Assessing linkages between watershed nutrient loading and water quality in a subtropical estuary with semiparametric models.

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Abstract

Lavaca Bay is a small secondary embayment on the Texas coast that is displaying early signals of water quality degradation. This study applied a semiparametric approach to assess both watershed nutrient loads and the responses in estuary water quality to nutrient loading and streamflow. Cross-validation indicated that, despite data constraints, semiparametric models performed well at nutrient load prediction. Based on these models, delivered annual nutrient loads varied substantially from year to year. In contrast, minimal changes in calculated flow-normalized loads indicate that changes in nutrient loading were driven by natural variation in precipitation and runoff as opposed to changes in nonpoint sources. Estuary water quality models did not identify significant long-term changes within Lavaca Bay for dissolved oxygen or chlorophyll-*a*. However, site specific long-term increases in both organic and inorganic nitrogen are concerning. Further analysis found freshwater inflow was a strong driver of nutrient and chlorophyll-*a* concentrations within Lavaca Bay but there was no evidence that changes in nutrient loading explained variation in dissolved oxygen or chlorophyll-*a* concentrations. In addition to providing a baseline assessment of watershed nutrient loading and water quality responses in the Lavaca Bay watershed, this study provides methodological support for the use of semiparametric methods in load regression models and estuary assessments.

Keywords: key, dictionary, word

1 Introduction

Like many coastal areas globally, the 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 degrade water quality in downstream estuaries (Bricker et al, 2008; Kennicutt, 2017; Bugica et al, 2020). Despite these escalating pressures, national scale assessments have classified coastal estuaries in Texas as moderate or low risk for eutrophic conditions (Bricker et al, 2008). However, a suite of recent studies indicates 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 (Wetz et al, 2016, 2017; Bugica et al, 2020; Chin et al, 2022). With identification of localized areas of estuary water quality concern along the Texas coast (Bugica et al, 2020), localized studies are being prioritized to better inform management actions.

This project aims to provide 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 faces 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. More recently identified long-term declines in abundance, biomass, and diversity of benthic fauna in Lavaca Bay have been linked to reductions in freshwater inflows and changes in estuary salinity (Beseres Pollack et al, 2011; Palmer and Montagna, 2015; Montagna et al, 2020) and are a concern to local stakeholders. Water quality assessments identified monotonic increases in total phosphorus (TP), orthophosphate, total Kjeldahl nitrogen (TKN), and chlorophyll-a at sites within Lavaca Bay (Bugica et al, 2020). Although long-term changes in DO concentrations were not 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 (Gardner et al, 2006; Hou et al, 2012; Dorado et al, 2015; Paudel et al, 2019; Wetz et al, 2017) and the ramifications that 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 (Jain and Schramm, 2021; Schramm et al, 2018; Berthold et al, 2021). 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 downstream water quality to identify and replicate effective management actions across the state (Schramm et al, 2022). 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 (Wazniak et al, 2007; Lloyd et al, 2014). The development and application of statistical methods such as Weighted Regressions on Time, Discharge and Season (WRTDS, Hirsch et al, 2010) and Generalized Additive Models (GAMs, Wood, 2011) has provided effective tools for researchers to quantify and communicate non-linear changes in river and estuary pollutant loadings.

WRTDS calculates a time series of in stream concentrations or loads (daily, monthly, or annually) and flow-normalized estimates of concentrations and loads using locally weighted regresssion for unique combinations of time, discharge, and season. WRTDS has been widely used to assess and identify trends in riverine nutrients (Oelsner and Stets, 2019; Stackpoole et al, 2021), chlorides (Stets et al, 2018), and other pollutants of concern (Shoda et al, 2019). WRTDS has also been succesfully adapted to assess trends in estuarine water quality concentrations (Beck et al, 2018).

While WRTDS is a statistical approach developed specifically for water quality applications, GAMs are a broadly applicable statistical method. 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 (Wood, 2011). Although the underlying parameter estimation procedure of GAMs is substantially different than WRTDS, both the functional form and results have been demonstrated to be similar (Beck and Murphy, 2017). Water quality applications of GAMs include river and catchment nutrient concentration and load models (Wang et al, 2011; Kroon et al, 2012; Kuhnert et al, 2012; Robson and Dourdet, 2015; Hagemann et al, 2016; McDowell et al, 2021; Biagi et al, 2022). GAMs can also be used to identify non-linear temporal trends (including flow-normalized trends) in pollutant concentrations and loads (Beck and Murphy, 2017; Murphy et al, 2019). Recently GAMs have also been used to link water quality responses in receiving water bodies to changes in nutrient inputs (Murphy et al, 2022). Beck and Murphy (2017) provides a substantial discussion on the differences (and similarities) between GAMs and WRTDS for water quality applications.

To provide actionable information for resource managers in Lavaca Bay, water quality conditions must be evaluated relative to changes in natural environmental drivers to better understand and manage potential human impacts. This study utilizes GAMs to develop estimates of delivered and flow-normalized nutrient loads and assess changes in loads delivered to Lavaca Bay. GAMs were chosen over other regression-based approached for use in this study due to; (1) the ability to easily explore and incorporate different model terms; (2) the incorporation of non-liner smooth functions that do not require explicit a priori knowledge of the expected shape; and (3) inclusion of a link function that related the expected value of the response to linear predictors thus avoiding unneeded data transformations and bias corrections. The study also assesses the response of water quality parameters in Lavaca Bay over time and in response to freshwater inflow controlled for seasonality and to watershed nutrient loads that are controlled for environmentally driven variation.

2 Methods

2.1 Location and Data

Lavaca Bay is 190 km^2 with the majority of freshwater inflow provided by the Lavaca and Navidad River systems (Figure 1). The Garcitas-Arenosa, Placedo Creek, and Cox Bay watersheds provide additional freshwater inflows. The entire watershed land area is 8,149 km^2 and primarily rural. Watershed land cover and land use is 50% grazed pasture and rangeland, 20% cultivated cropland (primarily rows crops such as corn, cotton, and sorghum), and 5% suburban/urban. Pasture and rangeland is concentrated in the Lavaca River watershed, while cultivated crops are generally located along the eastern tributaries of the Navidad river. 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 0.210 km^3 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 Lavaca Bay.

Daily discharges for the Lavaca River (USGS-08164000, Figure 1) were obtained from the United States Geologic Survey (USGS) National Water Information System using the *dataRetrieval* R package (De Cicco et al, 2022). 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

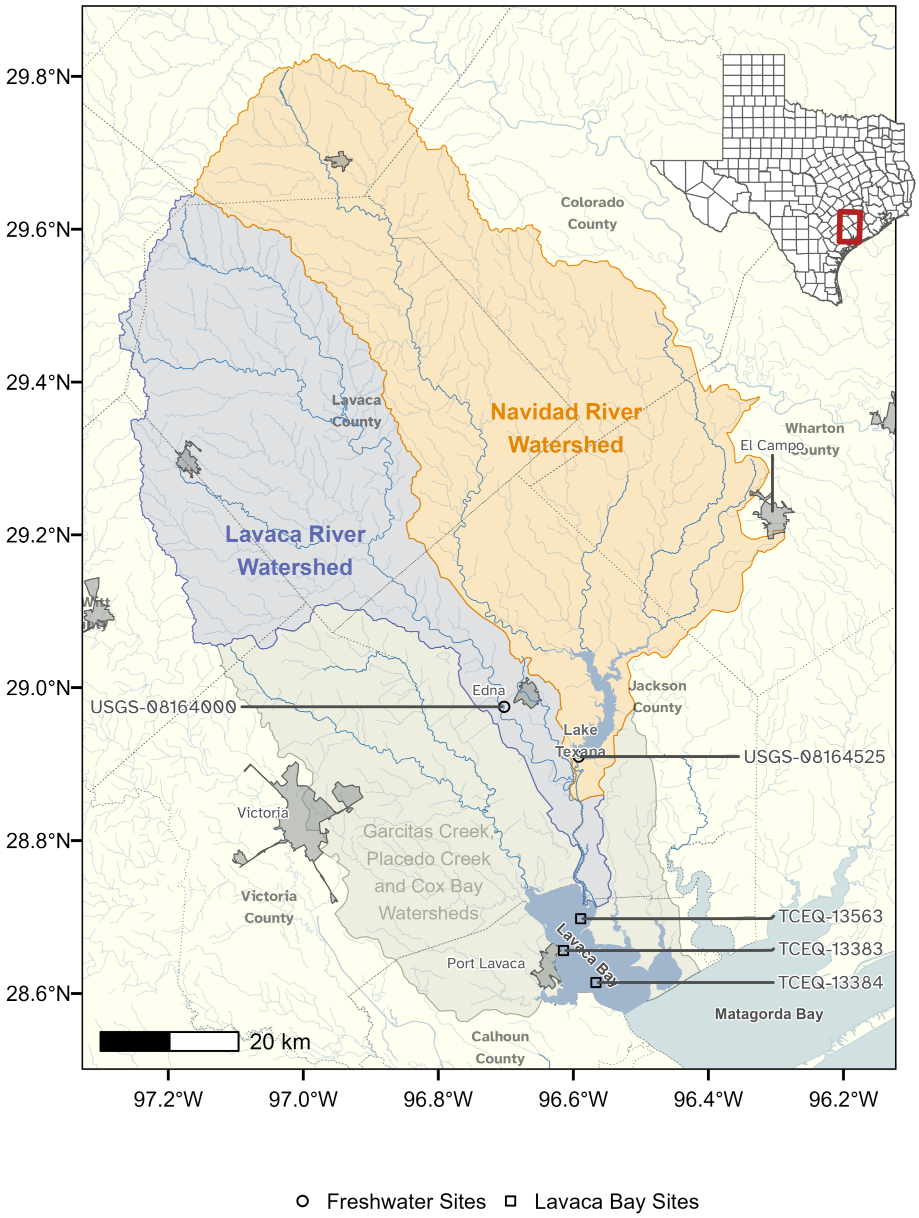

Fig. 1 Map of Lavaca Bay and the contribution watershed. The freshwater sites are the most downstream freshwater stream locations with water quality and streamflow data used for nutrient load models. Water quality concentration data at the three Lavaca Bay sites were used to assess relationships between freshwater flows, loads and estuary water quality.

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
	NO3 (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
	NO3 (mg/L)	0.29	0.26	62
	Mean Daily Streamflow (cfs)	666.14	2957.79	7671

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 (Horowitz, 2003; Snelder et al, 2017). 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).

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-08164525) were developed using 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 were fit using the *mgcv* package in R which

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
	NOx (mg/L)	0.07	0.15	51
	TKN (mg/L)	0.94	0.49	45
	Chlorophyll-a (ug/L)	9.43	5.31	47
	DO (mg/L)	7.22	1.35	55
TCEQ-13384	TP (mg/L)	0.08	0.03	51
	NOx (mg/L)	0.06	0.08	52
	TKN (mg/L)	0.76	0.40	48
	Chlorophyll-a (ug/L)	8.22	6.44	46
	DO (mg/L)	7.51	1.32	54
TCEQ-13563	TP (mg/L)	0.13	0.06	50
	NOx (mg/L)	0.09	0.13	53
	TKN (mg/L)	0.94	0.37	49
	Chlorophyll-a (ug/L)	9.67	5.33	49
	DO (mg/L)	7.91	1.34	56

makes available multiple types of smooth functions with automatic smoothness selection (Wood, 2011). The general form of the model related NO_3 or TP concentration to a long term tend, season, streamflow, and two different antecedent discharge terms, shown in Eq. 1:

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

where μ is the conditional expected NO_3 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, 2011; Kuhnert et al, 2012; Zhang and Ball, 2017), using Eq. 2:

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

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 (Vecchia et al, 2009; Zhang and Ball, 2017). Long-term flow anomaly (*ltfa*) is the streamflow over the previous year relative to the entire period (Zhang and Ball, 2017) and calculated using Eq. 3:

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

and the short-term flow anomaly (*stfa*) calculated as the current day flow compared to the preceding 1-month streamflow using 4:

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

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 previous work demonstrating major improvements in NO_x regression models that incorporated *ltfa* and moderate improvements in TP regression models that incorporated *stfa* (Zhang and Ball, 2017).

The calculation of model terms for the Lake Texana site were 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. Thin-plate 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 (Hornung and Reed, 1990), 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 (McDowell et al, 2021).

Daily loads were estimated as the predicted concentration multiplied by the daily streamflow. For the Navidad River (USGS-08164525) site, daily 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 (Hirsch et al, 2010). 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 (Wood, 2006; Marra and Wood, 2012; McDowell et al, 2021). Uncertainty in loads were calculated as 90% credible intervals estimated 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 (Burman, 1989) and average Nash-Sutcliffe

Efficiency (NSE), Pearson sample correlation (r) 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 (Eq. 5), temporal trends and inflow (Eq. 6), and temporal trends, inflow, and nutrient loads (Eq. 7):

$$g(\mu) = \alpha + f_1(ddate) + f_2(yday), \tag{5}$$

$$g(\mu) = \alpha + f_1(ddate) + f_2(yday) + f_3(Q), \tag{6}$$

$$g(\mu) = \alpha + f_1(ddate) + f_2(yday) + f_3(Q) + f_4(Load) \tag{7}$$

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 Navidad River) 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 3.

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 (2022), both streamflow and nutrient loads were prepossessed to account for season and flow. Freshwater inflow and nutrient loads were replaced by seasonally adjusted log transformed inflow and flow-adjusted log transformed nutrient loads obtained by fitting a GAM relating season (day of year) to log transformed daily

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

freshwater inflow values and a GAM relating log transformed NO_3 or TP loads to log transformed daily inflow.Response residuals from the respective GAM models were used as Q and $Load$ in Eq. 6 and Eq. 7.

This study used an information theoretic approach to evaluate if nutrient loads and/or freshwater inflows provided evidence of effects on water quality concentrations in Lavaca Bay. Model probabilities were calculated and compared using the AIC_c scores between each group of temporal, flow, and flow+load models (Burnham et al, 2011). Improvements in model probabilities provide evidence that the terms explain additional variation in the response variable. If model probabilities were tied, there wasn't evidence the more complicated model explains additional variation in water quality.

3 Results

3.1 Watershed Nutrient Loads

Using evaluation criteria recommended by Moriasi et al (2015), predictive performance of nutrient loading GAMs ranged from "satisfactory" to "very good" based on

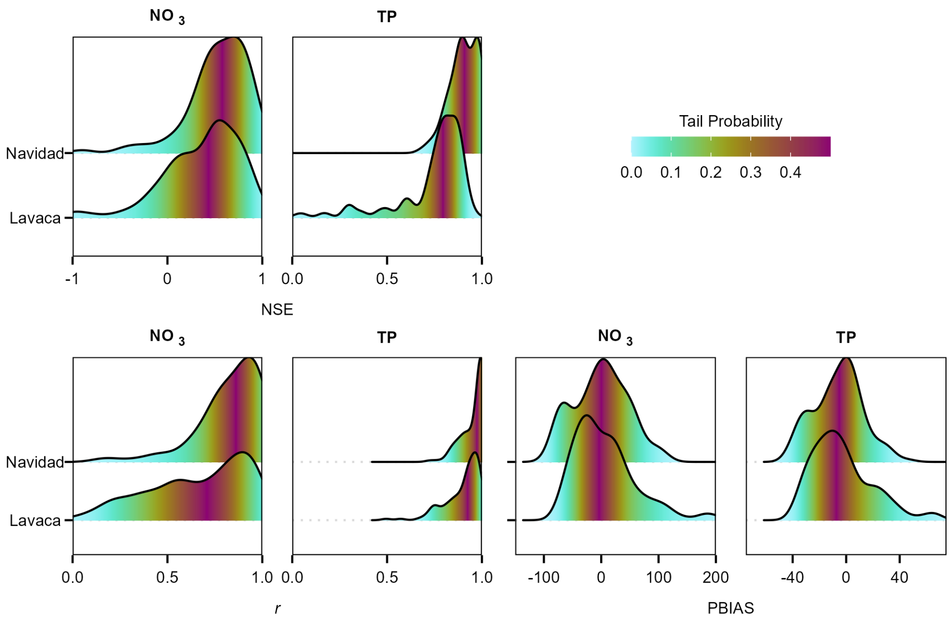

Fig. 2 Density plots of goodness-of-fit metrics (NSE, r, and PBIAS) from repeated 5-fold cross validation between predicted nutrient loads from GAM models and measured nutrient loads. Color indicates the tail probability calculated from the empirical cumulative distribution of the goodness-of-fit metrics.

median NSE, r, and PBIAS metrics calculated using 5-fold cross validation. Median goodness-of-fit metrics for NO_3 models in the Lavaca River were 0.34 NSE, 0.70 r, and 2.00 PBIAS. Navidad River NO_3 models appeared to perform slightly better with 0.48 NSE and 0.87 r but with higher bias at 10.90 PBIAS. Generally, TP models performed better than NO_3 models. Median goodness-of-fit metrics for TP in the Lavaca River were 0.81 NSE, 0.93 r, and -7.20 PBIAS. Navidad River TP models has similar performance with 0.91 NSE, 0.99 r, and -3.30 PBIAS. Density plots of metrics show similar distribution of values between sites for the same parameter, with the exception r values for NO_3 loads where Lavaca River had a much larger variance in values compared to the Navidad River (Figure 2). TP GAMS had higher average NSE and r values and lower variance in metric values compared to NO_3.

Predicted annual NO_3 and TP loads show considerable variation, generally following patterns in discharge (Figures 3, 4). Flow-normalized TP loads at both sites

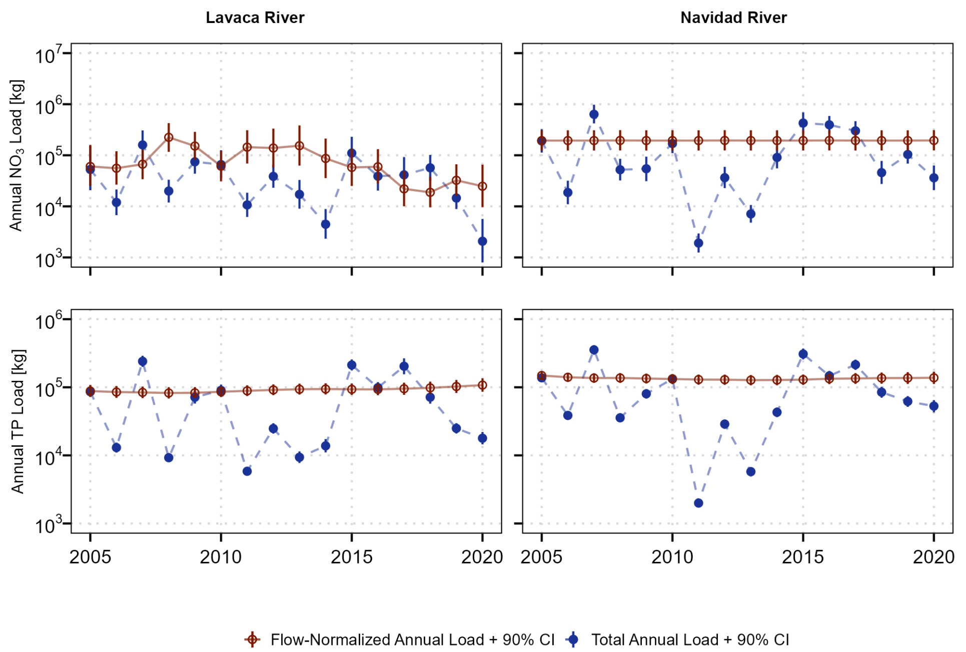

Fig. 3 Aggregated estimated annual and flow-normalized annual NO 3 and TP loads for USGS-08164000 and USGS-08164525.

and flow-normalized NO_3 loads in the Lavaca River indicated watershed based loads did not change much over time when accounting for variation driven by streamflow (Figure 3). Flow-normalized loads in the Lavaca River showed 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, the Navidad River 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 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).

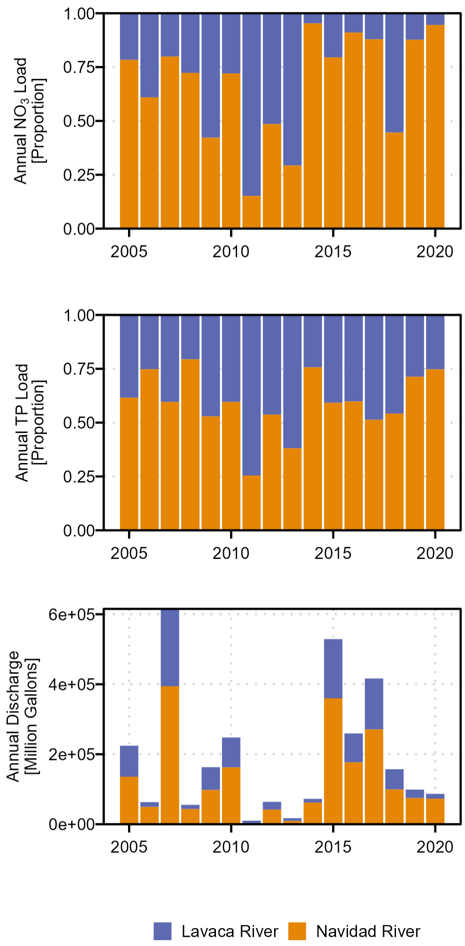

Fig. 4 Comparison of delivered annual loads and annual discharge at the Lavaca (USGS-08164000) and Navidad (USGS-08164525) Rivers.

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

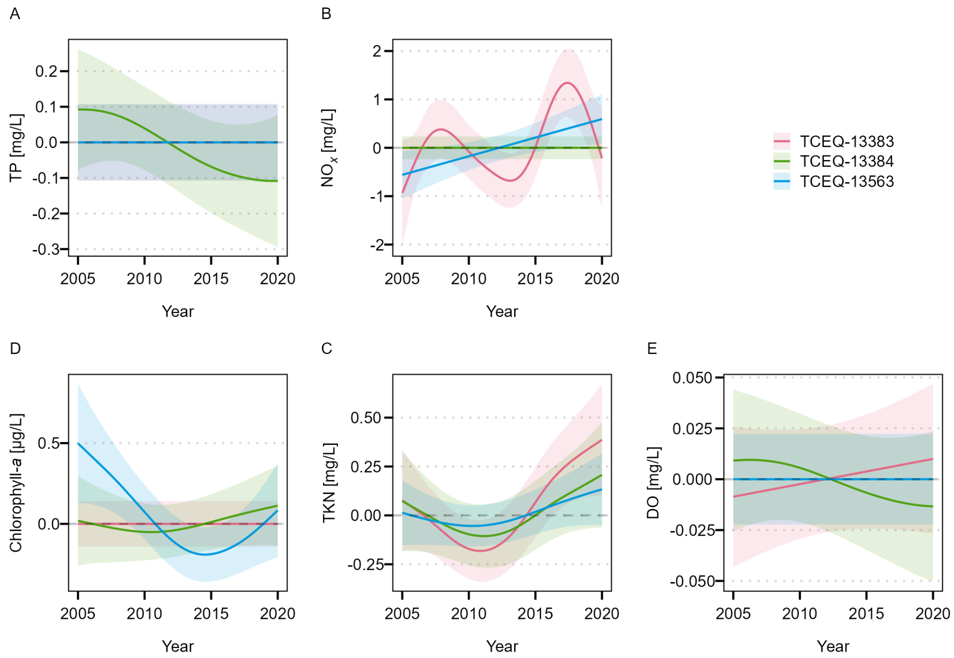

Fig. 5 Smoothed temporal trend component for water quality paramaters obtained from temporal estuary GAMs.

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.

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

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)

only provided marginal improvements in the explanation of NO_X concentration at the lower-bay TCEQ-13384 site.

GAMs showed that increases in freshwater inflow resulted in nearly linear increases in TP and NO_x concentration at all three sites (Figure 6). At the upper-bay TCEQ-13563 site, GAMs showed that 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 in response to increases freshwater inflow. Freshwater flow did not have significant effects on chlorophyll-a, TKN, or DO at the lower-bay TCEQ-13384 site.

Increased 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. Increased NO_3 loads only showed an effect at the lower-bay TCEQ-13384 site.

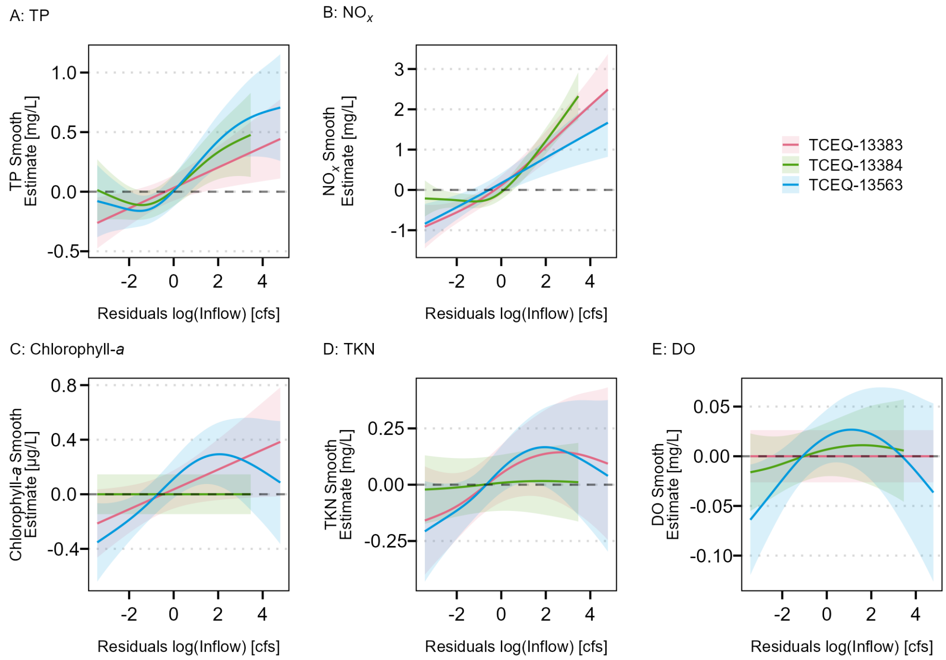

Fig. 6 Estimated effects of mean daily inflow residuals on mean TP, NO *x* , chlorophyll-*a*, TKN, and DO concentrations in Lavaca Bay obtained from flow estuary GAMs.

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

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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 water quality planning and implementation efforts in the watershed to water quality outcomes, efforts to increase agricultural producer participation in the

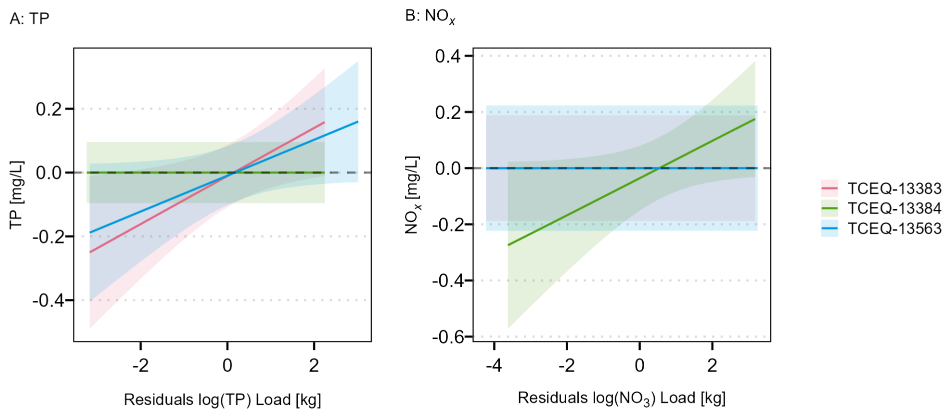

Fig. 7 Estimated effects of nutrient load residuals on TP and NO *x* concentrations in Lavaca Bay obtained from flow+load estuary GAMs.

watershed have been ongoing since 2016 (Schramm et al, 2018; Berthold et al, 2021). 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.

Converted to average annual yield, the estimates of annual TP loads for the Lavaca River are within the ranges in previous published studies (Table 5, Dunn, 1996; Rebich et al, 2011; Omani et al, 2014; Wise et al, 2019). It isn't obvious why TP estimates in Dunn (1996) were notably lower. Given that none of the studies identify substantially sized trends in TP, it is is possible that the period used in Dunn (1996) was drier on average than the other studies. The SPARROW models used in Rebich et al (2011) and Wise et al (2019) utilize a version of LOADEST in the underlying load estimation procedure, so a difference due to methodology alone is unlikely.

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. Because all of the water quality data for these two locations in the TCEQ databases were ambient water quality data, collected to be representative of typical flow conditions, there were few data at the highest

Table 5 Comparisons of previously published estimates of mean annual TP yield at the Lavaca River site.

Reported Yield (kg·km^2·year^{-1})	Approach	Time Period	Reference
35.2 (28.8, 43.3)[a]	GAM	2005-2020	This work
45.2	SPARROW	2000-2014	Wise et al (2019)
42	SWAT	1977-2005	Omani et al (2014)
20.81-91.58[b]	SPARROW	1980-2002	Rebich et al (2011)
28.9	LOADEST	1972-1993	Dunn (1996)

[a] Mean of the annual point estimates and the lower and upper 95% credible intervals.

[b] Represents a binned value range from a choropleth map.

portions of the flow-duration curve. It was beyond the scope of the current study to evaluate the subsets of cross-validation data and scores. However, the cross-validation procedure is indicative that more robust sampling would be beneficial for reducing prediction variance. Supplementary flow-biased monitoring targeting storm- or high-flow conditions is recommended here to improve the precision of GAM predictions (Horowitz, 2003; Snelder et al, 2017).

The non-linear temporal water quality trends identified using GAMs differed slightly from previously identified trends (Bugica et al, 2020). 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 prior work. The trend in DO and cholorophyll-a concentrations are stable in comparison to other Texas estuaries that are facing larger demands for freshwater diversions, higher population growth, and more intense agricultural production (Wetz et al, 2016; Bugica et al, 2020). 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 (Gardner et al, 2006; Hou et al, 2012; Dorado et al, 2015; Paudel et al, 2019; Wetz et al, 2017) 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 with other studies relating freshwater inflow with nutrient concentrations in Lavaca Bay and other estuaries (Russell et al, 2006; Caffrey et al, 2007; Peierls et al, 2012; Palmer and Montagna, 2015; Cira et al, 2021). 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 could be responsible for diluting TKN and acting as a control on the effects of freshwater inflow in lower reaches of Lavaca Bay. Previous work suggests the processing of organic loads in the upper portions of Lavaca Bay might reduce the transport of nutrients into the lower reaches of the Bay (Russell et al, 2006).

Freshwater inflow 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 (Peierls et al, 2012; Cloern et al, 2014). 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 (Örnólfsdóttir et al, 2004; Dorado et al, 2015; Cira et al, 2021; Tominack and Wetz, 2022), 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 small 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 (Caffrey, 2004). 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 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 (Russell et al, 2006). Finally, it is reasonable to assume that fluctuations in DO may not occur immediately in response to nutrient pulses or freshwater inflow. Work has has shown that various water quality parameters may have lagged effects lasting days or even months following storms and large discharge events (Mooney and McClelland, 2012; Wetz and Yoskowitz, 2013; Bukaveckas et al, 2020; Walker et al, 2021). However, our work only evaluates responses to loading and inflows occuring the day of water quality observations.

5 Conclusion

GAM models appear to provide reliable estimates of nutrient loads in the Lavaca Bay watershed. However, additional flow-biased data collection efforts would decrease the prediction variance and improve accuracy at critical high flow events. Ongoing projects will fill data gaps for total nitrogen and TKN loading. This study, consistent with others along the Texas coast, found strong effects of freshwater flow on nutrient and chlorophyll-a concentrations. DO concentrations, dominated by seasonal effects, did not show strong direct responses to freshwater flow. Small variance in flow-adjusted nutrient loads indicates that (1) there have been limited changes in non-point sources of nutrients and (2) that there isn't strong evidence that those small changes have had effects on chlorophyll-a or dissolved oxygen in Lavaca Bay. Although the study did not identify strong responses to changes in nutrient loading, this does provide a baseline assessment for future water quality management activities in the watershed.

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Data Availability. Reproducible code and datasets generated during this study are available in the Zenodo repository, https://doi.org/10.5281/zenodo.733075.

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