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 secondary embayment on the Texas coast displaying early signals of water quality degradation. This study applied a semiparametric approach to assess both watershed nutrient loads and 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 flow-normalized loads indicate that nutrient loadings were driven by natural variation in precipitation and runoff as opposed to changes in management of nonpoint sources. Models indicated there was no evidence of long-term changes in dissolved oxygen or chlorophyll-a within Lavaca Bay. 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 watershed 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

Remaining Items

- keywords
- discussion
- formalize language (reduce first person active?)
- final language review
- springer doesn't want figures in subdirectory, need to knit to same directory as tex file.

1 Introduction

Similar to 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 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 several 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 watershed nutrient loading and resulting 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. Despite largely meeting statewide water quality standards, there have been concerning recently identified declines in abundance, biomass, and diversity of benthic fauna in Lavaca Bay (Beseres Pollack et al, 2011). These declines are partially attributed to reductions in freshwater inflow and changes in estuary salinity and are indicative of an already stressed system (Beseres Pollack et al, 2011; Palmer and Montagna, 2015; Montagna et al, 2020). More recently, significant linear increases in total phosphorus (TP), orthophosphate, total Kjeldahl nitrogen (TKN), and chlorophyll-a concentrations were identified at monitoring sites within Lavaca Bay (Bugica et al, 2020). Although long-term changes in DO concentrations have not been 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 flexible statistical methods such as Weighted Regressions on Time, Discharge and Season (WRTDS, Hirsch et al, 2010) and Generalized Additive Models (GAMs, Wood, 2011) have 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 when assessing nutrient concentration trends (Beck and Murphy, 2017). Water quality applications of GAMs have included include river and catchment nutrient concentration and load estimation (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), assessment of temporal trends of nutrients (Beck and Murphy, 2017; Murphy et al, 2019), phytoplankton (Bergbusch et al, 2021), and cyanobacteria (Hayes et al, 2020). Recently GAMs have also been used to link water quality responses in receiving water bodies to changes in nonpoint source 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 exploratory 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 are required to be collected under Quality Assurance Project Plans and lab method procedures outlined by the TCEQ's procedures manual to ensure 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).

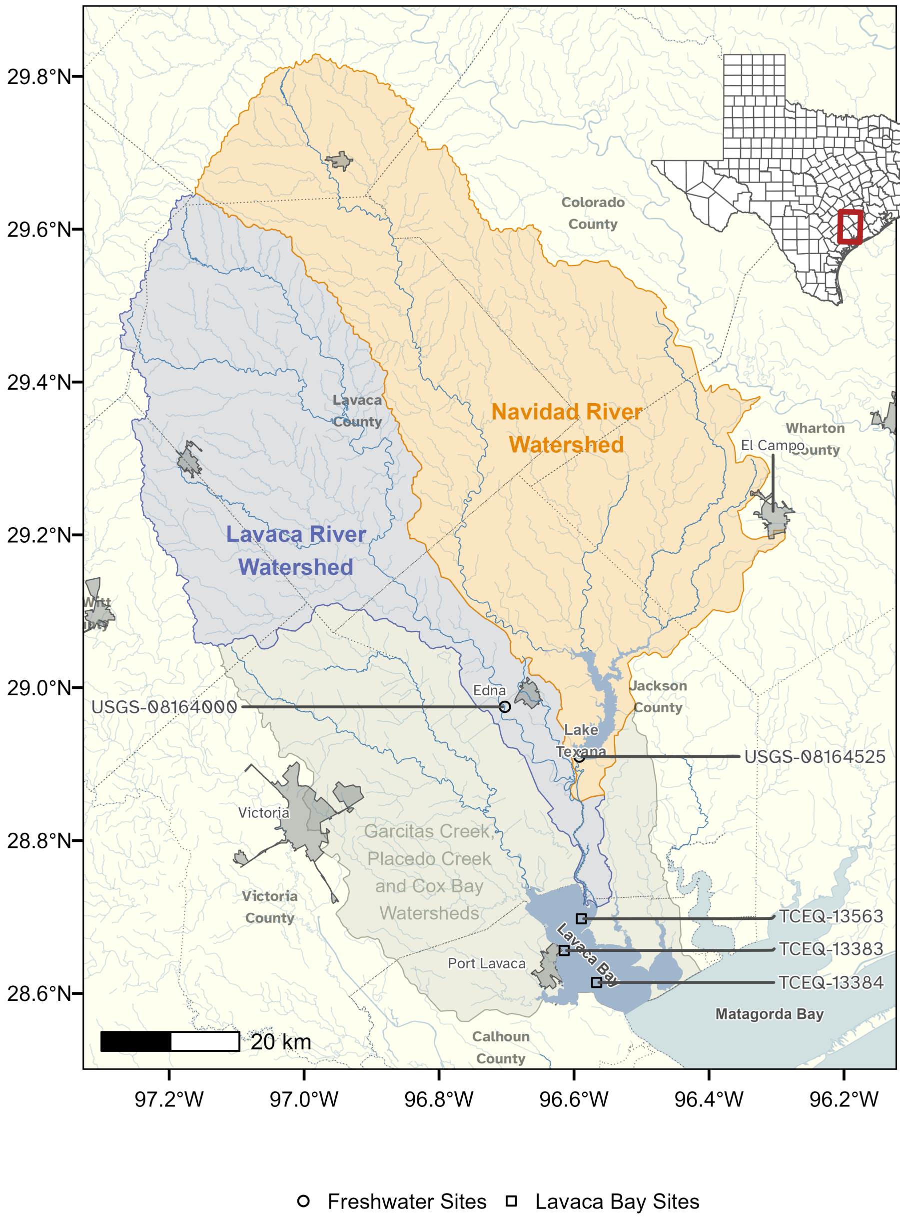

Fig. 1 Map of Lavaca Bay and the contribution watershed. The freshwater sites are the most downstream freshwater stream locations with water quality and streamflow data used for nutrient load models. Water quality concentration data at the three Lavaca Bay sites were used to assess relationships between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

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 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:

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

where μ is the conditional expected NO_3 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). Moving averages and flow anomalies were calculated with the *adc* R package (Schramm, 2023).

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 90% 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). 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):

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

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

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

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)

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 freshwater inflow and nutrient loads were prepossessed to account for season and streamflow respectively. Raw freshwater inflow values were replaced by seasonally adjusted log transformed inflow obtained from the residuals of a GAM model fit between season(day of year) and log transformed daily freshwater inflow. Raw nutrient loads were replaced with flow-adjusted values obtained from the residuals of a GAM model 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 evidence of model covariate effects on Lavaca Bay water quality. Model probabilities were calculated and compared using the AIC_c scores between each group of temporal, inflow, and inflow+load models (Burnham et al, 2011). Improvements in model probabilities provide evidence that the terms explain additional variation in the response variable.

3 Results

3.1 Watershed Nutrient Loads

Predictive performance of nutrient loads ranged from "satisfactory" to "very good" based on standardized evaluation metrics of NSE, r, and PBIAS (Moriasi et al, 2015) 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.

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

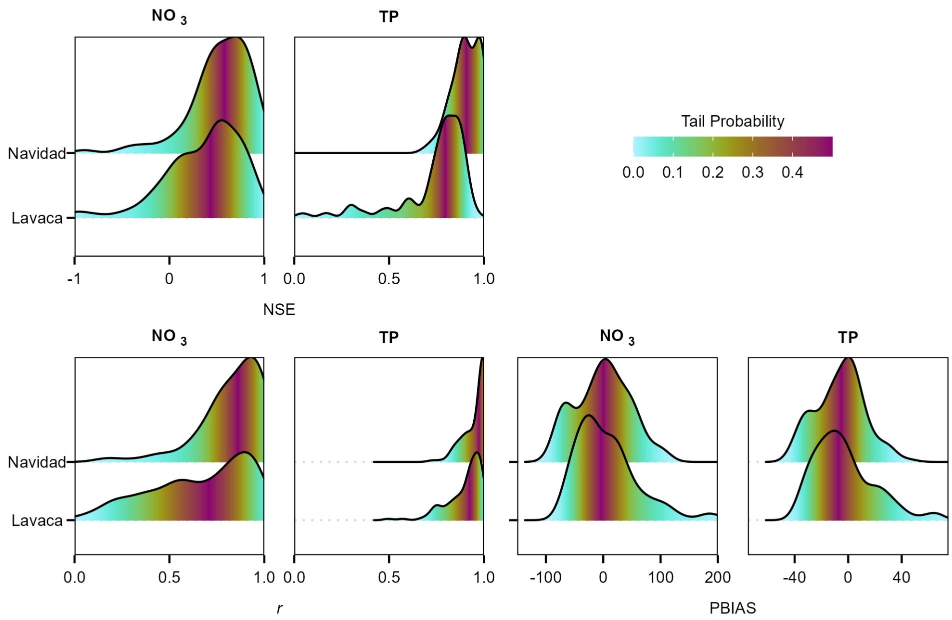

Fig. 2 Density plots of goodness-of-fit metrics (NSE, r, and PBIAS) from repeated 5-fold cross validation between predicted nutrient loads from GAM models and measured nutrient loads. Color indicates the tail probability calculated from the empirical cumulative distribution of the goodness-of-fit metrics.

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).

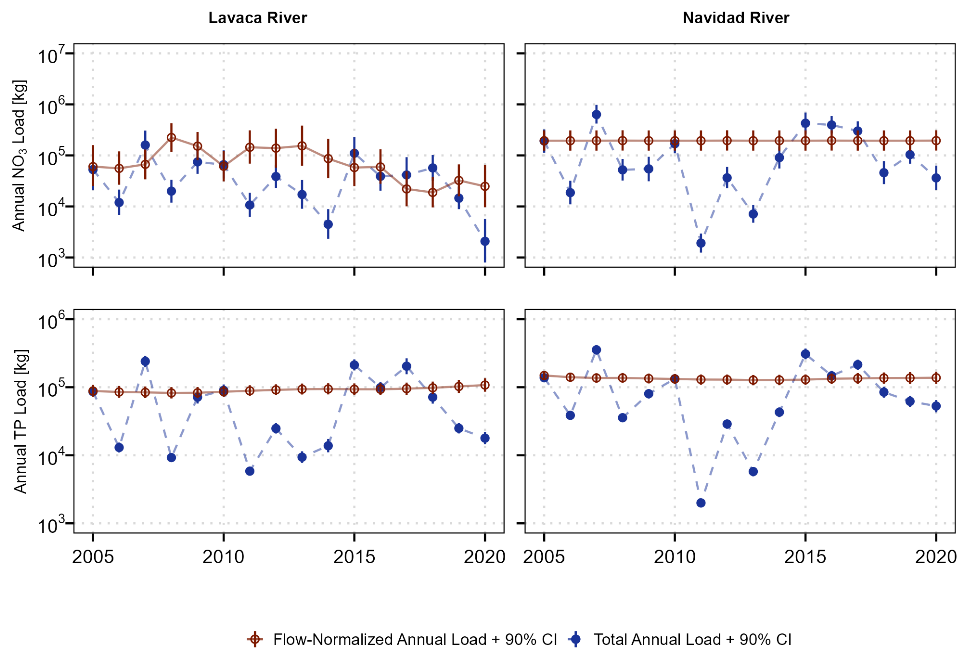

Fig. 3 Aggregated estimated annual and flow-normalized annual NO_3 and TP loads for USGS-08164000 and USGS-08164525.

3.2 Linkages Between Water Quality and Watershed Flows and Loads

There is no evidence of long-term changes in TP or DO concentrations at any Lavaca Bay site (Figure 5). The upper-bay site, TCEQ-13563, shows evidence of a long-term linear increase in NO_x while chlorophyll-a decreased from 2005 through 2014 (Figure 5). NO_x concentration at the mid-bay site, TCEQ-13383, displayed an unusual periodic pattern that is indicative of a strong influence from inflow or precipitation. The temporal GAMs did not provide evidence of long-term trends in any of the water quality constituents at 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

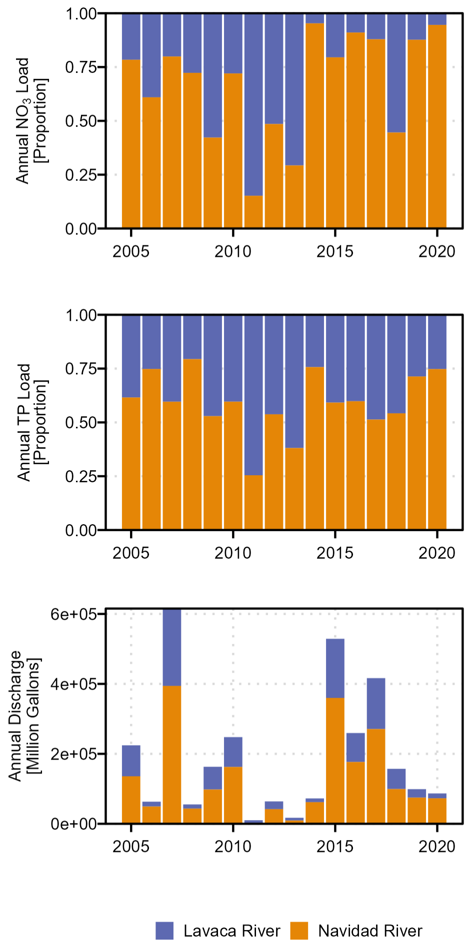

Fig. 4 Comparison of delivered annual loads and annual discharge at the Lavaca (USGS-08164000) and Navidad (USGS-08164525) Rivers.

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

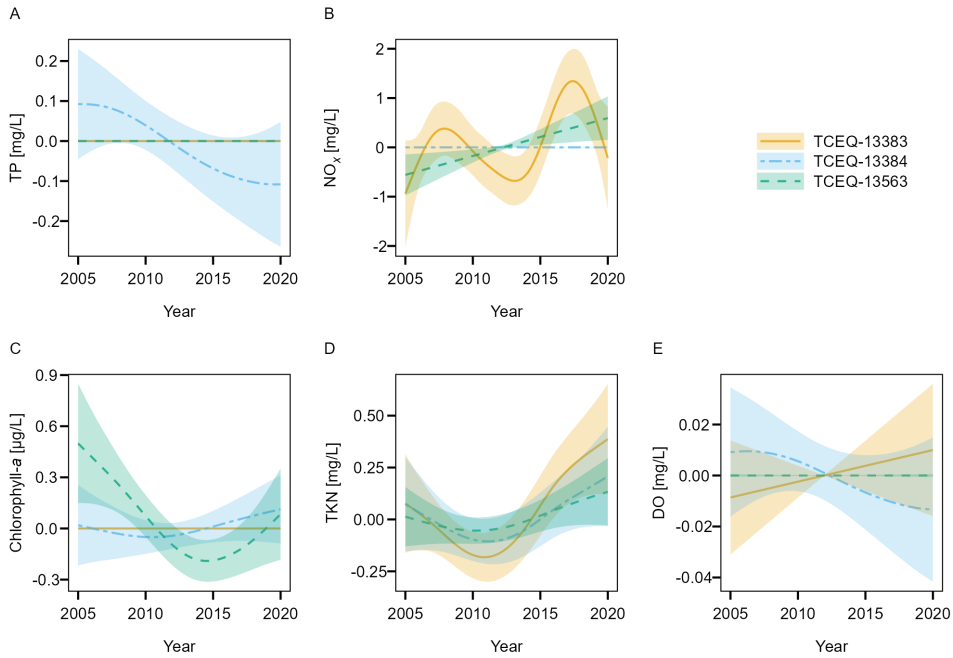

Fig. 5 Fitted splines (shaded regions indicate 90% confidence intervals) from the temporal estuary GAM display the marginal smoothed effect of date on TP (A), NO_x (B), chlorophyll-*a* (C), TKN (D), and DO (E) concentrations at each site in Lavaca Bay.

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).

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	Inflow	Inflow + 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)

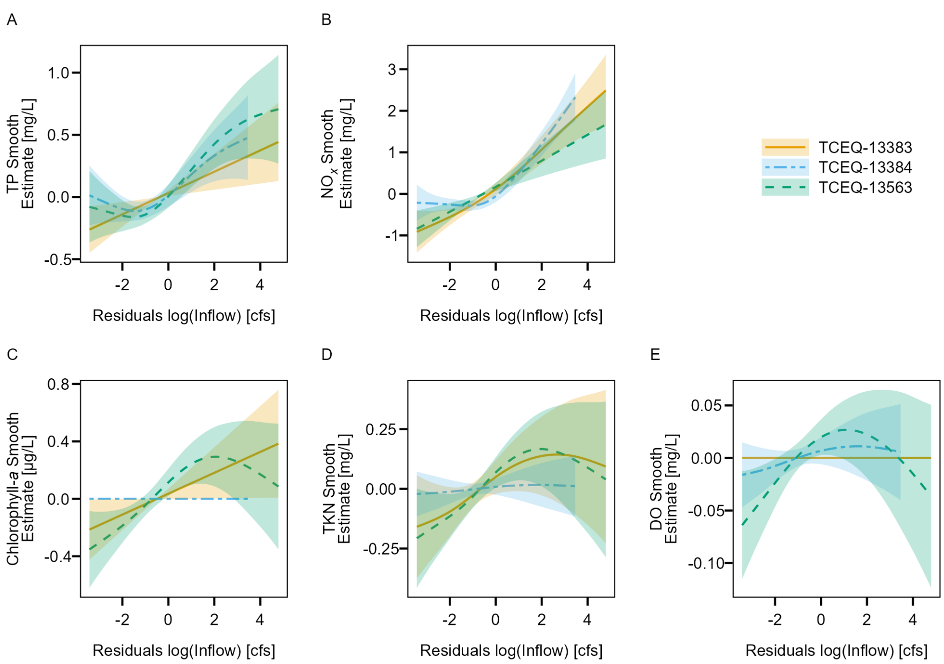

Fig. 6 Fitted splines from estuary GAMs display the marginal smoothed effect of freshwater inflow (controlled for season) on TP (A), NO_x (B), chlorophyll-a (C), TKN (D), and DO (E) concentrations at each site in Lavaca Bay.

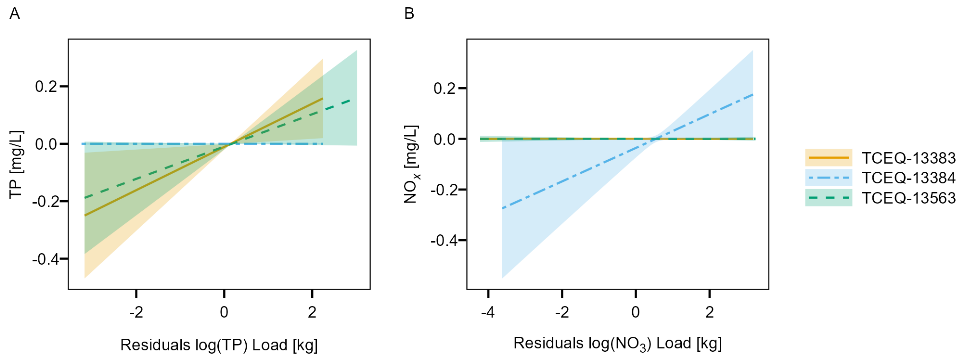

Fig. 7 Fitted splines from the nutrient loading GAMs display the marginal smoothed effect of TP and NO_3 loads on TP (A) and NO_x (B) concentrations at each site in Lavaca Bay.

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. 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 driven primarily by fluctuations in discharge. Notably, there were no indications of trends in flow-normalized NO_3 and TP loads in the Navidad River. Freshwater discharges in the Navidad River are regulated by the Palmetto Dam forming Lake Texana at the lower extent of the river. While the dominant agricultural land uses differ between the Lavaca (primarily grazed pasture and rangeland) and Navidad (mix of pasture and row crops) catchments, we did not have a reason to expect different flow normalized trends between the two systems from land use alone. Lentic nitrogen uptake and cycling may have regulating effects that mask changes in upstream nitrogen loadings. Additional nutrient data collection in the contributing tributaries of Lake Texana is needed to fully assess the role of Lake Texana in regulating nutrient delivery to the Lavaca Bay system. However, these results suggest that there have been no changes in the NO_3 or TP loading from the Navidad River system at the Lake Texana discharge point when accounting for variations in year to year discharge.

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 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 the lack of evidence for similar changes in flow-normalized TP loads provide contrary support. The inconsistent flow-normalized trends may also reflect some of the weakness of the water quality dataset that is primarily composed of ambient water quality measurements.

Cross-validation of the nutrient loading models highlights that predictions are prone to high bias, owing to the lack of targeted storm or flow biased measurements. The high biases are indicative that subsets of values were unable to capture the functional relationships with the flow based dependent variables. 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 critical to improve model performance and strength of evidence produced by these models (Horowitz, 2003; Snelder et al, 2017).

Some prior studies have generated estimates of mean annual TP yields in the Lavaca River watershed (Table 5, Dunn, 1996; Rebich et al, 2011; Omani et al, 2014; Wise et al, 2019). Although these studies differ in time periods and methodologies, they provide a sanity check for the reasonableness of the annual estimates generated in the current study. In a regional assessment of nutrient loading in river's along the

Gulf of Mexico, Dunn (1996) used the LOADEST model to develop an estimated mean annual yield of 28.9 kg/km^2. LOADEST is a multiple linear regression model that fits log transformed pollutant concentrations to long term, seasonal, and flow based predictors and includes methods for bias correction when exponentiating the response variable. While the annual yield estimated in Dunn (1996) is substantially lower than the estimate of 35.2 kg/km^2 in this study, it is important to note that both hydrology and land use was substantially different for a proportion of the study period. For almost the first ten years of the study period in Dunn (1996), the Navidad River was still undammed. More recently the eastern portion of the watershed has seen large conversions of rice fields to more traditional row crop production (Kulat et al, 2019). Studies of long-term precipitation patterns indicate both the total amount (Dixon and Moore, 2011) and variability of rainfall (Mishra and Singh, 2010; Fagnant et al, 2020) has increased in the region which provide plausible reasons for increased runoff driven nutrient loading.

Rebich et al (2011) and Wise et al (2019) used SPARROW to provide recent assessments of regional catchment based loadings to the Gulf of Mexico (Table 5). SPARROW is a hybrid statistical-process model with the underlying nutrient load estimation methods based on the previously described LOADEST (Schwarz et al, 2006). The functional form of the LOADEST regression model is similar to the terms applied in the GAMs used in the current study. The only study to apply a mechanistic watershed model (SWAT) to estimate nutrient loadings in the Lavaca River watershed Omani et al (2014) developed estimated yields (42 kg/km^2) similar to the two SPARROW models. The base periods used by Rebich et al (2011), Omani et al (2014) and Wise et al (2019) differ from the current study, making direct comparisons difficult. However, the ranges of estimated yields suggest the current estimates of TP loading are reasonable. Similar results would be expected given all of the previous studies have also relied primarily on ambient water quality measurements instead of flow and storm biased measurements.

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

Reported Yield ($kg \cdot km^2 \cdot year^{-1}$)	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.

The non-linear estuary water quality trends identified in the current study 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. Both DO and cholorophyll-a concentrations at all three Lavaca Bay sites were stable from 2005 through 2020. This is a positive outcome in comparison to other Texas estuaries that are facing larger demands for freshwater diversions, higher population growth, and more intense agricultural production which have resulted in more direct signs of eutrophication (Wetz et al, 2016; Bugica et al, 2020). Despite the stability of DO and cholorophyll-a, there are concerning site specific increases in NO_x and TKN concentration over the same time period. These trends are espcially 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 concentration 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. The results suggest that tidal flushing maybe diluting TKN and act as a control on the effects of freshwater inflow in the lower reaches of Lavaca Bay. The results are also consistent with previous work that suggest processing of organic loads in the upper Lavaca Bay or tidal portions of the Lavaca River reduce transport of nutrients to the lower reaches of Lavaca Bay (Russell et al, 2006).

Freshwater inflow also displayed 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.

There was no evidence that adjusted freshwater inflow and nutrient loads had effects on DO concentration in Lavaca Bay. 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 concentration indicates physical factors (such as temperature, wind, and turbidity) 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 many 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, this study only evaluated responses to loading and inflows occurring the day of water quality observations.

The GAM approach proved useful for both estimating loads and assessing downstream responses in water quality. Although we did not compare other models, it is likely similar estimates of loadings would be obtained by methods suggest as LOADEST, WRTDS, or SPARROW given the functionally similar dependent variable structures. The underlying weakness in the estimates of loading in the current study is the reliance on ambient water quality data used for statewide water quality assessments. These data do not frequently include storm or flow biased samples which is needed to improve model performance. Although there is existing work on the samples sizes required for reliable performance of both LOADEST (Park and Engel, 2014) and WRTDS (Kumar et al, 2019) models, similar work does not appear to have been extended to water quality applications of GAMs. Due to the concerning increases in eutrophication associated parameters in Lavaca Bay and other Texas estuaries (Bugica et al, 2020), and the increasing need to better link environmental outcomes with on the ground management actions (Schramm et al, 2022) there is a strong requirement for reliable estimates of pollutant loadings and responses along the Texas coast.

To date, state wide water quality monitoring programs have focused on collection of ambient condition data. A framework for establishing pollutant load monitoring programs across catchments that explicitly incorporate flow biased data is needed for assessing estuary health along the Texas coast. Additional efforts focused on identifying relevant effect sizes, sampling designs, and funding mechanisms that can support long term efforts are also needed to adequately design such a framework. Large long-term monitoring programs in and around the Chesapeake Bay, San Francisco Bay, and along the Mississippi River have proven extremely effective at informing management actions and tracking progress towards long-term pollutant reduction goals. Similar coordinated efforts across Texas coastal watersheds would provide useful for resource management efforts intended to protect biological and water quality integrity of Texas's estuaries.

5 Conclusion

Reliable estimates of nutrient loads for the Lavaca Bay were obtained using GAMs. However, additional flow-biased data collection efforts is needed to decrease the prediction variance and improve accuracy at critical high-flow loading events. While some ongoing projects will fill data gaps for total nitrogen and TKN loading, additional efforts are needed to coordinate data collection efforts specifically for load estimation across Texas estuaries. 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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