Article

Assessing linkages between watershed nutrient loading and estuary water quality in Lavaca Bay, Texas

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Abstract: A single paragraph of about 200 words maximum. For research articles, abstracts should give a pertinent overview of the work. We strongly encourage authors to use the following style of structured abstracts, but without headings: 1) Background: Place the question addressed in a broad context and highlight the purpose of the study; 2) Methods: Describe briefly the main methods or treatments applied; 3) Results: Summarize the article's main findings; and 4) Conclusion: Indicate the main conclusions or interpretations. The abstract should be an objective representation of the article, it must not contain results which are not presented and substantiated in the main text and should not exaggerate the main conclusions.

Keywords: estuary; nutrient loading; water quality; Texas

1. Introduction

Like many coastal areas globally, coastal watersheds along the Texas Gulf coast are facing pressures from increasing population, increases in point source and non-point source pollution and alterations to freshwater flows that alter water quality in downstream estuaries [1–3]. Despite these increasing pressures, national scale assessments have classified coastal estuaries in Texas as moderate or lower for exhibiting eutrophic conditions [1]. However, a suite of recent studies indicate that estuary water quality dynamics in both agriculturally dominated and urban watersheds within Texas are in fact expressing conditions that are increasingly conducive to algal blooms and eutrophication [3–6]. With identification of localized areas of estuary water quality concern along the Texas coast [3], localized studies are being prioritized to better inform management actions.

This project provides an assessment of nutrient loading and water quality responses in Lavaca Bay, Texas. Lavaca Bay is a secondary bay in the larger Matagorda Bay system located roughly halfway between Houston, Texas and Corpus Christi, Texas. Lavaca Bay has faced substantial challenges associated with legacy contamination but general water quality parameters such as dissolved oxygen (DO), nutrients, and biological parameters have been well within state water quality standards. However, long-term declines of benthic fauna abundance, biomass, and diversity in Lavaca Bay that are linked to reductions in freshwater inflows and changes in estuary salinity [7–9] and are a concern to local stakeholders. Bugica *et al.* [3] identified monotonic increases in total phosphorus (TP), orthophosphate, total Kjeldahl nitrogen (TKN), and chlorophyll-*a* at sites within Lavaca Bay. Although no long-term changes in DO concentrations were identified, the trends in nutrient concentrations are concerning due to the role of nitrogen as a limiting factor for primary production in many Texas estuaries [5,10–13] and the ramifications that major changes in nitrogen loadings could have for productivity and eutrophication in Lavaca Bay.

There are ongoing efforts between local, state, and federal agencies to address water quality impairments in the freshwater portions of the Lavaca Bay watershed [14–16]. However, at a statewide scale, these approaches have shown limited success and emphasize a need for improved efforts at assessing and linking management actions with down-

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stream water quality to better identify and replicate effective management actions across the state [17]. The identification and communication of changes and trends in water quality is complicated by the fact that trends are often non-linear and confounded by precipitation and runoff that hinder traditional analysis [18,19]. To provide actionable information for resource managers, water quality conditions must be evaluated relative to changes in natural environmental drivers to better understand and manage potential anthropogenic effects.

Here something some we present methodology and results for estimating nutrient loading- and assessing linkages in estuary water quality parameters with watershed derived nutrient loads. . . .

2. Materials and Methods

2.1. Study Area and Data

Lavaca Bay is a secondary bay in the Matagorda Bay system located on the Texas Gulf coast, roughly halfway between the cities of Houston and Corpus Christi (Figure **??**). Lavaca Bay is 190 km^2 with the majority of freshwater inflow provided by the Lavaca and Navidad River systems. The Garcitas-Arenosa, Placedo Creek, and Cox Bay watersheds provide additional freshwater inflows. The entire watershed land area for Lavaca Bay is 8,149 km^2. The Lavaca and Navidad River watersheds are a combined 5,966 km^2, or approximately 73% of the entire Lavaca Bay watershed area. Discharge from the Navidad River is regulated by Lake Texana which has been in operation since 1980. Lake Texana provides 170,000 acre-feet of water storage and discharges into the tidal section of the Navidad River which ultimately joins the tidal section of the Lavaca River 15 km upstream of the confluence with the Bay.

Need to add a short description of land use

Daily discharges for the Lavaca River (USGS-08164000, Figure **??**) were obtained from the United States Geologic Survey (USGS) National Water Information System using the *dataRetrieval* R package [20]. Gaged daily discharges from the outlet of Lake Texana on the Navidad River (USGS-0816425) were provided by the Texas Water Development Board (TWDB) (April 21, 2022 email from R. Neupane, TWDB).

Water quality sample data for the two freshwater and three estuary locations were obtained from the Texas Commission on Environmental Quality (TCEQ) Surface Water Quality Monitoring Information System. Data submitted through the system are required to be collected under Quality Assurance Project Plans and lab method procedures outlined by the TCEQ's procedures manual. The QAPP and procedures manuals ensure the consistent collection and laboratory methods are applied between samples collected by different entities and under different projects. All sites had varying lengths of and availability of data. For freshwater locations, TP from January 2000 through December 2020 and nitrate-nitrogen (NO_3) data from January 2005 through December 2020 were downloaded (Table 1). Less than 5-years of total nitrogen and TKN concentration data were available at the freshwater sites and deemed insufficient to develop load estimation models (**CITE**). The three estuary sites included an upper Lavaca Bay site near the outlet of the Lavaca River system (TCEQ-13563), a mid-Lavaca Bay site (TCEQ-13383), and the lower Lavaca Bay site near the mouth of the Bay (TCEQ-13384). For estuary locations, we obtained data for TP, Nitrite+Nitrate (NO_x), TKN, chlorophyll-*a*, and DO concentrations from January 2005 through December 2020 (Table 2).

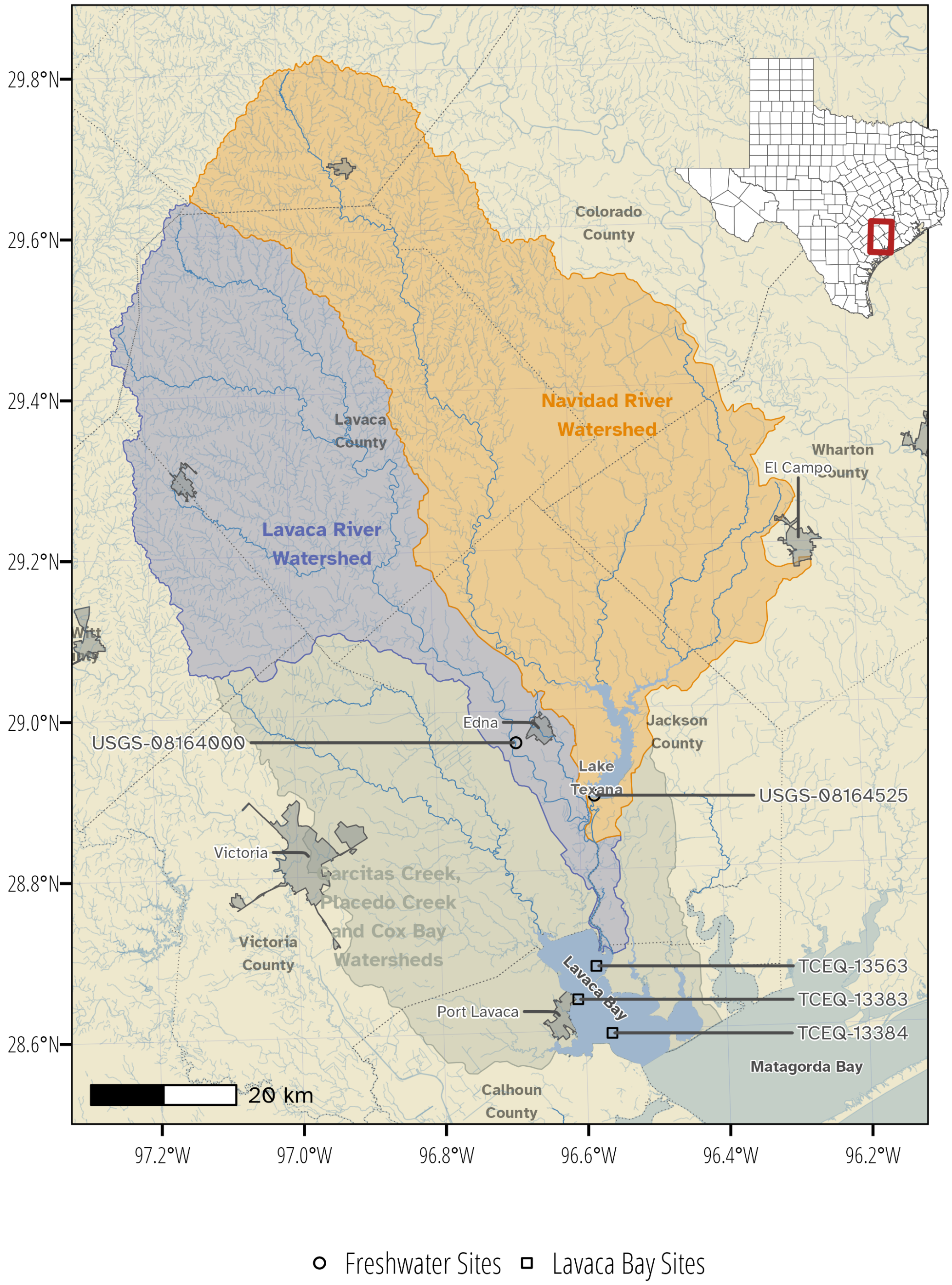

Figure 1. Map of the Lavaca Bay watershed, location of USGS gages where nutrient loads were calculated, and location of estuary water quality sampling sites.

Table 1. Summary of gauged streamflow and freshwater water quality samples between January 1, 2000 and December 31, 2020.

Station ID		Mean	SD	N
USGS-08164000	TP (mg/L)	0.21	0.09	80
	NO_3 (mg/L)	0.18	0.24	74
	Mean Daily Streamflow (cfs)	332.78	1667.47	7671
USGS-08164525	TP (mg/L)	0.20	0.08	81
	NO_3 (mg/L)	0.29	0.26	62
	Mean Daily Streamflow (cfs)	666.14	2957.79	7671

Table 2. Summary of estuary water quality samples collected between January 1, 2005 and December 31, 2020.

Station ID		Mean	SD	N
TCEQ-13383	TP (mg/L)	0.11	0.05	47
	NO_x (mg/L)	0.07	0.15	51
	TKN (mg/L)	0.94	0.49	45
	Chlorophyll-*a* (μg/L)	9.43	5.31	47
	DO (mg/L)	7.22	1.35	55
TCEQ-13384	TP (mg/L)	0.08	0.03	51
	NO_x (mg/L)	0.06	0.08	52
	TKN (mg/L)	0.76	0.40	48
	Chlorophyll-*a* (μg/L)	8.22	6.44	46
	DO (mg/L)	7.51	1.32	54
TCEQ-13563	TP (mg/L)	0.13	0.06	50
	NO_x (mg/L)	0.09	0.13	53
	TKN (mg/L)	0.94	0.37	49
	Chlorophyll-*a* (μg/L)	9.67	5.33	49
	DO (mg/L)	7.91	1.34	56

2.2. Estimating Watershed Based Nutrient Loads

Estimates of NO_3 and TP loads at the Lavaca River (USGS-08164000) and the outlet of Lake Texana on the Navidad River (USGS-08164250) were developed using Generalized Additive Models (GAMs) relating nutrient concentration to river discharge, season, and time. Separate models were fit at each station for each parameter and used to predict nutrient concentrations for each day in the study period. GAMs can be specified in a functionally similar manner to the commonly used LOADEST [21] or WRTDS [22] regression models and have been shown to produce reliable estimates of nutrient and sediment loadings [23–29]. GAMs are a semiparametric extension of generalized linear models where the linear predictor is represented as the sum of multiple unknown smooth functions and parametric linear predictors [30]. Although the underlying parameter estimation procedure of GAMs is substantially different than WRTDS, both the functional form and results are demonstrated to be similar [31]. The use of GAMs over other regression-based approaches was (1) the ability to easily explore and incorporate different model terms, (2) the ability to incorporate non-linear smooth function without explicit apriori knowledge of the expect shape, and (3) the ability to specify a link function that relates the expected value of the response to the linear predictors and allows use to avoid data transformations as much as possible.

GAMs were fit using the *mgcv* package in R which makes available multiple types of smooth functions with automatic smoothness selection [30]. The general form of the model relating NO_3 and TP concentration to streamflow, season, and time was:

$$g(\mu) = \alpha + f_1(ddate) + f_2(yday) + f_3(log1p(Q)) + f_4(ma) + f_5(fa)$$
$$y \sim \mathcal{N}(\mu, \sigma^2), \tag{1}$$

where μ is the conditional expected NO_3-N or TP concentration, *g()* is the log-link, α is the intercept, $f_n()$ are smoothing functions. y is the response variable (NO_3 or TP concentration) modeled as normally distributed with mean μ and standard deviation σ. *ddate* is the date converted to decimal notation, *yday* is numeric day of year (1-366), and *log1p(Q)* is the natural log of mean daily streamflow plus 1.

Moving average (*ma*) is an exponentially smoothed moving average that attempts to incorporate the influence of prior streamflow events on concentration at the current time period. Wang *et al.* [23], Kuhnert *et al.* [25] and Zhang and Ball [32] refer to this as averaged or smoothed discounted flow and demonstrated improvements in nutrient loading models by including the term. Kuhnert *et al.* [25] expresses *ma* as:

$$ma(\delta) = d\kappa_{i-1} + (1-\delta)\hat{q}_{i-1} \quad \text{and} \quad \kappa_i = \sum_{m=1}^{i} \hat{Q}_m, \tag{2}$$

where δ is the discount factor (here, set equal to 0.95), κ_i is the cumulative flow (Q) up to the ith day.

Flow anomaly (*fa*) is a unitless term that represents how wet or dry the current time period is from a previous time period [32,33]. Long-term flow anomaly (*ltfa*) is the streamflow over the previous year relative to the entire period and calculated as described by Zhang and Ball [32]:

$$ltfa(t) = \bar{x}_{1\,year}(t) - \bar{x}_{entire\,period} \tag{3}$$

and the short-term flow anomaly (*stfa*) calculated as the current day flow compared to the preceding 1-month streamflow:

$$stfa(t) = x_{current\,day}(t) - \bar{x}_{1\,month}(t) \tag{4}$$

where x are the averages of log-transformed streamflow over the antecedent period (*1-year*, *1-month*, etc.) for time t. We used *ltfa* in NO~3 models and *stfa* in TP models based on results from Zhang and Ball [32] demonstrating major improvements in NO_x regression models that incorporated *ltfa* and moderate improvements in TP regression models that incorporated *stfa*.

The calculation of model terms for the Lake Texana site were slightly modified because daily loads are not a function of natural stream flow processes alone, but of dam releases and nutrient concentrations at the discharge point of the lake. Q, *ma*, and *fa* terms were calculated based on total gaged inflow from the 4 major tributaries to the lake. Thinplate regression splines were used for *ddate*, *log1p(Q)*, *fa*, and *ma*. A cyclic cubic regression spline was used for *yday* to ensure the ends of the spline match (day 1 and day 366 are expected to match). First order penalties were applied to the smooths of flow-based variables which penalize departures from a flat function to help constrain extrapolations for high flow measurements.

Left-censored data were not uncommon in this dataset. Several methods are available to account for censored data. We transformed left-censored nutrient concentrations to one-half the detection limit. Although this simple approach can introduce bias [34], we considered it acceptable because high concentrations and loadings are associated with high-flow events and low-flow/low-concentration events will account for a small proportion of total loadings [28].

Daily loads were estimated as the predicted concentration multiplied by the daily streamflow. For the Lake Texana site, model terms were slightly modified because daily loads are a function of dam releases and nutrient concentration, but concentration will be a function of lake inflows and or other lake processes. Q, *ma*, and *fa* terms were calculated based on total gaged inflow from the 4 major tributaries to the lake and laily loads at the dam were calculated from the discrete daily concentration at the discharge point of the lake and corresponding reported daily discharge from the dam. Flow-normalized loads were estimated similar to WRTDS by setting flow-based covariates on each day of the year equal to each of the historical values for that day of the year over the study period [22]. The flow-normalized estimate was calculated as the mean of all the predictions for each day considering all possible flow values. Standard deviations and credible intervals were obtained by drawing samples from the multivariate normal posterior distribution of the fitted GAM [28,35,36]. Uncertainty in loads were reported as 90% credible intervals developed by drawing 1000 realizations of parameter estimates from the multivariate normal posterior distribution of the model parameters. GAM performance was evaluated using repeated 5-fold cross validation [37] and average Nash-Suttclifee Efficiency (NSE), r^2 and percent bias (PBIAS) metrics across folds were calculated for each model.

2.3. Linking Estuary Water Quality to Hydrology and Nutrient Loads

To test if changes in freshwater inflow and nutrient loading had explanatory effect on changes in estuary water quality a series of GAM models were fit at each site relating parameter concentration to temporal trends, inflow, and nutrient loads [38]:

$$\begin{aligned} g(\mu) &= \alpha + f_1(ddate) + f_2(yday) \\ y &\sim \Gamma(\mu, \lambda), \end{aligned} \tag{5}$$

$$\begin{aligned} g(\mu) &= \alpha + f_1(ddate) + f_2(yday) + f_3(Q) \\ y &\sim \Gamma(\mu, \lambda), \end{aligned} \tag{6}$$

$$\begin{aligned} g(\mu) &= \alpha + f_1(ddate) + f_2(yday) + f_3(Q) + f_4(Load) \\ y &\sim \Gamma(\mu, \lambda), \end{aligned} \tag{7}$$

where μ is the conditional expected response (nutrient concentration), *g()* is the log link, and response variable was modeled as Gamma distributed with mean μ and scale λ. $f_1(ddate)$ is decimal date smoothed with a thin-plate regression spline, $f_2(yday)$ is the numeric day of year smoothed with a cyclic cubic regression spline, $f_3(Q)$ is mean daily inflow (the combined measurements from Lavaca River and Lake Texana) and $f_4(Load)$ is the total NO_3 or TP watershed load. The set of models specified for each water quality response are in Table **??**.

Table 3. Set of GAM models specified for each water quality parameter response.

Water Quality Response Parameter	Model	Model Terms
TP	Temporal	s(ddate) + s(yday)
	Flow	s(ddate) + s(yday) + s(Q)
	Flow+Load	s(ddate) + s(yday) + s(Q) + s(TP Load)
NO_x	Temporal	s(ddate) + s(yday)
	Flow	s(ddate) + s(yday) + s(Q)
	Flow+Load	s(ddate) + s(yday) + s(Q) + s(NO_3 Load)
Chlorophyll-*a*	Temporal	s(ddate) + s(yday)
	Flow	s(ddate) + s(yday) + s(Q)
	Flow+Load	s(ddate) + s(yday) + s(Q) + s(TP Load) + s(NO_3 Load)
Dissolved Oxygen	Temporal	s(ddate) + s(yday)
	Flow	s(ddate) + s(yday) + s(Q)
	Flow+Load	s(ddate) + s(yday) + s(Q) + s(TP Load) + s(NO_3 Load)
TKN	Temporal	s(ddate) + s(yday)
	Flow	s(ddate) + s(yday) + s(Q)

Because streamflow and nutrient loads are tightly correlated, freshwater inflow can mask signals from nutrient loads alone. Following the methodology implemented by Murphy *et al.* [38], both streamflow and nutrient loads were prepossessed to account for season and flow. Instead of using raw freshwater inflow and nutrient loading values, these values were replaced by seasonally adjusted inflow and flow-adjusted nutrient loads by fitting a GAM relating season (day of year) to log transformed daily freshwater inflow values:

$$g(\mu) = \alpha + f_1(yday), \tag{8}$$

and a GAM relating log transformed NO_3 or TP loads to log transformed daily inflow:

$$g(\mu) = \alpha + f_1(log(Q)), \tag{9}$$

where the response variables were modeled as normally distributed with an identity link function. Response residuals from the respective GAM models were used as Q and *Load* in Equation 6 and Equation 7.

3. Results

3.1. Watershed Nutrient Loads

Based on criteria provided by Moriasi *et al.* [39], GAMs ranged from “satisfactory” to “very good” using median NSE, r^2 and PBIAS calculated using repeated 5-fold cross validation on predicted and measured nutrient loads. NO_3 GAM models had median NSE values of 0.34 and 0.48 at USGS-08164000 and USGS-08164390 respectively. Median r^2 values were 0.70 (USGS-08164000) and 0.87 (USGS-08164525) and PBIAS values were 2.00 (USGS-08164000) and 10.90 (USGS-08164525) for NO_3 loads. Median NSE values were 0.80 (USGS-08164000) and 0.91 (USGS-08164525) for TP loads. Median r^2 values for TP loads were 0.93 (USGS-08164000) and 0.99 (USGS-08164525) and median PBIAS values were -7.20 (USGS-08164000) and -3.30 (USGS-08164525). Density plots of metrics show similar distribution of values between sites for the same parameter, with the exception r^2 values for NO_3 loads where USGS-08164000 showed a much larger variance in values compared to USGS-08164525 (Figure 2).In addition to higher average NSE and r^2 values, GAMs had smaller variance in metric values for TP compared to NO_3.

Annual NO_3 and TP loads show considerable variation, generally following patterns in discharge (Figure 3, Figure 4). Flow-normalized TP loads at both sites and flow-normalized loads at USGS-08164000 indicate watershed based loads have not changed much over time when accounting for changes in streamflow (Figure 3). Flow-normalized loads at USGS-08164000 show small variation over time with some decreases in NO_3 loads since 2013. Aggregated across both sites, the mean annual NO_3 load 2005 through 2020 was 205,405 kg (126,867 kg - 341,569 kg, 90% CI). Annual NO_3 loads ranged from 12,574 kg in 2011 to 794,510 kg in 2007.

Total annual TP loads ranged from 7,839 kg in 2011 to 595,075 kg in 2007. Mean annual TP loading from 2005 through 2020 was 182,673 kg (152,227 kg - 219,310 kg, 90% CI). On average, USGS-08164525 accounted for 68% of NO_3 loads and 59% of TP loads from 2005 through 2020. However, during periods of extreme drought the Lavaca River (USGS-08164000) became the primary source of nutrient loading in the watershed with the Navidad River only accounting for 15% and 25% of NO_3 and TP loads in 2011 (Figure 4).

3.2. Linkages Between Water Quality and Watershed Flows and Loads

GAMs did not identify significant changes in TP or DO concentrations at any of the Lavaca Bay sites from 2005 through 2020 (Figure 5). The upper-bay site, TCEQ-13563, had a linear increase in NO_x concentration and and decrease in chlorophyll-*a* from 2005 through 2014. The mid-bay site, TCEQ-13383, showed a periodic pattern in NO_x concentration that appeared similar to precipitation/inflow patterns, as well as a post 2011 increase in TKN concentrations. No significant long-term trends in concentrations were identified by GAMs for the lower-bay TCEQ-13384 site.

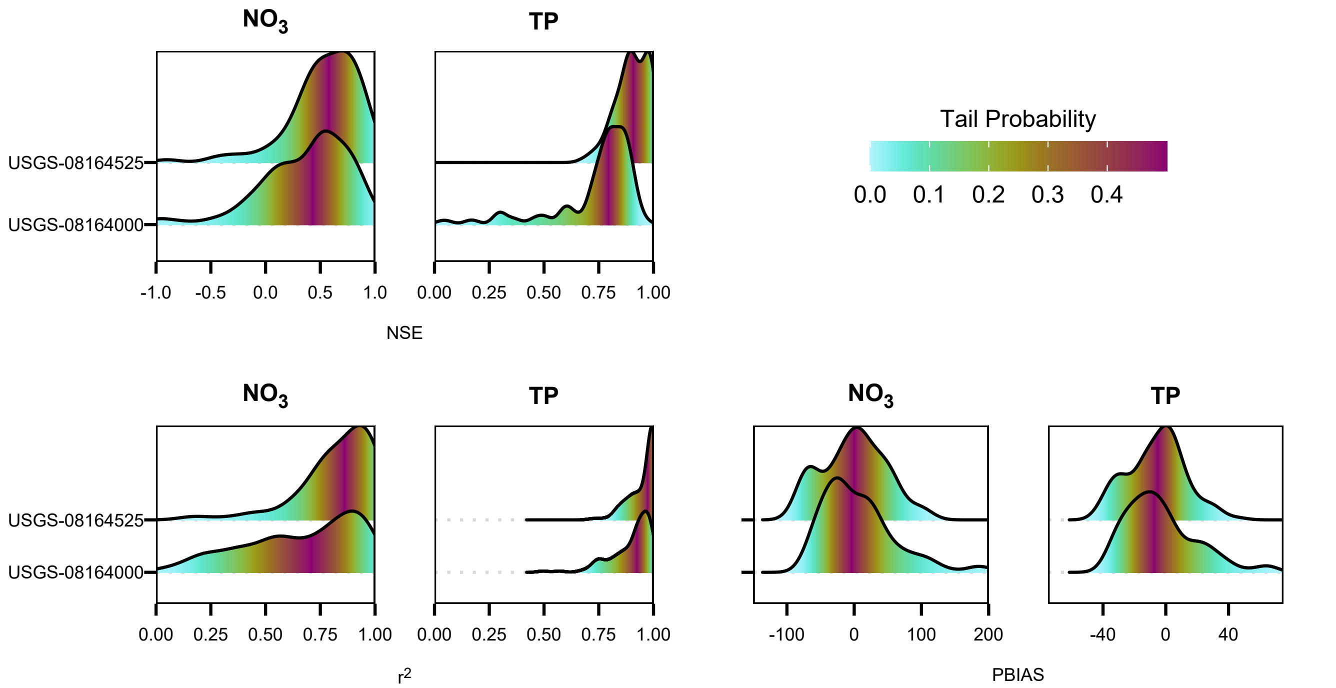

Figure 2. Density plots of goodness-of-fit metrics (NSE, r^2, and PBIAS) from repeated 5-fold cross validation between predicted nutrient loads from GAM models and measured nutrient loads. Color indicates the tail probability calcualted from the empirical cumulative distribution of the goodness-of-fit metrics.

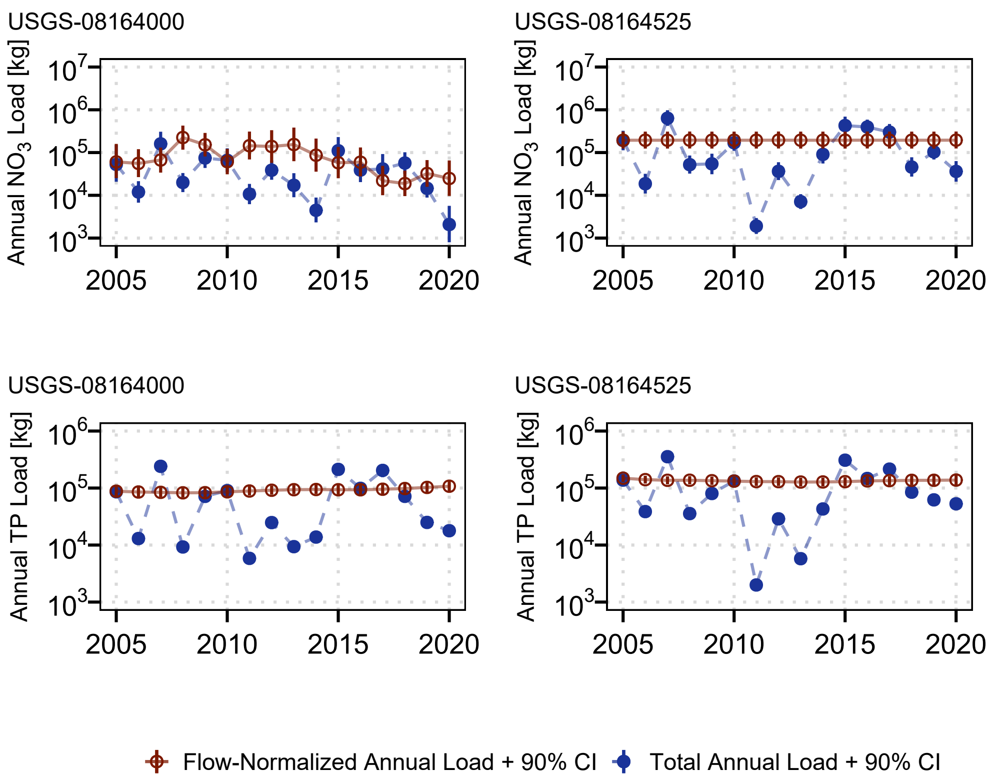

Figure 3. Aggregated estimated annual and flow-normalized annual NO_3 and TP loads for USGS-08164000 and USGS-08164525.

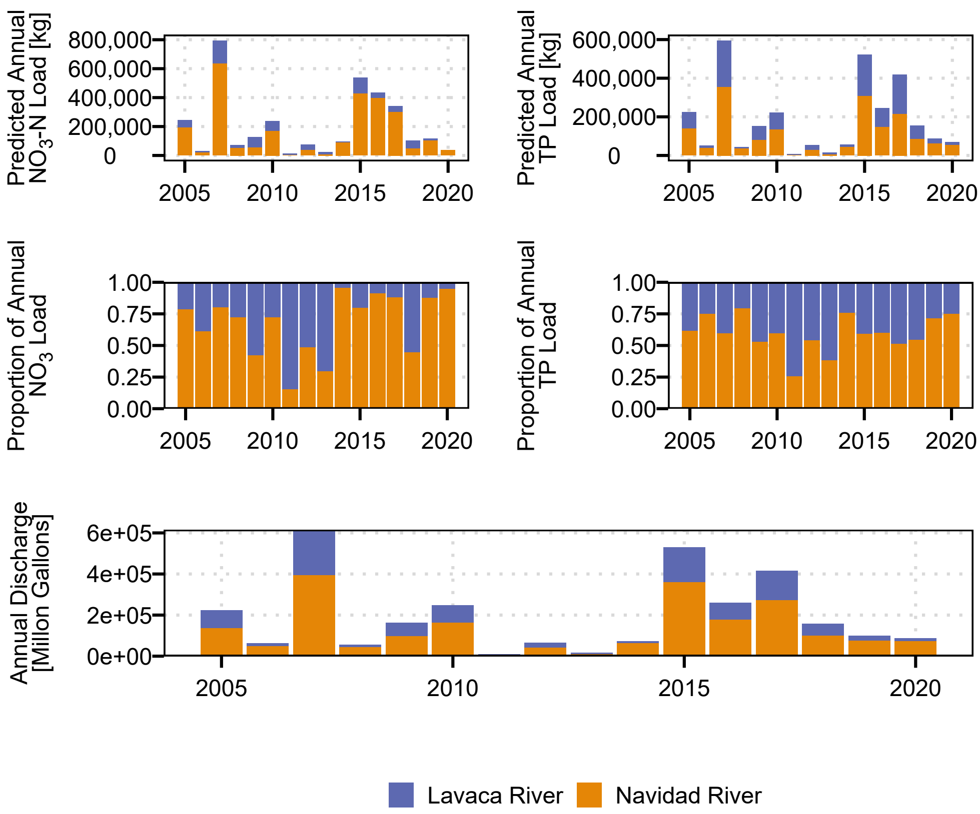

Figure 4. Comparison of delivered annual loads at USGS-08164000 and USGS-08164525.

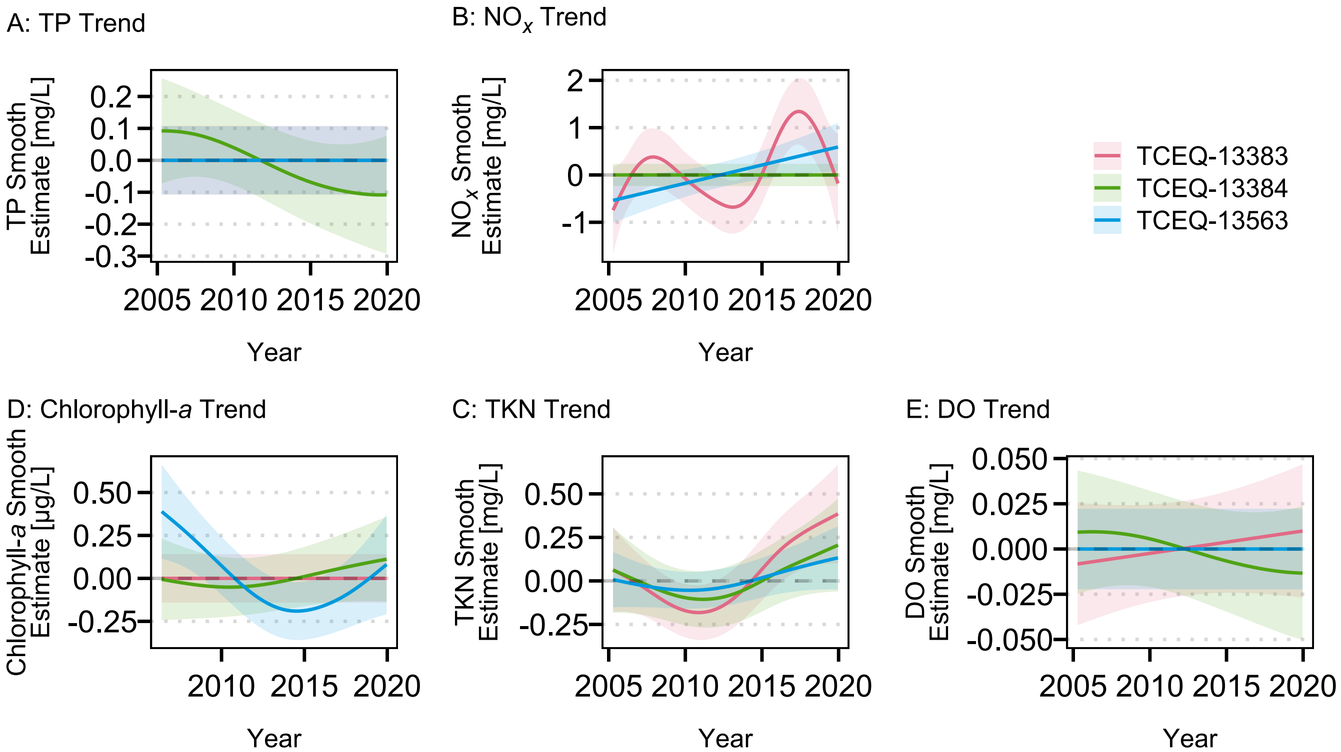

Figure 5. Smoothed temporal trend component for water quality paramaters obtained from temporal estuary GAMs.

Freshwater inflow provided additional explanation for changes in TP and NO_x concentration at all of the Lavaca Bay sites according to AIC_c and model probability values (Table 4). TCEQ-13563, the site closest to the river outlet, was the only site that had improvements in the explanations of DO and TKN concentration with the inclusion of inflow. Both TCEQ-13563 and TCEQ-13383, the mid-bay site, saw improvements in explanations for variations in chlorophyll-*a* with the inclusion of freshwater inflow. The addition of nutrient loads (both TP and NO_3) terms did not provide additional explanation for changes in chlorophyll-*a* or DO concentrations. Inclusion of TP loads provided additional explanation of TP concentrations at the upper- and mid-bay sites, TCEQ-13563 and TCEQ-13383. Inclusion of NO_3 loads only provided marginal improvements in the explanation of NO_X concentration at the lower-bay TCEQ-13384 site.

Table 4. Estuary GAM AIC_c values and associated model probabilities (in parenthesis). Models with the highest probability for each site and water quality parameter combination are bolded and italicized for emphasis.

Parameter	Site	Temporal	Flow	Flow + Load
TP	TCEQ-13383	-152.1 (0.03)	-156.1 (0.24)	***-158.2 (0.72)***
	TCEQ-13384	-194.4 (0.03)	***-200.2 (0.49)***	-200.2 (0.49)
	TCEQ-13563	-145.3 (0)	-156.6 (0.41)	***-157.3 (0.59)***
NO_x	TCEQ-13383	-218.9 (0)	***-244.8 (0.5)***	-244.8 (0.5)
	TCEQ-13384	-263.4 (0)	-311.7 (0.48)	***-311.9 (0.52)***
	TCEQ-13563	-175.1 (0)	***-190.2 (0.5)***	-190.2 (0.5)
Chlorophyll-*a*	TCEQ-13383	279.7 (0.18)	***278.1 (0.41)***	278.1 (0.41)
	TCEQ-13384	***268.2 (0.33)***	268.2 (0.33)	268.2 (0.33)
	TCEQ-13563	289.5 (0.08)	***286.1 (0.46)***	286.1 (0.46)
TKN	TCEQ-13383	***42.2 (0.66)***	43.5 (0.34)	-
	TCEQ-13384	***34.3 (0.57)***	34.8 (0.43)	-
	TCEQ-13563	31.1 (0.22)	***28.7 (0.78)***	-
DO	TCEQ-13383	***146.4 (0.34)***	146.4 (0.34)	146.5 (0.32)
	TCEQ-13384	***135.9 (0.47)***	137 (0.27)	137 (0.27)
	TCEQ-13563	138.3 (0.25)	***137.2 (0.43)***	137.8 (0.32)

GAMs show that increases in freshwater inflow result in nearly linear increases in TP and NO_x at all three sites (Figure **??**). At the upper-bay TCEQ-13563 site, GAMs showed increases in freshwater inflow initially increased chlorophyll-*a* and DO concentration but concentrations leveled and potentially decreased at higher flows. The mid-bay TCEQ-13383 site showed a nearly linear increased in chlorophyll-*a* concentration with increases in freshwater inflow. Freshwater flow did not have significant effects on chlorophyll-*a*, TKN, or DO at the lower-bay TCEQ-13384 site.

Increases in TP loads resulted in nearly linear increases of TP concentration at the upper- and mid-bay sites, TCEQ-13563 and TCEQ-13383 respectively (Figure 7). The relative effect size appeared to much smaller than the effect of freshwater inflow alone. Increases in NO_3 loads only showed an effect at the lower-bay TCEQ-13384 site. The effect was quite small compared to streamflow and provided only small improvements to the model (Table 4). As noted above, nutrient loadings did not provide any explanation in changes in the remaining assessed water quality parameters.

4. Discussion

TP and NO_3 loadings from the Lavaca Bay watershed showed high inter-annual variability tied with changes in discharge. There is little evidence for changes in flow-normalized TP loads in either rivers. There is some evidence of recent decreases in flow-normalized NO_3 loads in the Lavaca River. Although there is no work directly correlating

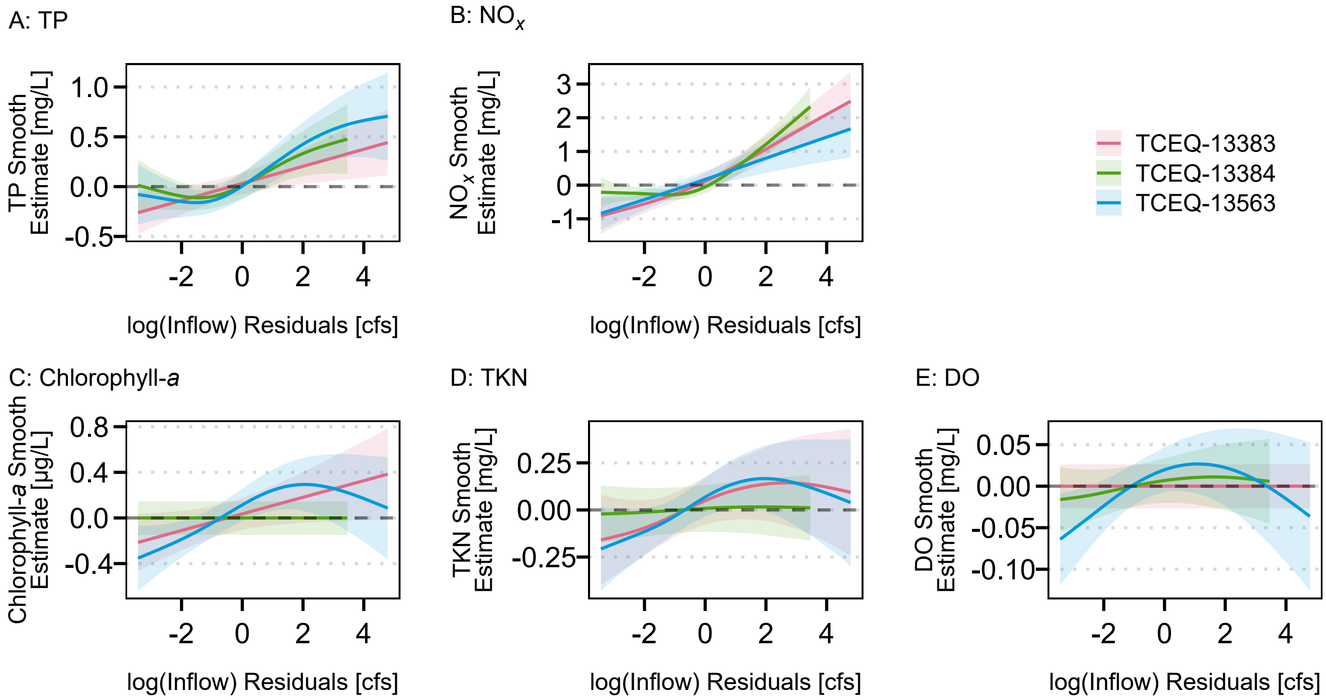

Figure 6. Estimated effects of mean daily inflow residuals on mean TP, NO_x, chlorophyll-*a*, TKN, and DO concentrations in Lavaca Bay obtained from flow estuary GAMs.

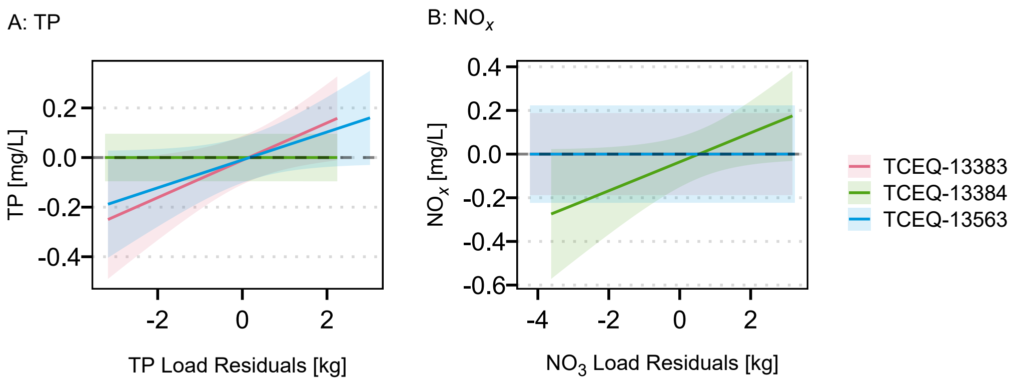

Figure 7. Estimated effects of nutrient load residuals on TP and NO_x concentrations in Lavaca Bay obtained from flow+load estuary GAMs.

water quality planning and implementation efforts in the watershed to water quality outcomes, efforts to increase agricultural producer participation in the watershed have been ongoing since 2016 [15,16]. The decrease in flow-normalized NO_3 loads could be a reflection of those collective efforts but further data collection and research is required to support that statement.

Other studies have published estimates of TP loading estimates for the Lavaca River (USGS-08164000) over various time periods (Table 5). Converted to average annual yield, the estimates of TP loading in this study are within the ranges of previous studies [40–43], with the TP estimates in Dunn [40] being notably lower. Given that none of the studies identify substantially sized trends in TP, it is is possible that the period used in Dunn [40] was drier on average than the other studies. The SPARROW models used in Rebich *et al.* [41] and Wise *et al.* [43] utilize a version of LOADEST in the underlying load estimation procedure, so a difference due to methodology alone is unlikely.

Table 5. Mean estimates of annual TP yield in the Lavaca River watershed in published studies.

Parameter	Reported Yield ($kg \cdot km^{-2} \cdot year^{-1}$)	Approach	Time Period	Reference
TP	35.2 (28.8, 43.3)*	GAM	2005-2020	This work
TP	45.2	SPARROW	2000-2014	Wise *et al.* [43]
TP	42	SWAT	1977-2005	Omani *et al.* [42]
TP	20.81-91.58†	SPARROW	1980-2002	Rebich *et al.* [41]
TP	28.9	LOADEST	1972-1993	Dunn [40]

* Values represent the mean of annual point estimates, lower and upper 95% credible intervals.
† A single point estimate was not reported, these value represent the range depicted on the choropleth map provided in the report.

Cross-validation of the GAM loading models indicated that GAMs performed well on average at predicting daily nutrient loading values. The variance in scores was very high indicating subsets of values were problematic at characterizing functional relationships between nutrients and predictors. All of the water quality data for the two river sites in the TCEQ database were ambient water quality data, collected to be representative of typical conditions. This results in few data collected at the highest portions of the flow duration curve under which the majority of loadings take place. Supplementary flow-biased monitoring targeting storm- or high-flow conditions is recommended here to improve the precision of GAM predictions [44,45].

The non-linear temporal water quality trends identified using GAMs differed slightly from the trends identified by **?**]. This is not unexpected due to the different time periods, different methodology, and generally small slopes identified for most of the significant water quality parameters in Bugica *et al.* [3]. The trend in DO and cholorophyll-*a* concentrations are stable in comparison to other Texas estuaries that might be facing larger demands for freshwater diversions, higher population growth, and more intense agricultural production [3,4]. The trend of increasing NO_x concentration at the upper-bay TCEQ-13563 site and recent increases in TKN concentration at the mid-bay TCEQ-13383 site are concerning due to the nitrogen limitation identified in many Texas estuaries [5,10–13] and the relatively low ambient concentrations observed in Lavaca Bay.

The strong positive effect of freshwater inflow on NO_x, TKN, and TP are suggestive of nonpoint watershed sources, consistent with watershed uses, and consistent with other studies relating freshwater inflow with nutrient concentrations in Lavaca Bay and other estuaries [8,46–49]. Inflow had a non-linear relationship with TKN at the two upstream sites, with TKN increasing as freshwater inflow transitioned from low to moderate levels. At higher freshwater inflows, the effect was attenuated, possibly indicating a flushing effect at higher freshwater inflow. No relationship between TKN and freshwater inflow were observed at TCEQ-13384 located in the lower reach of Lavaca Bay. Tidal flushing from Matagorda Bay might be responsible for decreasing TKN concentrations and limiting the effects of freshwater inflow in lower reaches of Lavaca Bay. Russell *et al.* [46] suggested the processing of organic loads in the upper portions of Lavaca Bay reduces the transport of nutrients into the lower reaches of the Bay.

Freshwater inflow also had a strong positive effect on chlorophyll-*a* at the upper- and mid-bay sites. The upper-bay site, TCEQ-13563, showed decreases in chlorophyll-*a* at the highest freshwater inflow volumes. Freshwater flushing or increases in turbidity are associated with decreases in chlorophyll-*a* in other estuaries [48,50]. No relationships between inorganic nitrogen or TP loadings with chlorophyll-*a* were observed. Due to the lack of TKN loading information, no assessment between organic nitrogen loads and chlorophyll-*a* were possible.

Although other studies have identified complex relationships between estuary nutrient concentrations, nutrient loading and chlorophyll-*a* concentrations in Texas estuaries [12,49,51**?**], this study specifically used flow-adjusted freshwater derived nutrient loads to parse out contributions from changes in nutrient loadings while accounting for variations in load due to flow. Loading GAMs indicated no evidence of changes in flow-normalized TP loads in either river, and no changes in flow-normalized NO_3 loads in the Navidad River. The small changes in flow-normalized NO_3 loads in the Lavaca River are probably masked under most conditions by discharge from the Navidad River. Given the relatively little variation in flow-normalized loads, it can be expected that they would contribute little to the variance in downstream water quality.

GAMs did not identify responses in DO concentration to inflows or nutrient loads. The seasonality term in the temporal GAM models explained a substantial amount of DO variation at all of the sites. Responses of estuary metabolic processes and resulting DO concentrations can be quite complicated and often locally specific [**?**]. While the lack of total nitrogen or TKN loading data hinders interpretation, the large seasonal effect on DO suggests physical factors play an important role and should be included in future models. Prior work suggests that there is some evidence the Lavaca Bay may not be limited by nutrients alone, with high turbidity or nutrient processing in upper portions of the Bay or intertidal river limiting production [46].

5. Conclusion

This section is not mandatory, but can be added to the manuscript if the discussion is unusually long or complex.

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