/L/day indicating an autotrophic system. Net ecosystem metabolism increased from slightly heterotrophic during the lowest amounts of freshwater discharge (below 500.00 ac-ft) to becoming more autotrophic as discharge increased (Figure 22). Two of the highest peaks in NEM values (8.06 in January and above 7.50 in March) correspond to rising cumulative discharge values that occur prior. Excluding the three peaks in cumulative discharge noted throughout the study, discharge values were relatively low.

Net Ecosystem Metabolism Response to Discharge

Pearson correlations for net ecosystem metabolism, salinity, and discharge indicated significant relationships (Table 6). Salinity did not exhibit a significant relationship with net ecosystem metabolism ($p > 0.05$) until salinity was lagged by one day ($p = 0.0012$). 10 day cumulative discharge did not exhibit a significant relationship ($p > 0.05$) with net ecosystem metabolism until discharge was also lagged by one day ($p = 0.0146$). Both salinity and cumulative discharge had weak but significant relationships with net ecosystem metabolism. Lagged salinity and lagged discharge further than a 1 day period did not have significant relationships with net ecosystem metabolism.

A linear regression was created for net ecosystem metabolism and 1 day lagged salinity (Figure 23). A significant but weak correlation ($p = 0.0012$, $R^2 = 0.11$) was found between salinity and NEM. An inverse relationship exists between salinity and NEM: As salinity decreases, net ecosystem metabolism tends to increase.

A linear regression was created for net ecosystem metabolism and 1 day lagged cumulative discharge (Figure 24). A significant but weak correlation ($p = 0.0146$, $R^2 = 0.07$) was found between 1 day lagged cumulative discharge and NEM. As discharge increases, net ecosystem metabolism tends to increase as well. Salinity and freshwater inflow have a significant (though weak) inverse relationship ($p = 0.0010$, $R^2 = .12$) (Figure 25).

DISCUSSION

Freshwater Inflow and Water Quality

Historical studies have analyzed the overall importance of freshwater inflow to estuarine systems and have conclusively assessed that inflow is a major driving factor to both estuary functioning and ecosystem health (Alber, 2002; Palmer et. al., 2011; Palmer et. al., 2015; Pollack et. al., 2009). Inflows serve several important roles within an estuary, including the creation and preservation of low-salinity nurseries, sediment and nutrient transport, allochthonous organic matter inputs, and the timing and extent of migration of critical estuarine species (Palmer et. al., 2011). Alber (2002) developed a conceptual framework that is now considered the basis for a freshwater inflow determination methodology. This framework suggests that inflow drives estuarine condition and in turn estuarine condition drives biological response (Alber, 2002). Many different biological responses within estuaries are affected by water quality, thus there exists a need to define the relationship between inflow, water quality, and estuary function.

As demand for freshwater continues to grow, more effort is being focused on developing technology and management geared towards capturing and diverting flows from rivers and streams. The hydrology of most bodies of freshwater in the U.S. have experienced withdrawals, diversions, dams and other alterations of some kind (Naiman et. al., 1995; Montagna et. al., 2013). Due to these increasing alterations, many estuaries and large freshwater bodies of water have experienced severely limited inflow leading to a drastic change in the functioning of these ecosystems (Montagna et. al., 2002; Montagna et. al., 2013). In recent years, legislative changes about the potential effects of altered inflow regimes on downstream bays and estuaries has caused resource managers and policy makers to put effort towards hydrological restoration. One way to restore environmental flow is to obtain water rights in certain areas to enhance freshwater flow in order to enhance or restore the ecological benefits provided by freshwater inflows. This has led to the question this study attempted to answer: Will small changes of freshwater inflow make a difference in the condition and functioning of an estuary? The hypothesis assumed in this study was that small changes to inflow will change water quality and ecosystem functioning within the estuary, most notably in areas that are closer to the river mouth or source of freshwater inflow. This study assessed how water quality variables and net ecosystem metabolism change with varying freshwater inflows, and analyzed the impact of small diversions of flow from estuarine systems.

Estuaries display gradients of both salinity and nutrients (Montagna et. al., 2013) that are determined by the timing, quantity, and quality of freshwater inflow. Previous study has proven that salinity is generally an inverse-indicator of nutrients, where low salinity ranges correspond to high freshwater inflow and are linked to an increase in nutrient concentrations (Pollack et. al., 2009; Montagna et. al., 2013; Longphuirt et. al., 2016). The relationship between freshwater inflow, salinity, and water quality variables was important to determine in each of the bays presented in this study.

In order to assess the relationship between freshwater inflow and water quality variables, discharge values from two different USGS gage stations were compared with water quality variables. Multivariate analytical techniques have been used previously to assess long term trends in hydrographic changes and water quality parameters (Pollack et. al., 2009). This study used a principal component analysis (PCA) to assess which water quality variables are most influenced by freshwater inflow, and to predict how water condition may change in response to alterations to hydrology. The results of the PCA indicated that PC1 represents a freshwater inflow effect within an estuary (Figure 11). PC1 sample scores were used to create a flow index representative of freshwater inflow's effect on water condition. A positive PC1 score correlated with estuaries with high nutrient concentrations and low salinities. A negative PC1 score, on the other hand, correlated with low nutrient concentrations and high salinities (Figure 11). Pearson correlation confirmed the results found through the PCA, and indicated that salinity, silicate, nitrate and nitrite, and phosphate were significantly correlated to discharge (Table 4). Other studies have created ecosystem health indices that function in the same manner as the flow index created in the present study (Christensen et. al., 1997; Adams et. al., 2002; Pollack et. al., 2009; Flint et. al., 2016). Pollack (2009) created a Freshwater Inflow Biotic Index (FIBI) for the Lavaca-Colorado Estuary, Texas that defined a relationship between water quality variables and inflow in Lavaca Bay, Texas, similar to those found within the present study. In Queensland, Australia, an Ecosystem Health Index (EHI) was created that combined water quality monitoring data with ecological response to produce an overall "ecosystem health score" (Flint et. al., 2016). The approach within the present study was similar, and intended to use flow's effect on water quality variables to create a flow index that can be used for management purposes.

Spatial variation for salinity and nutrient gradients also exists between primary and secondary bays (Sharp et. al., 1986; Montagna et. al., 2002; Palmer et. al., 2011). Secondary bays receive more direct inflow, and therefore it was assumed that secondary bays within the present study would be more responsive to variation in flow. PCA stations scores for hydrographic characteristics, using PC1 and PC2, indicated spatial patterns along the inflow gradient (Figure 13). Within Tres-Palacios Bay, N1 was the upper station closer to the river mouth and had lower salinity values than N2 (Table 3). N1 tended to have higher concentrations of nutrients (nitrate and nitrite, phosphate, ammonium, and silicate) as well as higher total suspended solid particulate amounts and chlorophyll-*a* concentrations. Stations N1 and N2 were closely related in their response to PC1's inflow effect. N1 is located closer to the source of freshwater inflow that feeds into the Tres-Palacios estuary, and it may have been more effected by freshwater inflow due to proximity. Within San Antonio Bay, N4 was the upper station closer to the river mouth and had lower salinity values than N5 (Table 3). N4 tended to have higher concentrations of nutrients (nitrate and nitrite, phosphate, and silicate) though N5, the lower station, had higher concentrations of ammonium, as well as total suspended solid particulate amounts. Both N4 and N5 had similar chlorophyll-*a* concentrations. As seen in Tres-Palacios Bay, N4 was located closer to freshwater inflow input and may have been more effected by inflow due to proximity. N3, located in Carancahua Bay, varied greatly between negative and positive correlations with the flow index (Figure 13).

Previous studies of estuarine systems has determined that freshwater inflow controls numerous ecological processes and that alterations to the natural flow regime can impact the quality of water by creating changes in nutrient concentrations (Grange et. al., 2000; Nagy et. al., 2002) leading to either eutrophic or oligotrophic conditions (Boesch et. al., 2001), and by changing the salinity gradient of an estuary (Ustach et. al., 1986; Palmer et. al., 2002) which may in turn effect the biota that live along it (Jassby et. al., 1995; Poff et. al., 1997; Pollack et. al., 2011). This has led to the idea that there would be value in protecting, enhancing, or restoring flows to estuaries.

Freshwater Inflow and Net Ecosystem Metabolism

Odum (1956) introduced the concept of net ecosystem metabolism as an informative index that was representative of both function and structure within an ecosystem. Net ecosystem metabolism serves as an equilibrium between production and consumption. Previous study has used net ecosystem metabolism to evaluate ecosystem services or budgets (Dodds and Cole, 2007) and, more specifically, to evaluate the use of NEM as an indicator of estuarine impairment (Russell and Montagna, 2007) and freshwater inflow effects (Russell et. al., 2006). This study examined the relationship between NEM, salinity, and discharge at a singular station (N3) in the upper reaches of Carancahua Bay. This location was chosen due to the area being easily accessible, as the dissolved oxygen probe needed to be recalibrated bi-weekly for accuracy, and also because it was within the upper portion of Carancahua Bay and therefore close to the source of freshwater inflow.

A positive NEM indicates autotrophic ecosystems that may rely on internal nutrient sources, while a negative NEM indicates heterotrophic ecosystems where external organic matter is dominant. Increased loading of nutrients can lead to increased NEM via stimulation of production over respiration (Caffrey, 2004). An estuary ecosystem that is dominated by allochthonous inputs tends to be heterotrophic, while those that rely less on allochthonous carbon inputs are autotrophic (Caffrey, 2004).

NEM has been observed in previous study to change from heterotrophy to autotrophy along the estuarine gradient (Howarth et. al., 1996) due to changes in salinity, and also along depth gradients (Caffrey et. al., 1998). Littoral zones and other shallow-water areas are often autotrophic while deep domains are net heterotrophic. Deeper channels are typically more light-limited which may account for their more heterotrophic state. Total estuarine area may determine the ratio of allochthonous organic inputs to autochthonous production (Caffrey, 2004).

Net autotrophy implies that Carancahua Bay acts as a carbon sink, either burying or exporting organic matter (Russell, 2006). Carancahua Bay is not only shallow (with depth ranging between 1-2 meters) but is also slender (roughly 1.6093 km in width). Routine monitoring of water quality by the state occurs in the upper portion of the bay which is most characteristic of the brackish water biotope controlled by inflow from Carancahua Creek (Jensen and Bowman, 1985). Monitoring efforts (Tremblay and Calnan, 2010) have revealed that Carancahua Bay is a part of Matagorda Bay that has known seagrass habitat. Previous study has suggested that sites adjacent to submerged aquatic vegetation (such as seagrass and macroalgal beds) tend to be either autotrophic or near balance (Caffrey, 2004). The location of this station in the upper reaches of Carancahua, combined with the Bay's shallow nature and its adjacent placement to potential submerged vegetation may explain the overall net autotrophy seen in the present study.

A few peaks in NEM (where it became extremely autotrophic) were correlated to drops in salinity, which were indicative of increases in freshwater inflow (Figure 22). A linear regression between NEM and lagged salinity showed a weak but significant relationship (Figure 23, Table 6). This inverse relationship indicated that as salinity decreases, NEM increases. Salinity is a function of many processes and integrates freshwater inflow, as well as evaporation and tidal exchange (Russell et. al., 2006). Due to this, the relationship between freshwater inflow and NEM was expected to be similar to the relationship between salinity and NEM. While a significant correlation between lagged flow and NEM was found, it was weak (Figure 24, Table 6). In Carancahua Bay: as freshwater inflow increases, salinity decreases, and net ecosystem metabolism tends to increase.

Time lags are defined as the period of time between two closely related events such as the time between stimulus and response. Time-lags are often used to measure response in estuaries and to inflow events (Kalke and Montagna 1991, Palmer et al. 2002, Pollack et al. 2011). Though direct correlations between flows (10 day cumulative discharge), salinity, and net ecosystem metabolism response at the time of the flows were not established, net ecosystem metabolism had strong correlations with time-lagged values of salinity and flow. A significant time-lagged response occurred for salinity and flow 1 day after the initial recorded date (Table 6). Lagged salinity and lagged discharge further than a 1 day period did not have significant relationships with net ecosystem metabolism. In this study, changes to net ecosystem metabolism were not seen immediately within the environment and instead took time to respond to the effects of the freshwater inflow.

A possible explanation for the lack of a strong correlation between NEM, salinity, and inflow in this study is that the discharge values came from the USGS gage that was above Tres-Palacios Bay, and therefore it is not a direct measurement of the discharge occurring in Carancahua Bay. More direct discharge data may be required to better assess the relationship between NEM and freshwater inflow in Carancahua Bay. It is also possible that freshwater inflow and NEM may not have a linear relationship, and the true relationship between NEM and FWI may be more complex than was assumed in this study. Most NEM data was clustered at relatively low flows, lacking in NEM data that corresponded to the higher freshwater inflow rates experienced in the bay. This makes it difficult to assess the shape of the relationship curve between these two variables. Another possible explanation is that the Carancahua Bay watershed is small and biogeochemical loading is small.

Other environmental factors may become more influential than freshwater inflow, especially during low base-flow periods (Russell et. al., 2006). The shallow depth of this specific estuary may also contribute to the overall variability of NEM. Inflow was also low for this specific estuary. Differences in land use and land cover may effect NEM as runoff becomes a more important contributing factor of inflow. Other factors which can led to variability in NEM include daily irradiance (D' Avanzo et. al., 1996), adjacent habitat and hydraulic residence time (Caffrey, 2004), microphytobenthic photosynthesis rates (Blanchard and Montagna, 1992), nutrient and organic loading (Smith and Hollibaugh, 1997), and physical factors that have effects on photosynthesis such as temperature and turbidity.

Previous study has shown that the use of NEM as an indicator of freshwater inflow effects within estuary systems may be constrained spatially (by the proximity of estuarine areas to areas of freshwater inflow) and also temporally, from periods of moderate-high flows (Russell et. al., 2006). The current study focused on one sample station (N3) and assumed that a single station measurement of NEM would be representative of the entire estuarine system, as has also been assumed in older studies (D' Avanzo et. al., 1996). However, more recent study indicated that NEM can be variable between upper and lower bays, and that a single NEM measurement is not enough to be representative of the whole estuary (Russell et. al., 2006). This is due to the fact that estuaries have unique signatures: the set of unique geologic, geographic, hydraulic, and climatic conditions that compose an estuarine system (Montagna et. al., 2013). Future study and monitoring programs that incorporate NEM should include multiple sampling locations along the inflow gradient throughout the whole of an estuary.

Management Implications

For years, coastal wetlands and estuarine habitats have been degraded due to diversions and alterations to freshwater inflow (Montagna et. al., 2013). Texas State Legislature passed the Texas Water Planning Act in 1957, and the creation of a Water Plan was adopted in 1969 that called for 2.5 million acre-feet annual for Texas bays and estuaries. In 1975 Texas enacted Senate Bill 137, which called for comprehensive study of the effects of freshwater inflow for bays and estuaries located within the State. Through these studies, several estuarine needs were directly associated with freshwater inflow and associated water quality components. In 1985 Texas enacted House Bill 2. The legislation directed the Texas Natural Resource Parks and Wildlife Department (TNRCC) to consider effects on bays and estuaries and to consider "conditions necessary to maintain beneficial inflows to any affected bay or estuary system"

(Longley, 1994). The legislation further clarified that the term "beneficial inflows" specifically meant "a salinity, nutrient, and sediment loading regime adequate to maintain an ecologically sound environment […] that is necessary for the maintenance of productivity of economically important and ecologically characteristic sport or commercial fish and shellfish species and estuarine life upon which such fish and shellfish are dependent" (Longley, 1994). House Bill 2 was driven to focus on species management. More recent Texas policy, as of Senate Bill 3 enacted in 2007, requires the State to maintain environmental inflow levels to the bays and estuaries that "promotes and protects a sound ecological environment" (Texas Legislation, 2007). Senate Bill 3 set out a new regulatory approach to protect flows through the use of environmental flow standards developed through Texas Commission on Environmental Quality (TCEQ). Senate Bill 3 directed the use of an environmental flow regime in developing flow standards and defined an environmental flow regime as "a schedule of flow quantities that reflects seasonal and yearly fluctuations that typically would vary geographically, by specific location in a watershed, and that are shown to be adequate to support a sound ecological environment and to maintain the productivity, extent, and persistence of key aquatic habitats" (Brandes et. al., 2009). Senate Bill 3, unlike House Bill 2, brought ecosystem management to the forefront of policy. From these laws, more funding has been made available for the creation and implementation of monitoring data and freshwater inflow needs of estuaries and environmental flow regimes.

Research and management efforts have previously focused on determining and implementing appropriate ranges and frequency of freshwater inflows that will sustain a functional estuary. Due to Senate Bill 3, there has been an incentive to develop environmental flow regime standards for Texas estuaries. On July 6, 2012, the Resources and Ecosystems Sustainability, Tourist Opportunities, and Revived Economics of the Gulf Coast States Act (RESTORE Act) was signed into law (112th Congress, 2011). This Act allowed for the creation of a Gulf Coast Restoration Trust Fund under the Federal Water Pollution Control Act. Under this Act, funds were made available for projects and programs that could restore and protect the environment in the Gulf Coast region. The available funding has led to an increase in hydrological restoration programs within Texas, and helped to fund The Texas Environmental Flows Initiative's protection and enhancement plan for bays and estuaries along the Texas Gulf coast.

The Texas Environmental Flows Initiative seeks to take advantage of the RESTORE act and proposed the creation of an estuarine protection and enhancement plan for the bays located within the Texas Gulf coast. This plan consists of either obtaining water rights in certain areas to enhance freshwater flow in order to enhance or restore the ecological benefits of returning freshwater into these, or to purchase a water right so that freshwater inflow won't be diverted. Because the amount of freshwater rights available may be low, it is possible that only upper reaches of the estuary may be affected by small changes to inflow. It is necessary to understand exactly how small changes in turn will impact the bays.

In order to assess whether or not small changes in freshwater inflow quantity would alter water quality, the created flow index was plotted against discharge (ac-ft/mo) across stations (Figure 16) and analyzed in a linear regression (Figure 17). This analysis indicated that flow index values increase with increasing amounts of discharge. This relationship was used to then predict discharge values by the flow index for each bay system (Figures 18-20). This served as a useful tool for analyzing how much freshwater inflow in ac-ft/mo. is necessary to cause a given percent change in water quality variables. San Antonio Bay required a larger amount of freshwater inflow change in order to see a response in the flow index, as it normally receives much more inflow than the other two bays (Figure 21). Small diversions to San Antonio Bay, such as those less than 10,000 ac-ft/mo., were estimated to change the flow index by less than 10%. Conversely, small diversions less than 10,000 ac-ft/mo. were estimated to change the flow index in both Carancahua Bay and Tres-Palacios Bay by more than 30%. Development of the percent-of-flow-index approach shown in this study emphasizes the interaction of freshwater inflow with the overlap of water quality conditions in estuarine systems and provides a framework for analyzing how specific amounts of flow diverted may in turn alter water quality condition in specific bays.

The ecohydrological approach used within this study links freshwater inflow to a changing flow index, which represents water quality variables. This is useful for determining the effects of changing freshwater inflow on estuarine water quality. Managers and policy makers may use this concept to determine the amount of flows which may be acceptable for diversions, taking water quality response into consideration. It is a generic approach that may be useful elsewhere when applied to other estuarine systems.

CONCLUSION

The primary purpose of this study was to examine effects of variation in freshwater inflow to make inference about the relationships between water quality variables, estuary function, and freshwater inflow. This study attempted to discover if small changes, such as 10,000 ac-ft/mo., to inflow can have impacts to the water quality and function of estuary systems, and it did.

It is evident that freshwater inflow is a primary driving factor in the estuaries examined in this study, as it is in many other estuarine environments. Freshwater inflow plays an important role in defining the water quality of estuaries. Freshwater inflow is responsible for driving nutrient concentrations and salinity ranges, as demonstrated by the PCA performed in this study. Small diversions can have significant impacts on water quality response, especially within bays that already receive low-flow amounts. Predictions on the bays within the current study show that San Antonio Bay requires a larger amount of freshwater inflow change (above 10,000 ac-ft/mo.) to see corresponding changes in water quality response, as it is a bay that typically receives large amounts of inflow. Conversely, predicted inflow in Carancahua Bay and Tres-Palacios Bay both indicate that small amounts of freshwater inflow (less than 10,000 ac-ft/mo.) have larger impacts on water quality response as these bays receive lesser amounts of inflow. This study created a flow-to-water-quality index that emphasized the interaction of freshwater inflow with the overlap of water quality variables. This concept provides a generic framework that can be used by managers and policy-makers to analyze how specific amounts of flow diverted may alter water quality condition in specific bays.

Freshwater inflow can also alter the net ecosystem metabolism of an estuary, though other environmental factors may play a more important role in this change. The current study focused on one sample station (N3) and assumed that a single station measurement of NEM would be representative of the entire estuarine system. Freshwater inflow and salinity both had a significant but weak relationship to NEM when lagged, due to a time lag experienced between drivers and estuary response. It is recommended that future studies attempting to define the same relationship between freshwater inflow and NEM use several stations along the freshwater gradient within the whole of an estuary.

As the human population increases, so does the need for freshwater. There has been rising concern, both scientific and legislative, over anthropogenic changes to the environment. Alterations, such as diversions and withdrawals, change natural freshwater inflow regimes to coastal habitats. Diversions to freshwater inflow alter the amount of nutrients and sediments being brought into an estuary, and alter the salinity gradient found therein. Historical studies and the present study both conclude that these alterations to flow can have drastic effects on estuary water quality and estuary function. Areas which receive low-flow amounts are more drastically affected by small changes in inflow amounts. As more funding becomes available for hydrological restoration, management planning should consider the amount of flows determined to be withdrawn from specific estuary systems and in turn how that will affect overall estuary health. The use of the FWI index created in the current study may be applied by managers and policy makers to determine the amount of flows that may be needed for diversions to maintain or restore water quality conditions in estuaries.

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Table 1. Summary statistics of 15 minute interval in-situ sonde data for salinity (S) at Stations N1-N5 from September 2015 to May 2016.

Bay	**Station**	**Salinity (S)**			
		Mean	**Std Dev**	**Min.**	**Max.**
Tres-Palacios Bay	N1	17.46	4.33	0.07	24.35
	N2	20.01	3.34	1.08	26.13
Carancahua Bay	N3	7.35	4.48	0.00	17.51
San Antonio Bay	N4	3.01	2.99	0.01	16.63
	N5	6.57	2.94	0.00	18.75

Table 2. Summary statistics of monthly sampled discrete sonde data for salinity (S) at Stations N1-N5 from September 2015 to September 2016

Bay	**Station**	**Salinity (S)**			
		Mean	**Std Dev**	**Min.**	**Max.**
Tres-Palacios Bay	N1	15.03	5.93	3.41	22.51
	N2	18.43	5.19	8.18	24.72
Carancahua Bay	N3	5.60	4.87	0.30	14.53
San Antonio Bay	N4	2.70	3.33	0.18	11.06
	N5	7.04	4.64	0.19	15.81

Table 3. Overall mean water quality values for discrete samples at each station. Standard deviation for all samples at each station are in parenthesis.

Variables	Station					Mean
	N1	**N2**	**N3**	**N4**	**N5**	
Salinity (S)	15.03(5.93)	18.43(5.19)	5.600(4.87)	2.700(3.33)	7.040(4.64)	9.76
Temperature (°C)	23.38(5.050)	23.11(5.680)	23.35(5.290)	23.68(4.740)	23.65(4.860)	23.43
Nitrate + Nitrite (µmol/L)	3.52(6.39)	0.63(0.48)	20.72(41.81)	79.63(48.63)	19.88(18.00)	24.88
Silicate (µmol/L)	63.85(56.13)	57.37(47.00)	140.51(86.07)	176.51(73.23)	152.93(68.92)	118.23
Phosphate (µmol/L)	1.31(0.99)	0.91(0.61)	2.93(2.55)	4.11(1.93)	2.00(1.42)	2.25
Ammonium (µmol/L)	1.79(3.20)	1.23(2.15)	1.41(1.63)	1.10(1.23)	3.86(4.05)	1.88
Chlorophyll-*a* (µg/L)	10.51(5.35)	6.16(4.21)	18.60(15.80)	16.66(14.76)	16.59(14.56)	13.70
pH	7.98(0.58)	8.08(0.54)	8.15(0.61)	8.14(0.56)	8.08(0.53)	8.09
Dissolved Oxygen (mg/L)	7.95(1.96)	7.50(1.33)	8.47(2.22)	8.47(2.36)	8.56(3.58)	8.19
TSS (mg/L)	71.52(33.27)	51.13(31.39)	78.29(24.77)	47.57(27.69)	67.22(48.56)	63.15

Table 4. Pearson correlation coefficients and p values between discharge and water quality variables.

Pearson Correlation Coefficients	
Prob > |r| under H_o: r=0	
Variable	**Discharge(ac-ft/mo)**
Salinity (S)	-0.5821 **<0.0001**
Dissolved Oxygen (mg/L)	0.0982 0.4364
pH	-0.0536 0.6716
Total Suspended Solids (mg/L)	-0.0782 0.5361
Particulate Organic Matter (mg/L)	-0.0347 0.7840
Phosphate (µmol/L)	0.4410 **0.0002**
Silicate (µmol/L)	0.4969 **<0.0001**
Nitrate + Nitrite (µmol/L)	0.4475 **0.0002**
Ammonium (µmol/L)	0.1064 0.3989
Chlorophyll-*a* (µg/L)	0.0420 0.7397

Table 5. Salinity, net ecosystem metabolism, and 10 day cumulative discharge means from December 2015 to March 2016.

Variable	Mean	Std Dev	Min.	Max.
Salinity (S)	10.60	3.03	1.09	15.90
Net Ecosystem Metabolism (mg O_2/L/Day)	1.00	2.01	-2.56	8.05
10 Day Cumulative Discharge (ac-ft)	483.74	293.14	176.93	1114.71

Table 6. Pearson correlation coefficients and p values between net ecosystem metabolism and inflow indicators of salinity and 10 day cumulative discharge.

Pearson Correlation Coefficients	
Prob > |r| under H_o: r=0	
Inflow Indicators	**NEM**
Salinity	0.0421 0.6887
1 Day Lagged Salinity	-0.3327 **0.0012**
10 Day Cumulative Discharge	-0.1269 0.2230
1 Day Lagged Cumulative Discharge	0.2552 **0.0146**

Table 7. Percent change estimate in flow index (PC1) values and corresponding estimates for flow (ac-ft/mo).

	Corresponding flow (ac-ft/mo)	
	Tres-Palacios Bay	
Percent Change	Estimate	90% CI
0%	-1	(-1, 0)
5%	954	(554, 1355)
10%	1909	(1109, 2710)
15%	2864	(1664, 4064)
20%	3819	(2220, 5419)
25%	4774	(2775, 6774)
30%	5729	(3330, 8128)
	Carancahua Bay	
Percent Change	Estimate	90% CI
0%	-1	(2, 0)
5%	448	(-83, 981)
10%	896	(-168, 1963)
15%	1344	(-253, 2944)
20%	1792	(-338, 3926)
25%	2241	(-423, 4907)
30%	2689	(-508, 5889)
	San Antonio Bay	
Percent Change	Estimate	90% CI
0%	0	(0, 0)
5%	5894	(4379, 7409)
10%	11788	(8759, 14818)
15%	17683	(13138, 22227)
20%	23577	(17518, 29636)
25%	29471	(21898, 37045)
30%	35366	(26277, 44454)

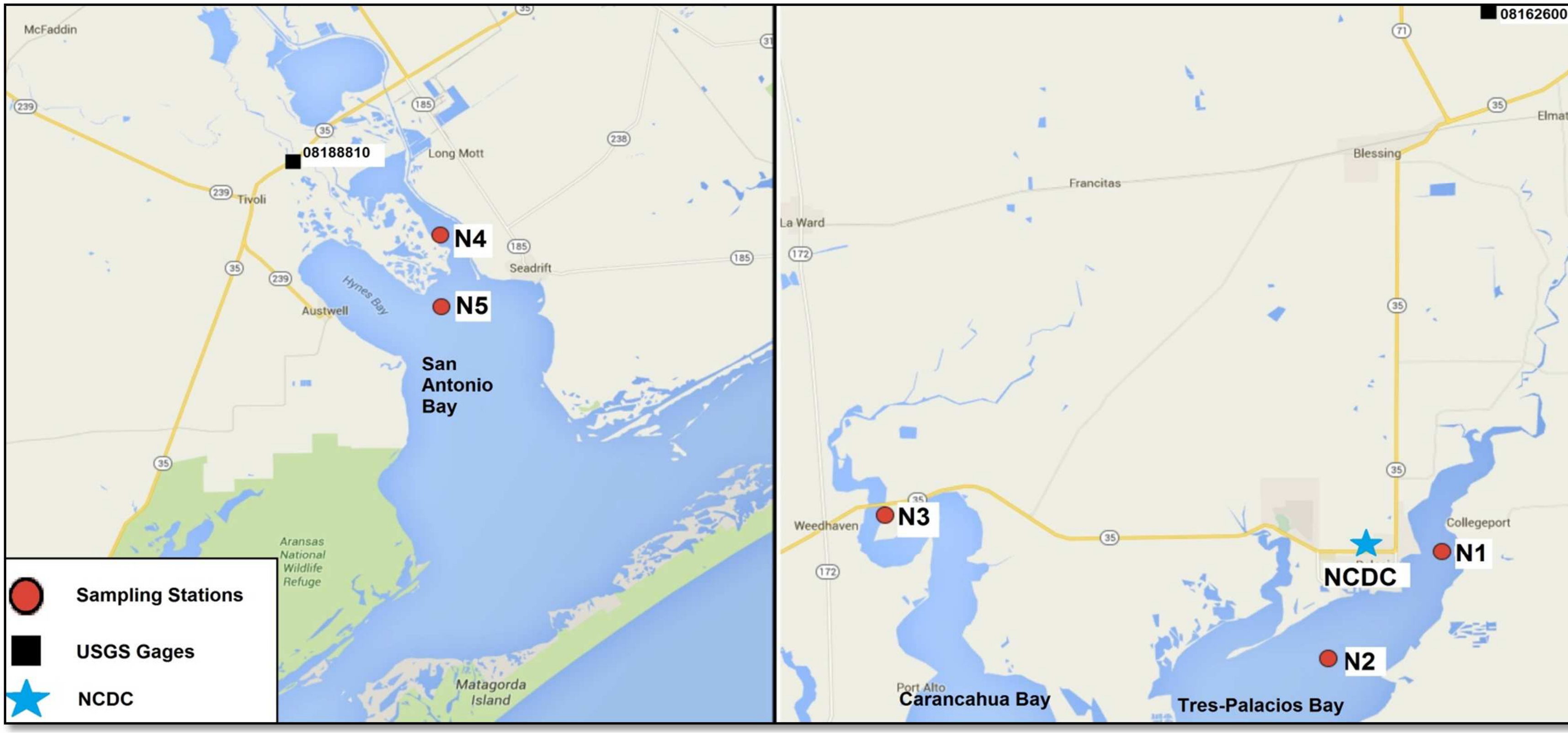

Figure 1. Map of sampling locations. Stations N1 and N2 located within Tres-Palacios Bay, station N3 located within Carancahua Bay, and stations N4 and N5 located within San Antonio Bay. National Climatic Data Center (NCDC) located between Carancahua and Tres-Palacios Bay. Two USGS Gage locations.

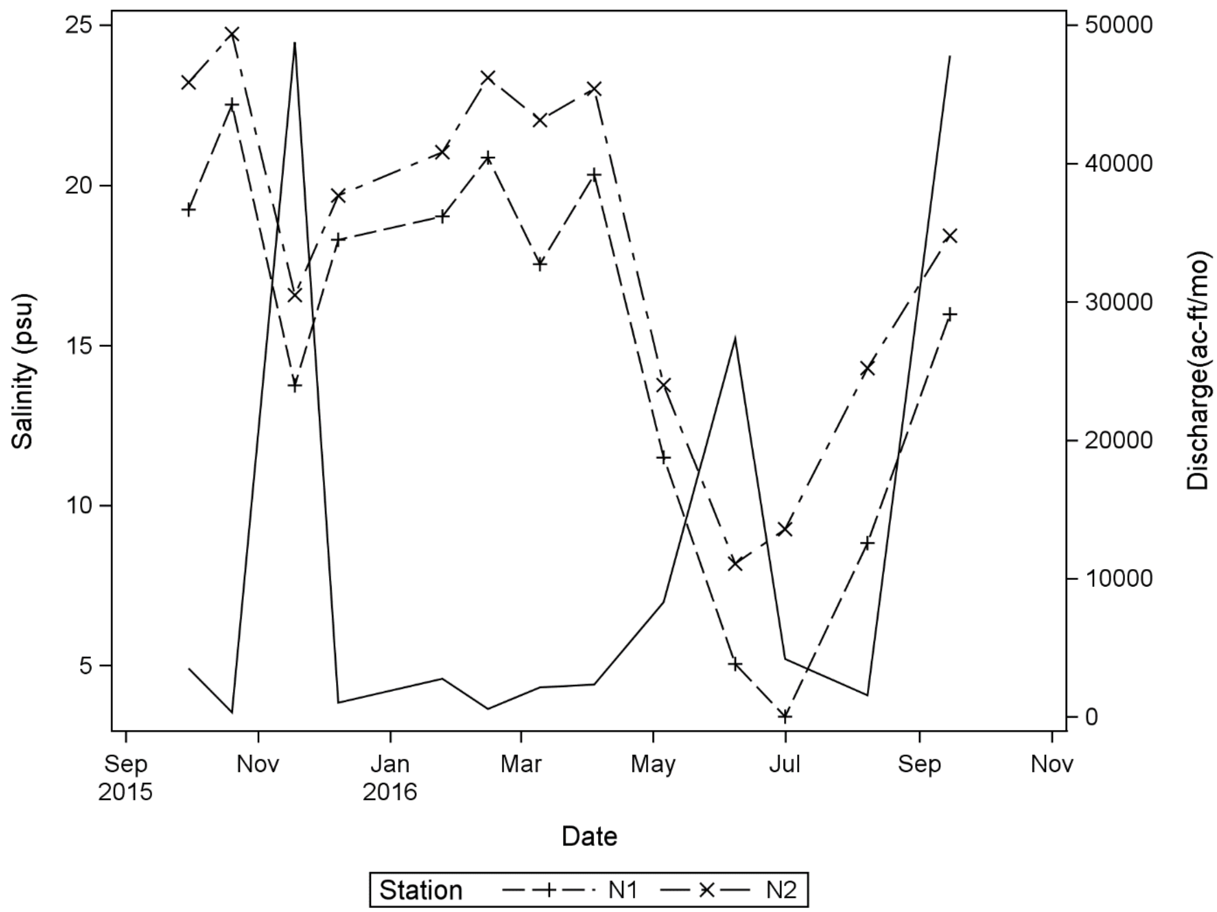

Figure 2. Cumulative inflow from USGS Gage 08162600 and monthly discrete salinity values for stations N1 and N2 in Tres-Palacios Bay. The solid line represents inflow.

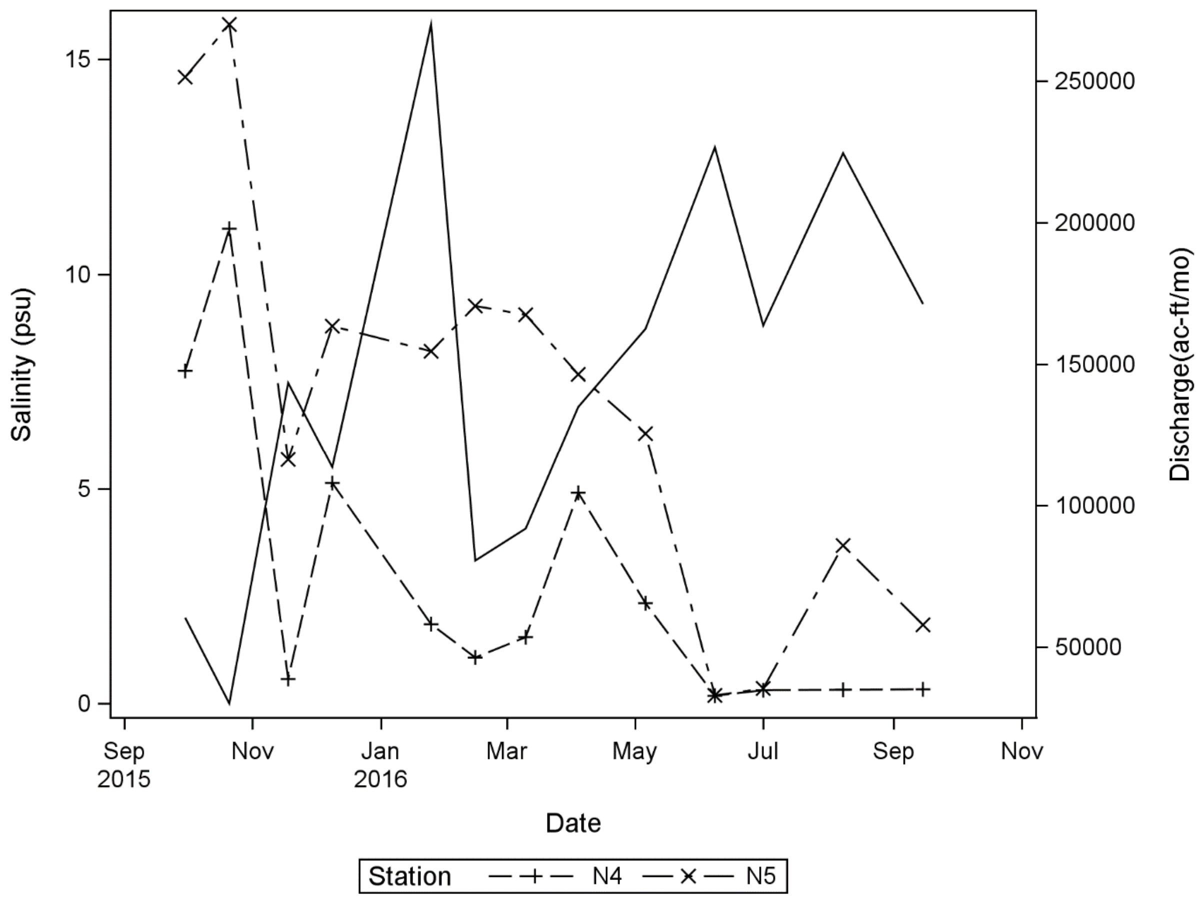

Figure 3. Cumulative Inflow from USGS Gage 08188810 and monthly discrete salinity values for stations N4 and N5 in San Antonio Bay. The solid line represents inflow.

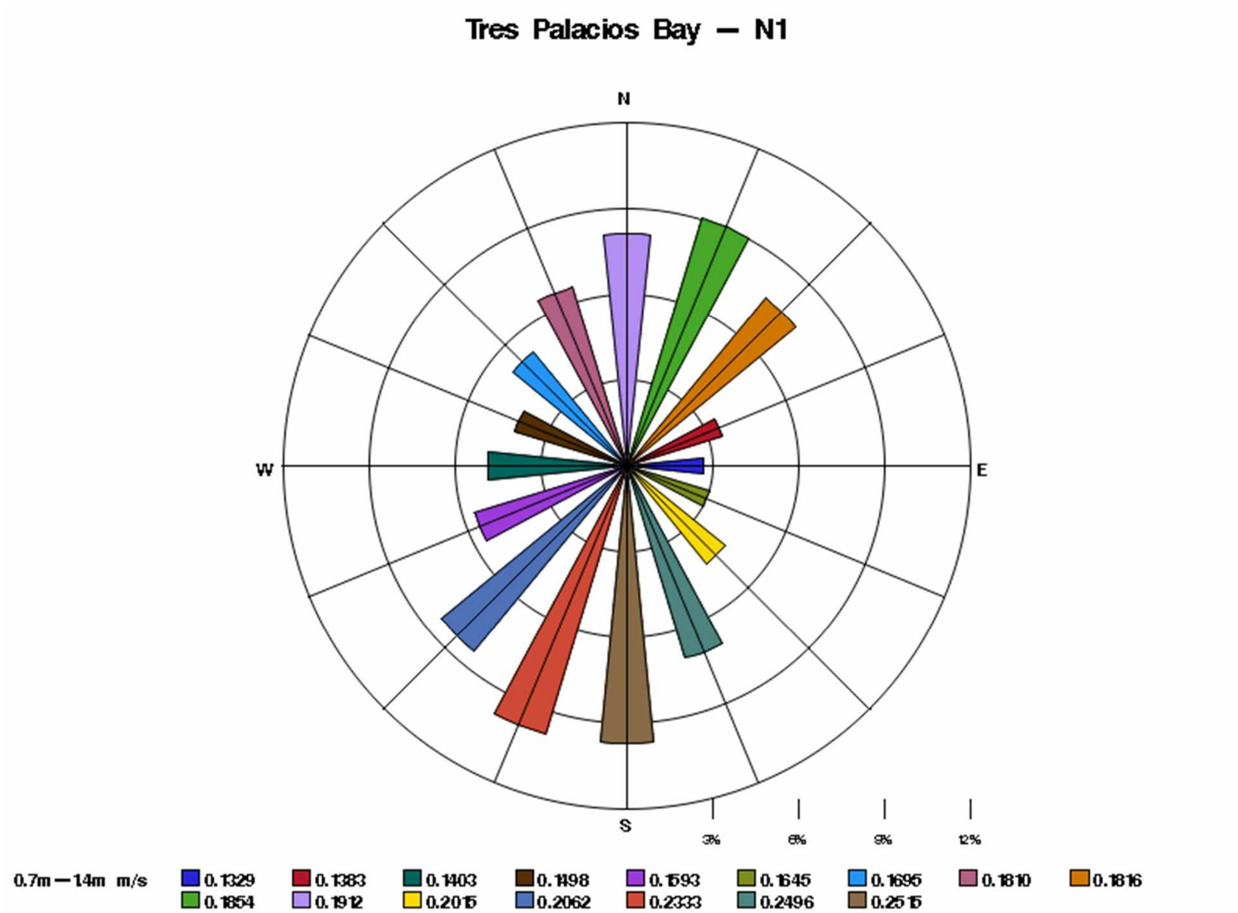

Figure 4. Current flow probability and average speed in upper Tres-Palacios Bay station N1 between October 2015 to March 2016.

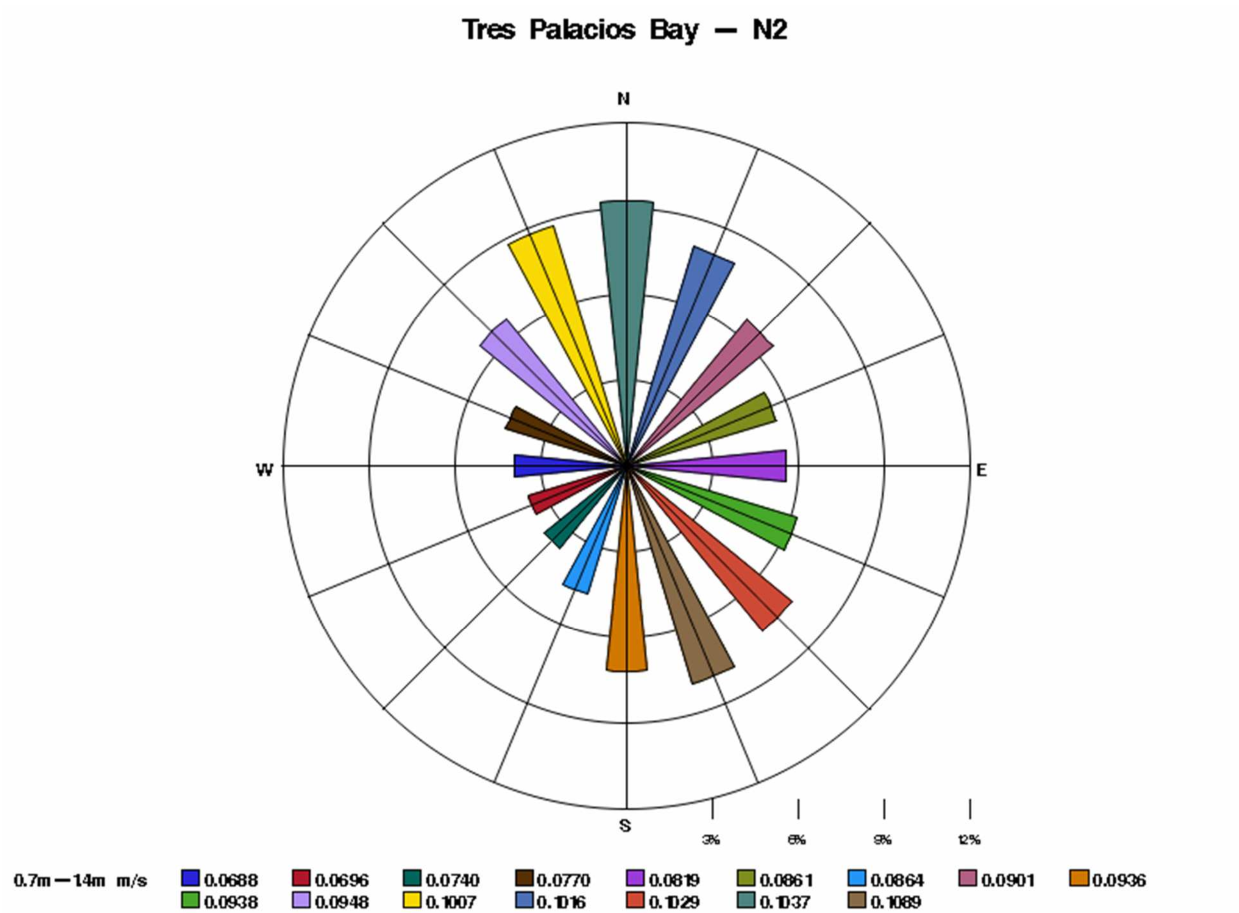

Figure 5. Current flow probability and average speed in lower Tres-Palacios Bay station N2 between October 2015 to March 2016.

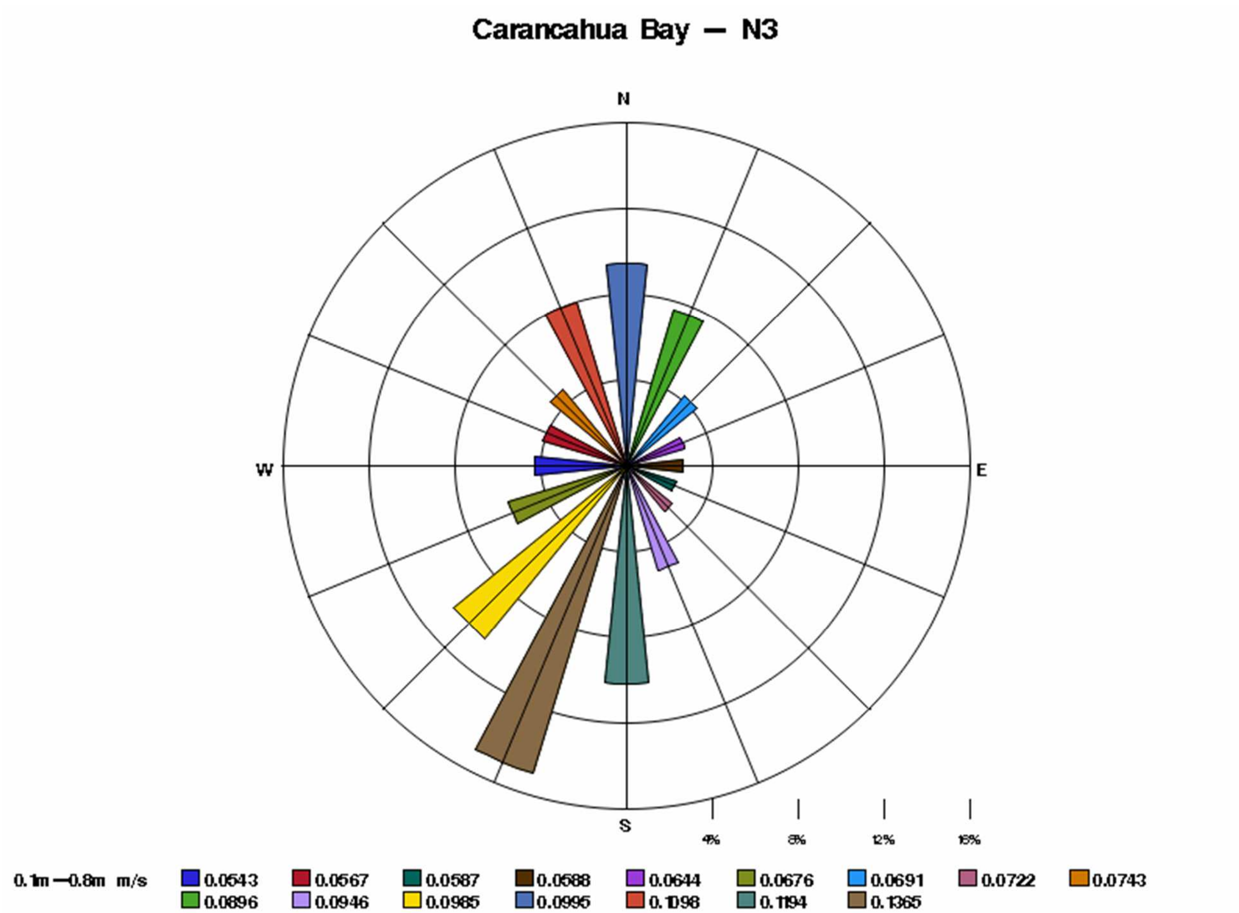

Figure 6. Current flow probability and average speed in Carancahua Bay station N3 between October 2015 to March 2016.

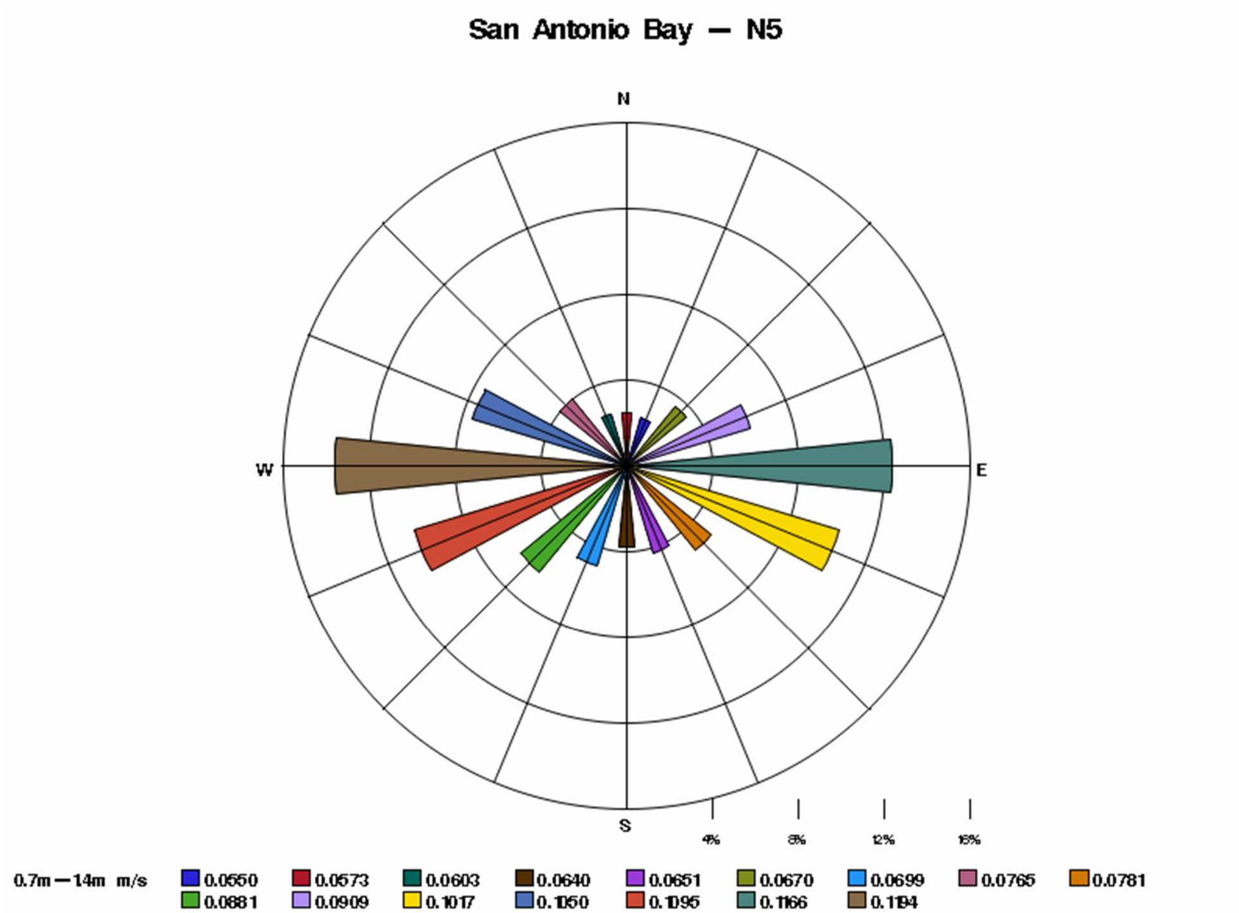

Figure 7. Current flow probability and average speed in lower San Antonio Bay station N5 between October 2015 to March 2016.

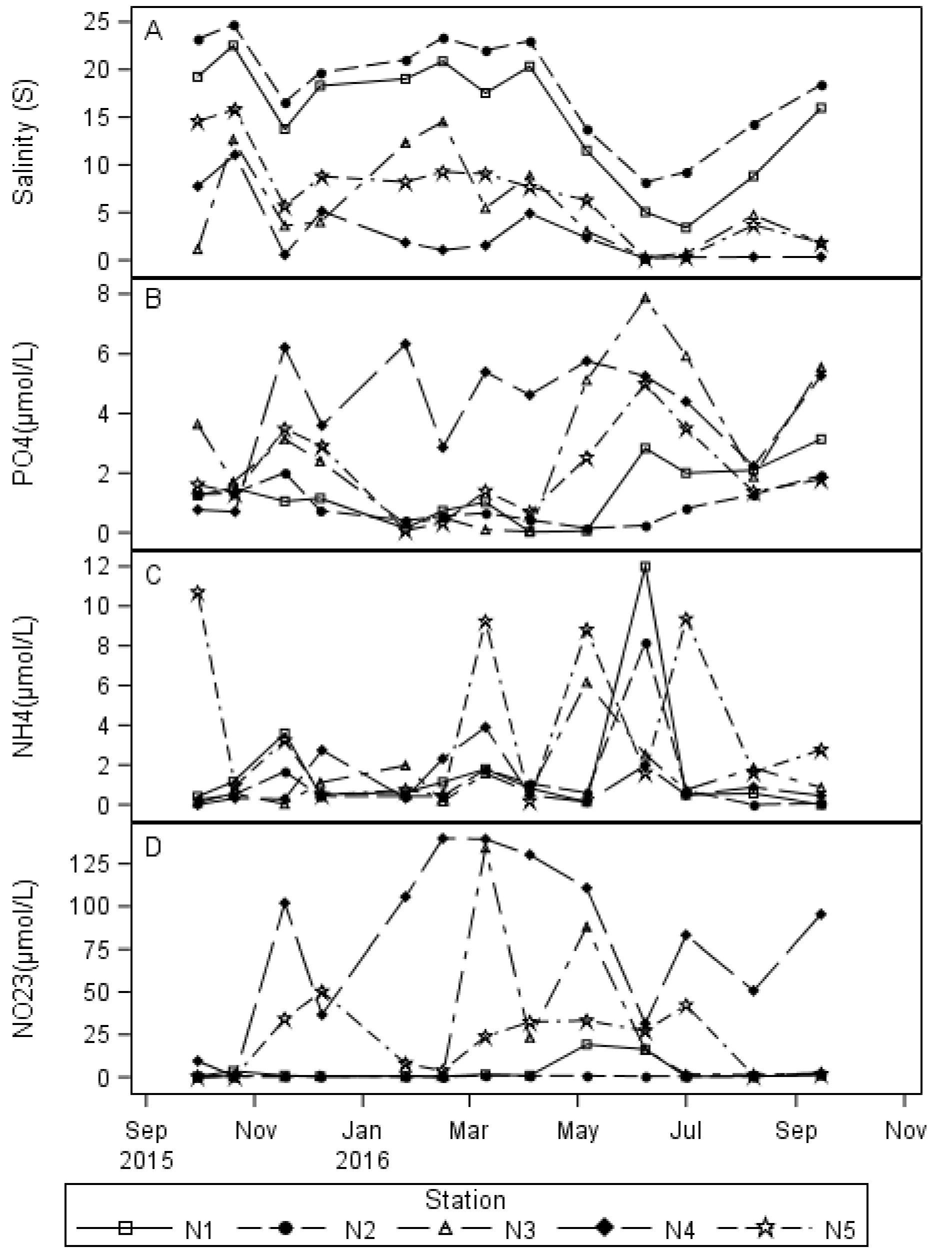

Figure 8. Monthly sampled nutrient concentrations as compared to salinity at Stations N1-N5 from September 2015 to September 2016.

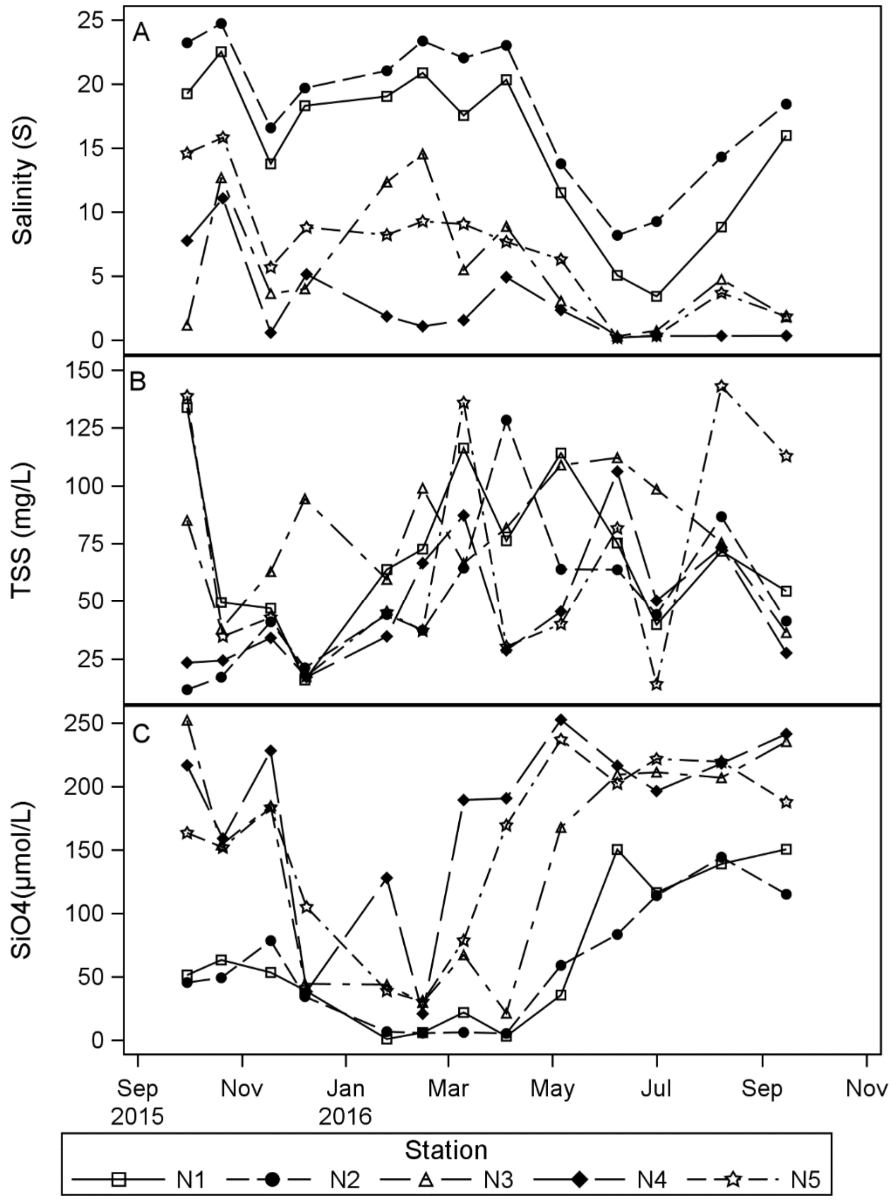

Figure 9. Monthly sampled suspended solids and silicate as compared to salinity at Stations N1-N5 from September 2015 to September 2016.

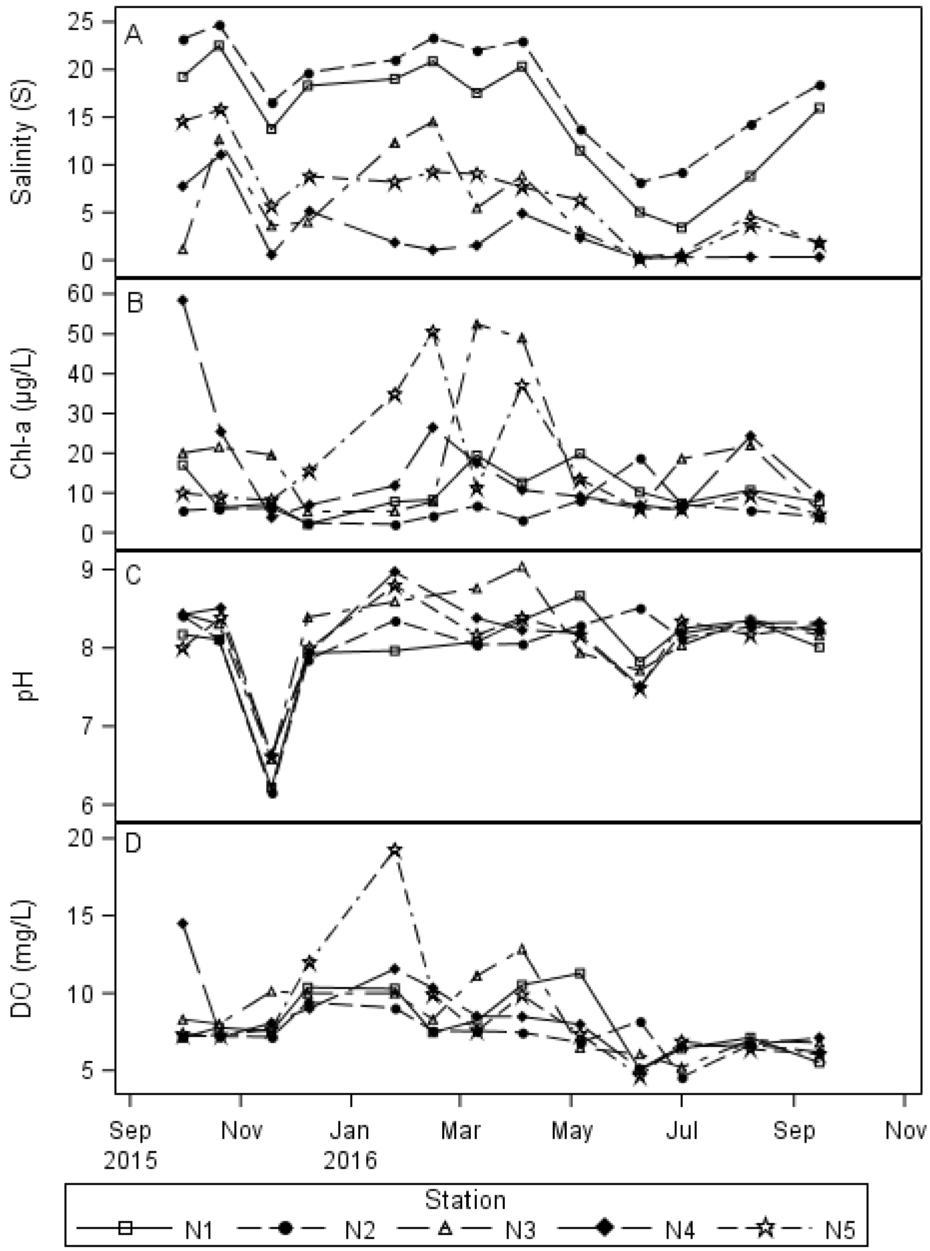

Figure 10. Monthly sampled salinity, chlorophyll-a, pH and dissolved oxygen concentrations as compared to salinity at Stations N1-N5 from September 2015 to September 2016.

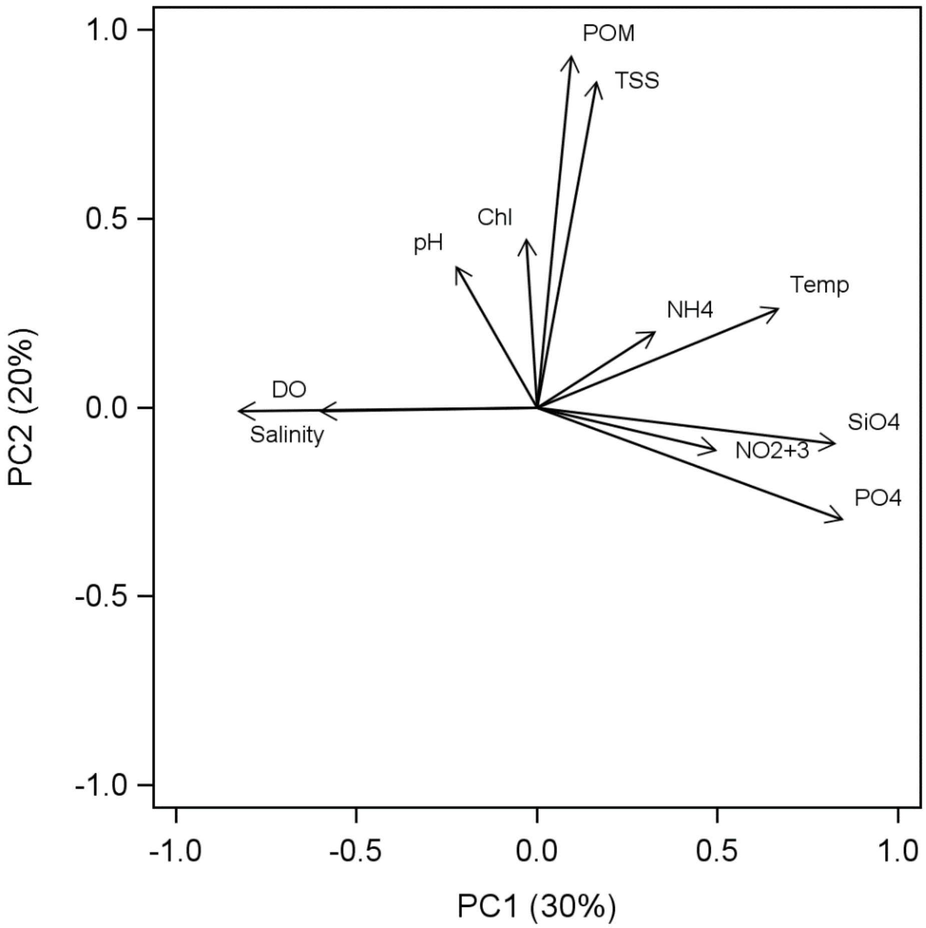

Figure 11. Principal components analysis (PCA) variable loads for hydrographic characteristics using PC1 and PC2, stations N1-N5, from September 2015 to September 2016. Abbreviations: DO = Dissolved Oxygen (mg/L), Temp = Temperature (° C), Chl = Chlorophyll-a (µg/L), NH4 = Ammonium (µmol/L), NO2+3 = Nitrate + Nitrite (µmol/L), SiO4 = Silicate (µmol/L), PO4 = Phosphate (µmol/L), TSS = Total Suspended Solids (mg/L), POM= Particulate Organic Matter (mg/L).

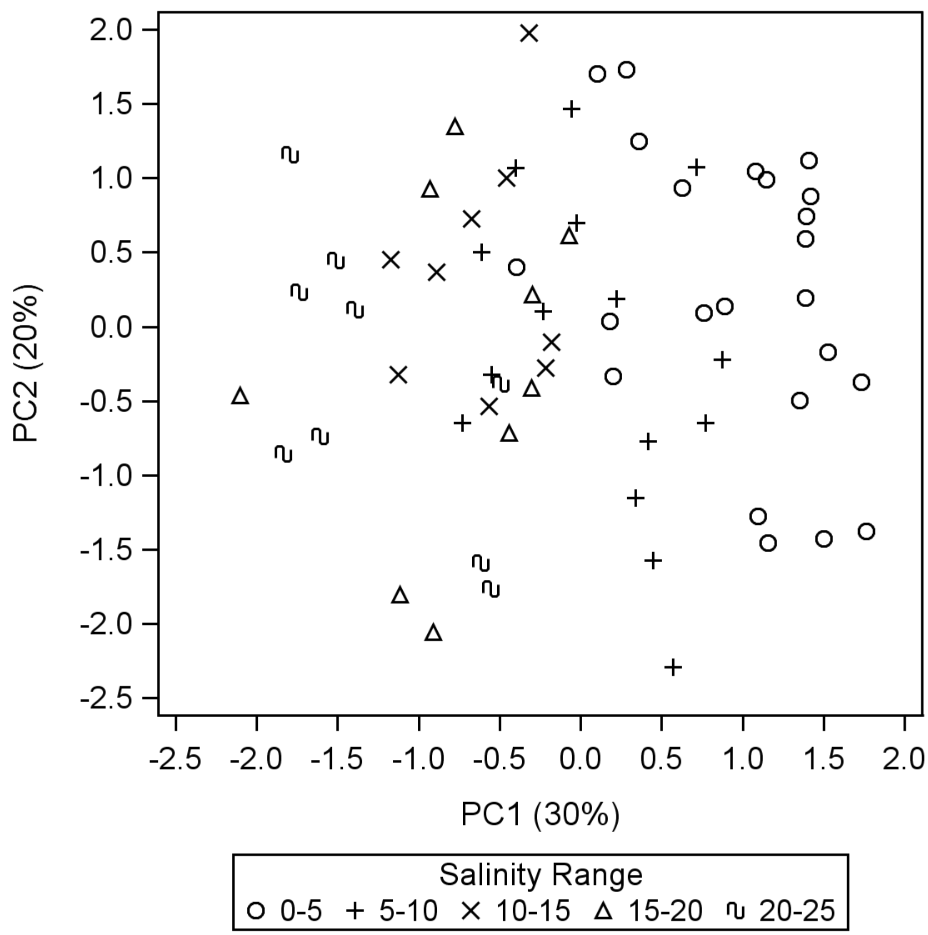

Figure 12. Principal components analysis (PCA) station scores for hydrographic characteristics using PC1 and PC2, stations N1-N5, from September 2015 to September 2016.

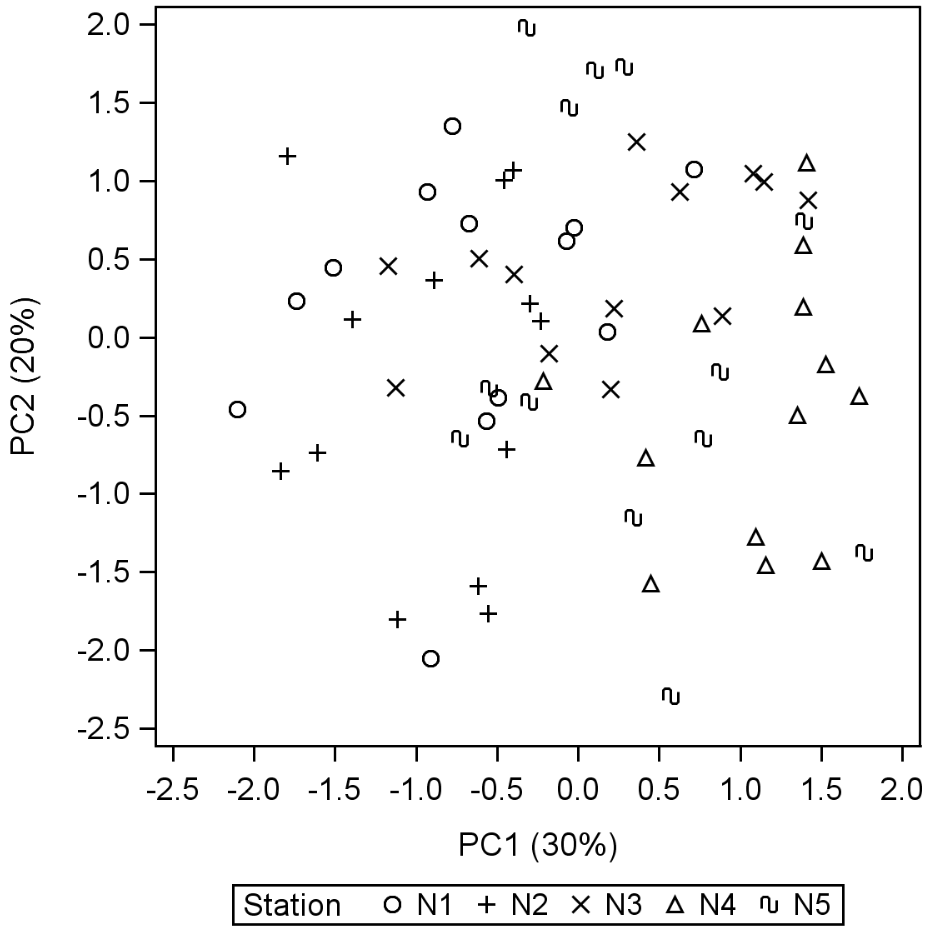

Figure 13. Principal components analysis (PCA) station scores for hydrographic characteristics using PC1 and PC2, stations N1-N5, from September 2015 to September 2016.

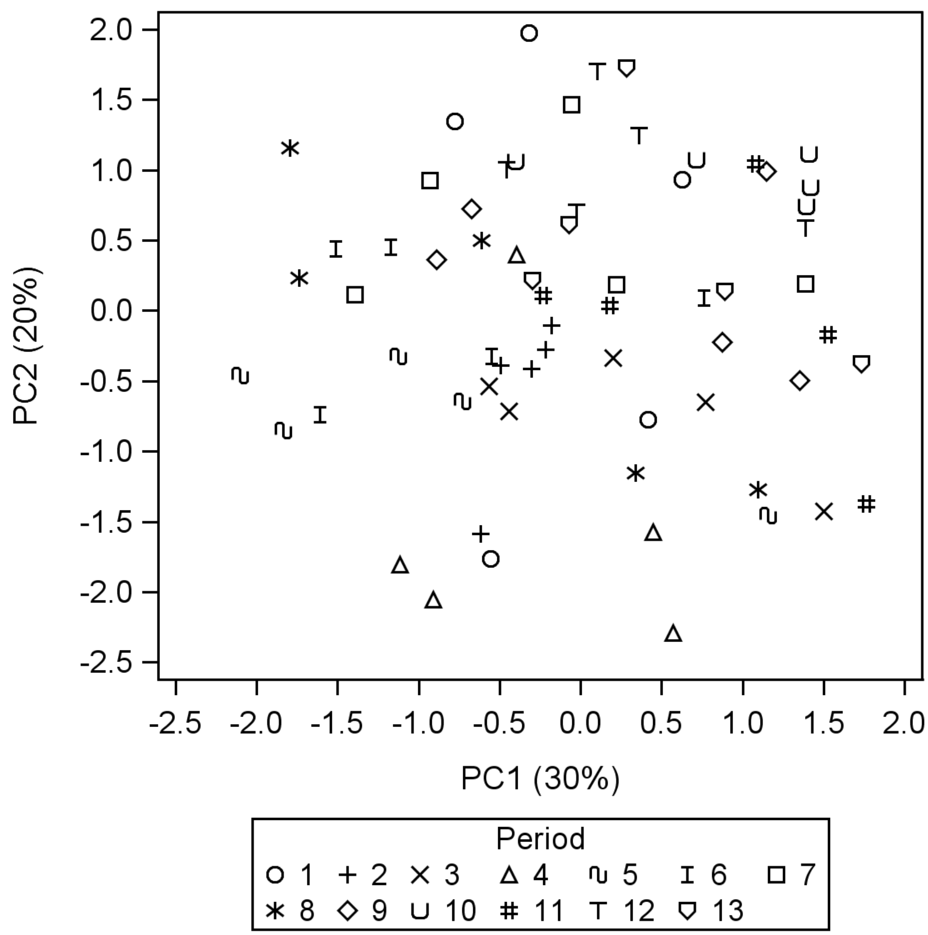

Figure 14. Principal components analysis (PCA) monthly scores for hydrographic characteristics using PC1 and PC2, stations N1-N5, from September 2015 to September 2016.

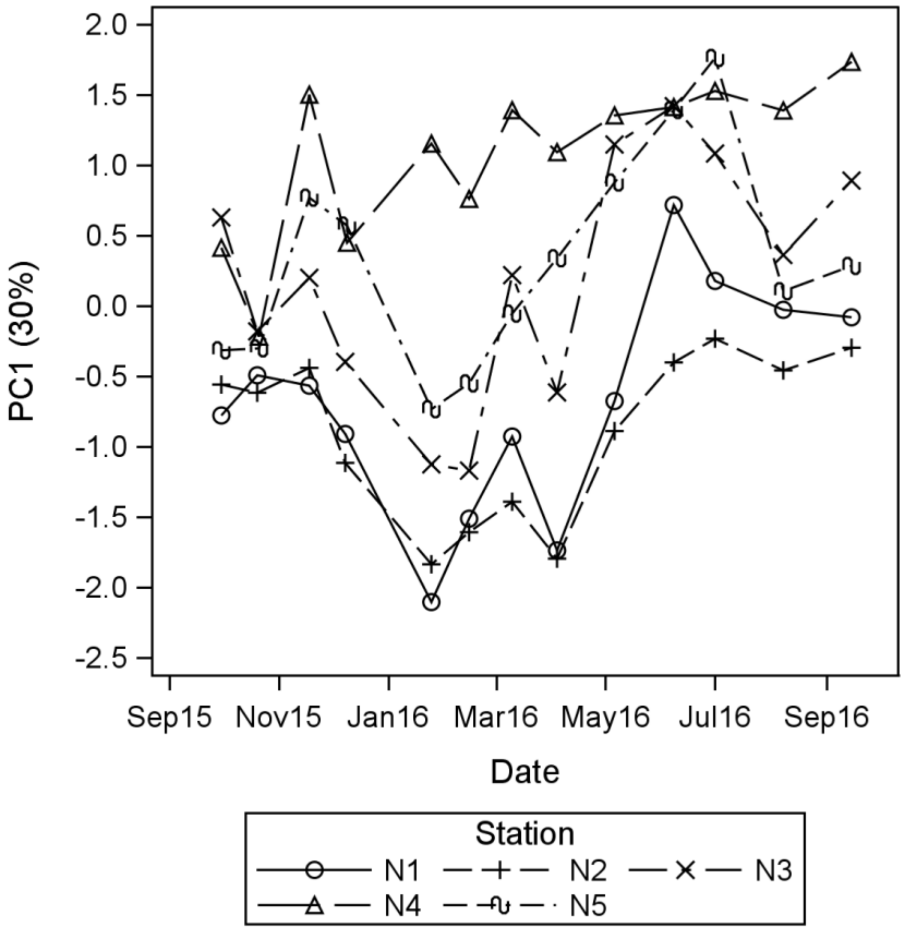

Figure 15. PC1 sample score coefficients plotted across stations N1-N2 from September 2015 to September 2016.

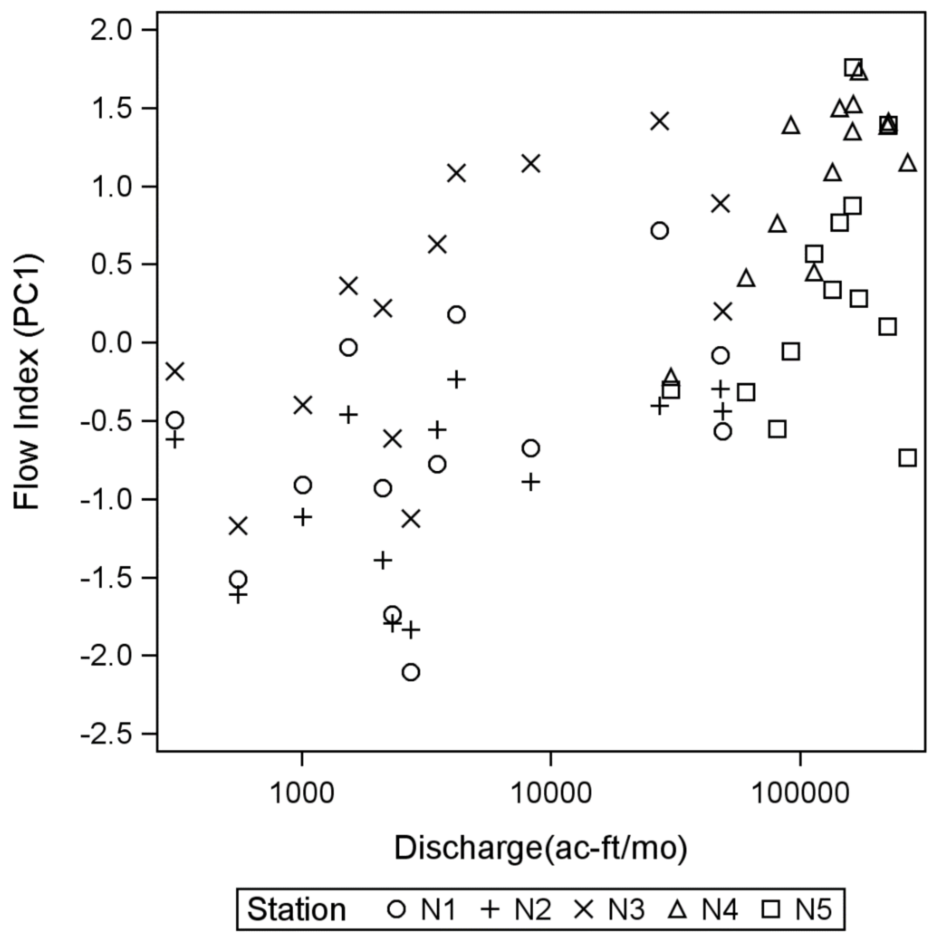

Figure 16. Flow Index (PC1 sample scores) plotted against discharge (ac-ft/mo) across stations N1-N5 from September 2015 to September 2016.

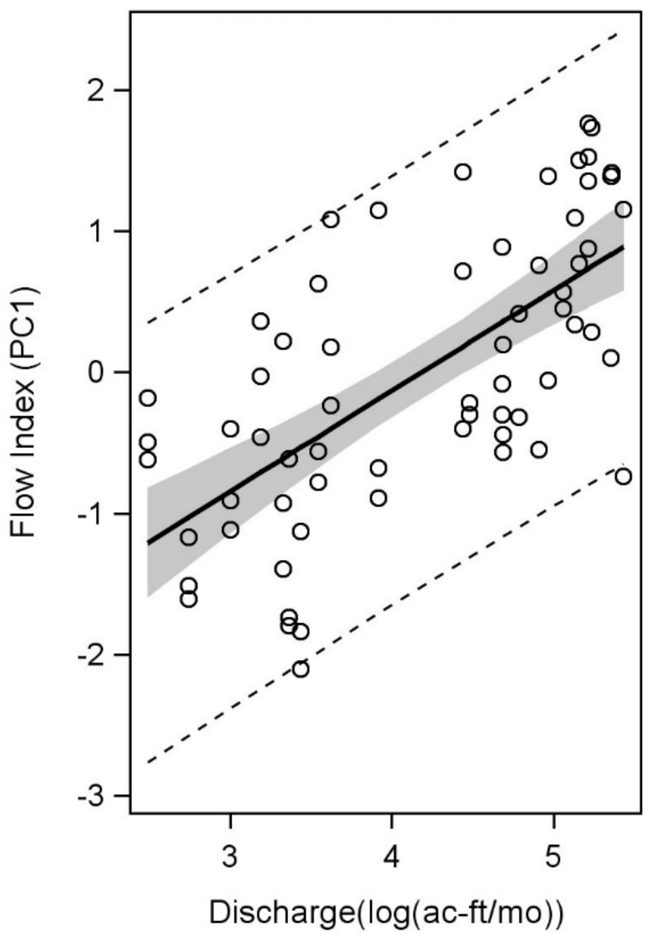

Figure 17. Linear regression on flow index (PC1 sample scores) and log discharge (ac-ft/mo). Shaded area represents 95% confidence limits, and dashed lines represent 95% prediction limits.

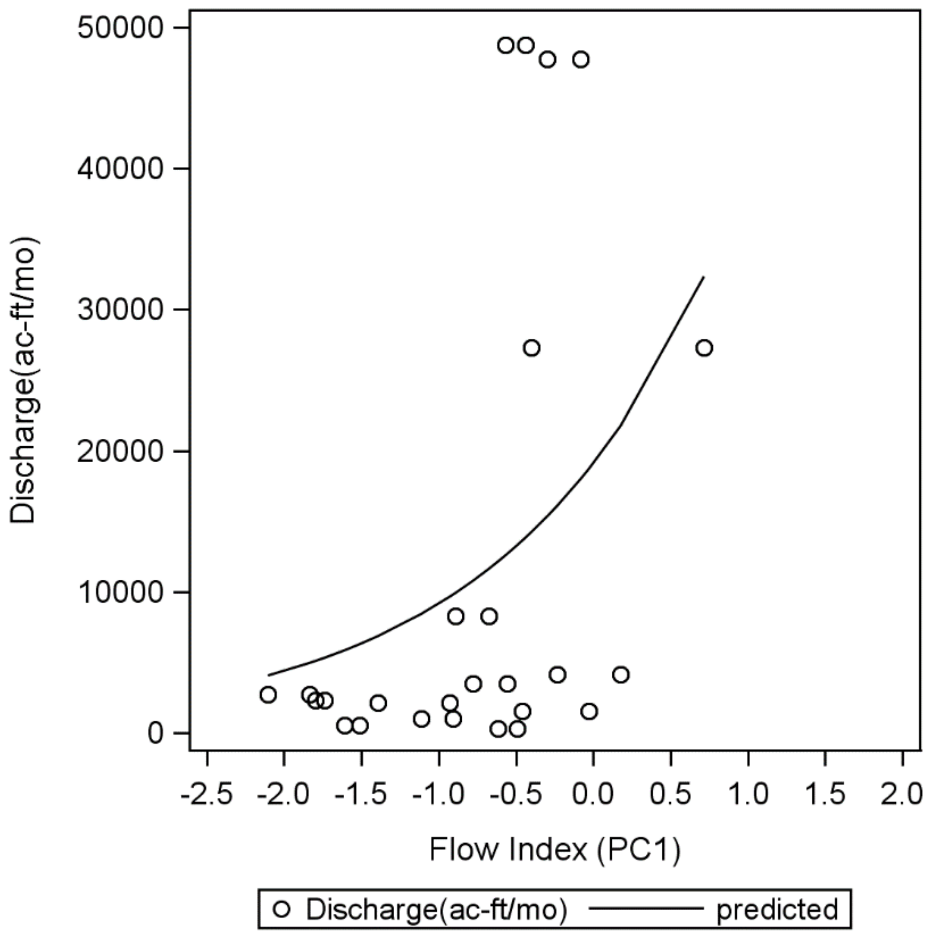

Figure 18. Predicting discharge by the flow index at Tres-Palacios Bay (stations N1 and N2).

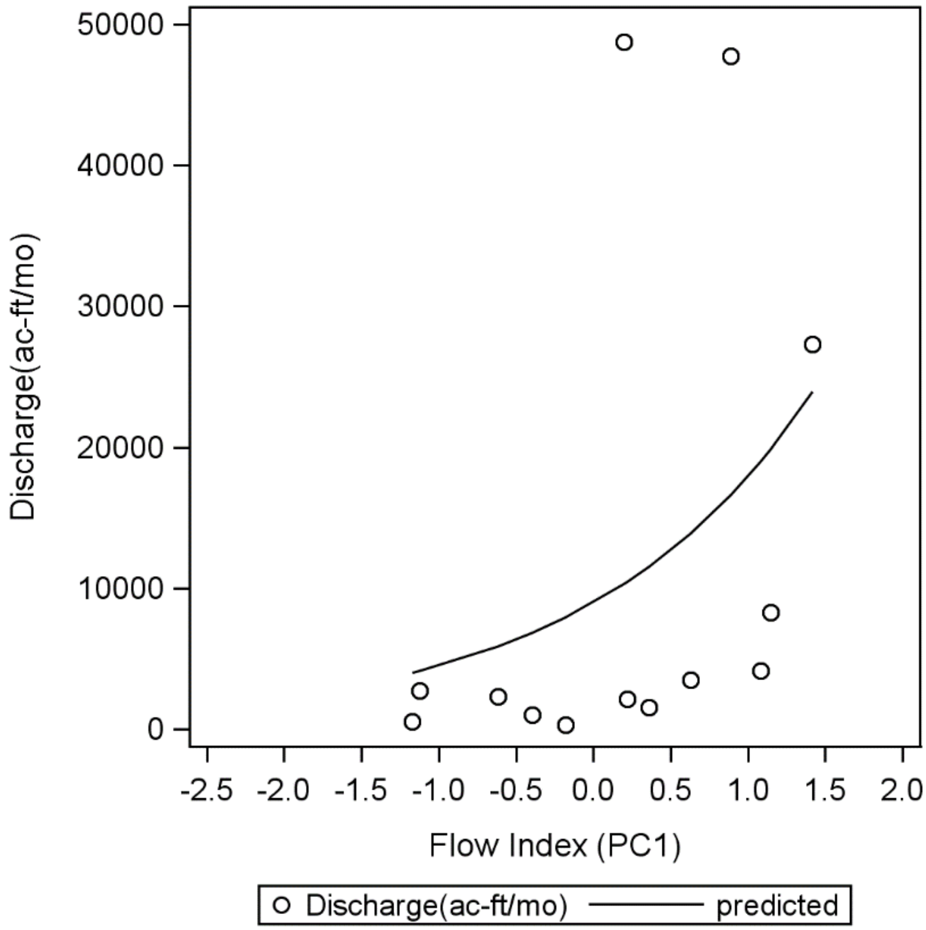

Figure 19. Predicting discharge by the flow index at Carancahua Bay (station N3).

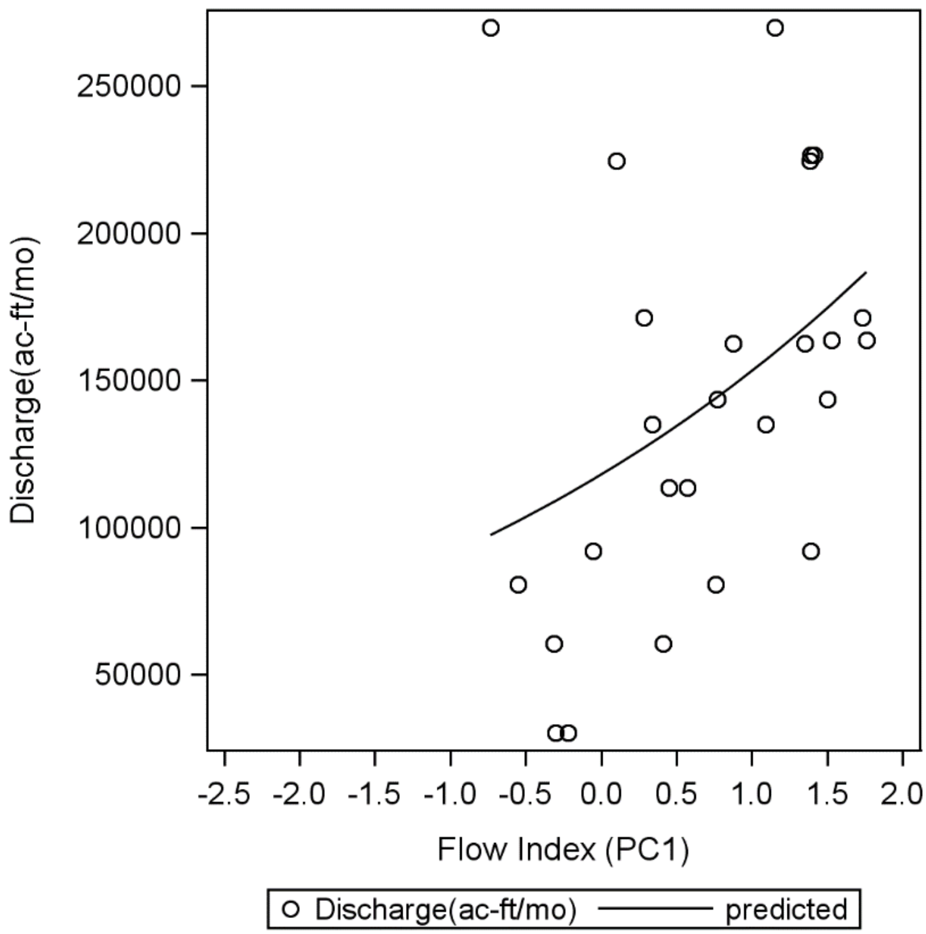

Figure 20. Predicting discharge by the flow index at San Antonio Bay (stations N4 and N5).

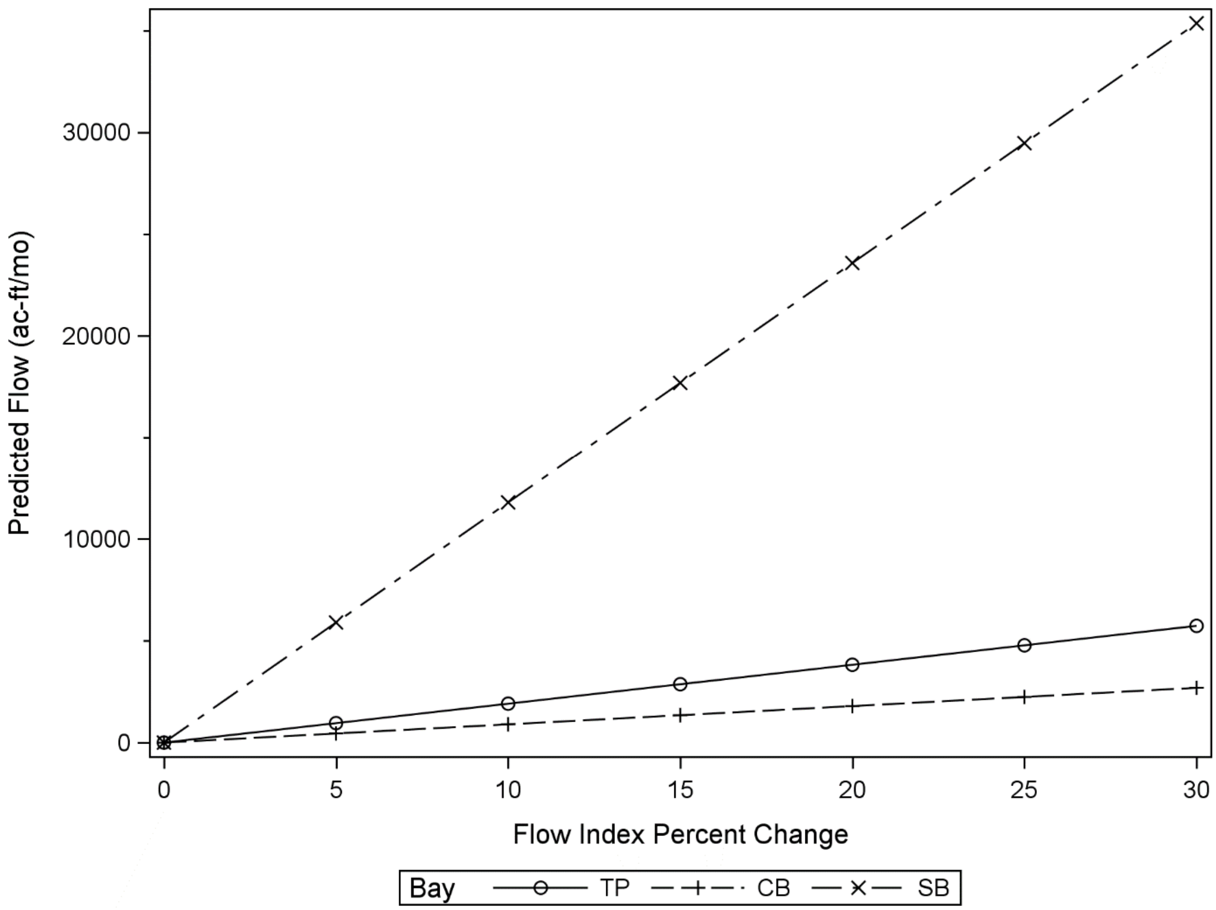

Figure 21. Percent change estimate in flow index (PC1) values and corresponding estimates for flow (ac-ft/mo) at Tres-Palacios Bay, Carancahua Bay, and San Antonio Bay.

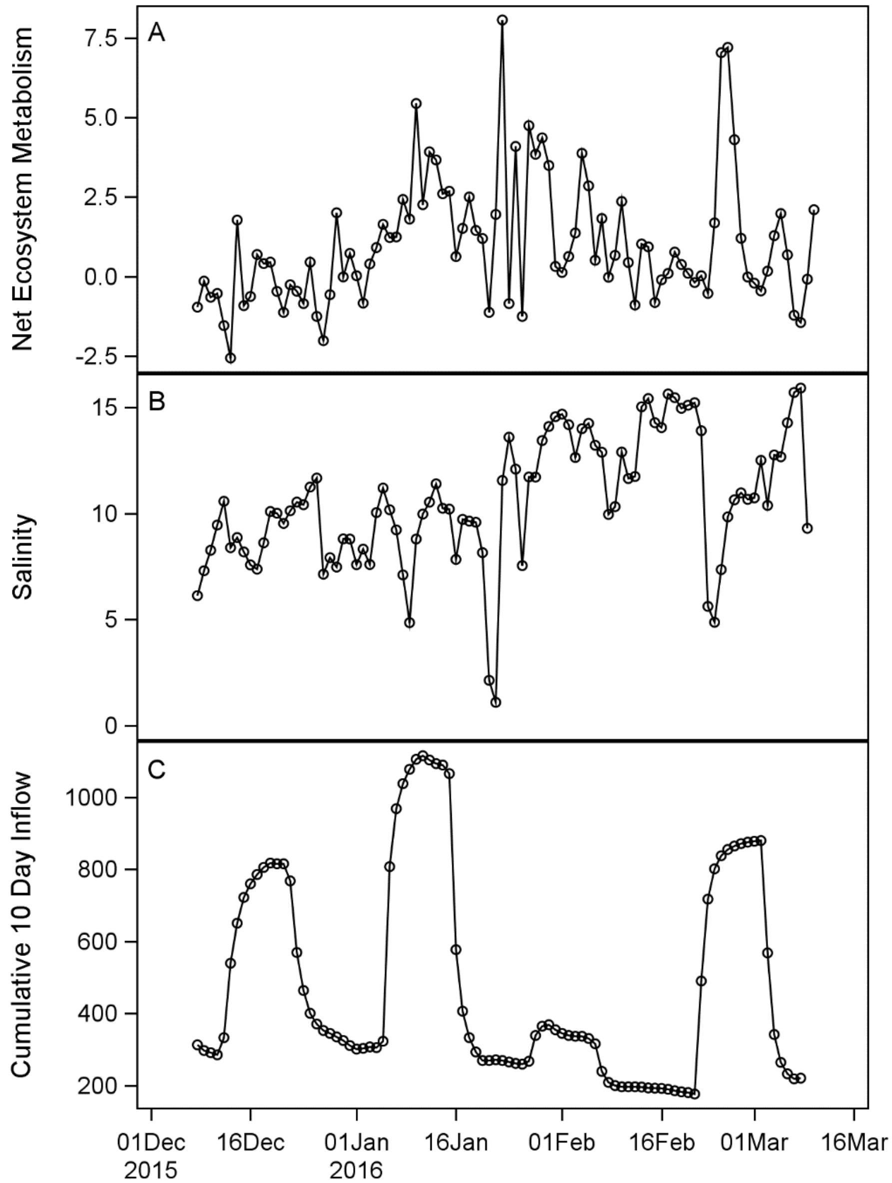

Figure 22. Daily averaged net ecosystem metabolism (mg O_2/L/Day), salinity (S), and cumulative 10 day inflow (ac-ft) at station N3 from December 2015 to March 2016.

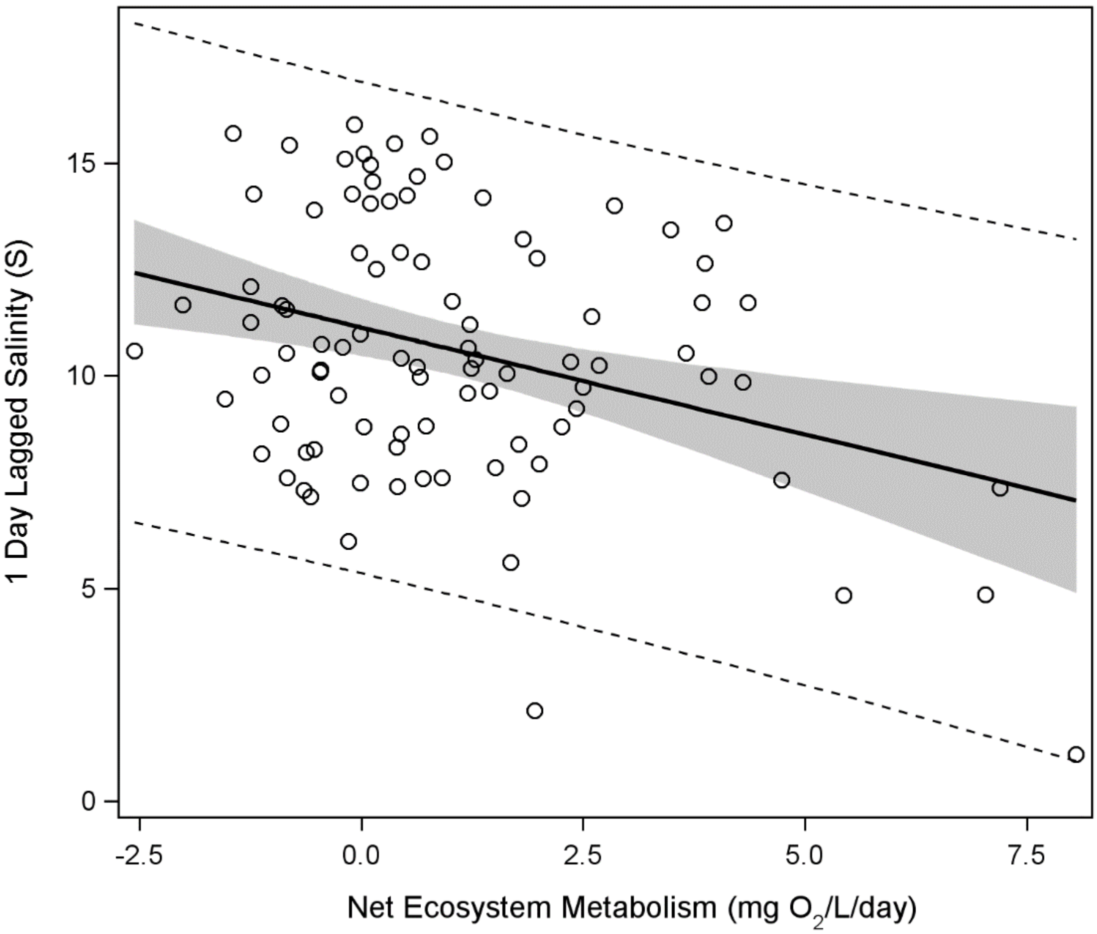

Figure 23. Linear regression between net ecosystem metabolism (mg O_2/L/Day) and lagged salinity (S) at station N3. Shaded area represents 95% confidence limits, and dashed lines represent 95% prediction limits.

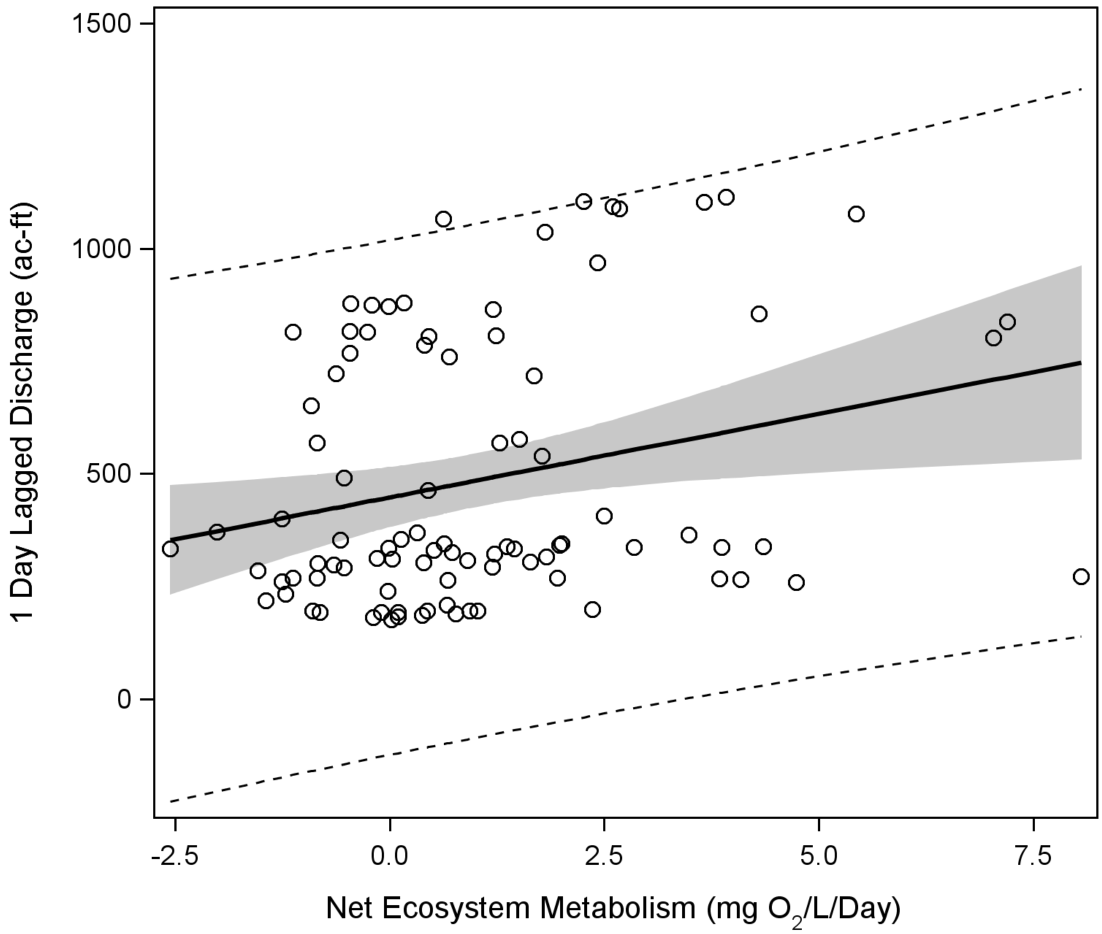

Figure 24. Linear regression between net ecosystem metabolism (mg O_2/L/Day) and lagged 10 day cumulative discharge (ac-ft) at station N3. Shaded area represents 95% confidence limits, and dashed lines represent 95% prediction limits.

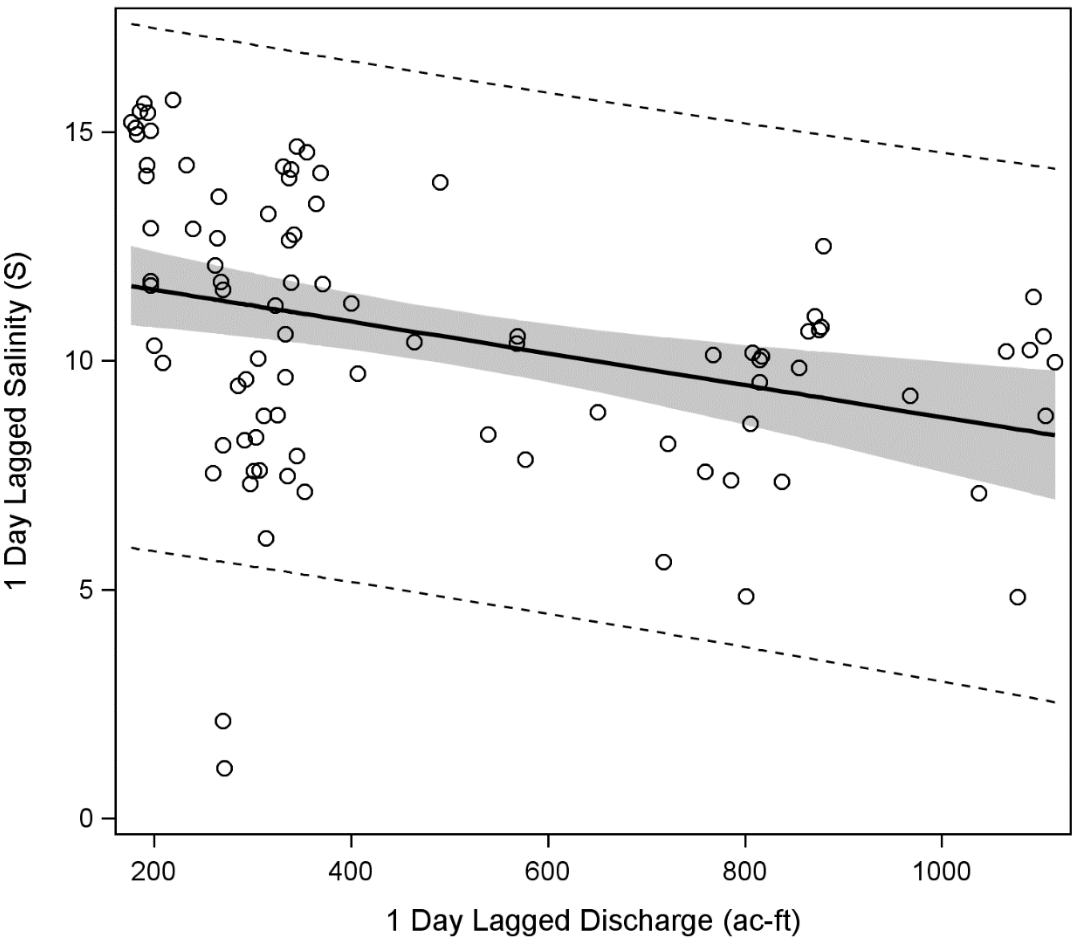

Figure 25. Linear regression between 1 day lagged salinity (S) and 1 day lagged cumulative discharge (ac-ft) at station N3. Shaded area represents 95% confidence limits, and dashed lines represent 95% prediction limits.