Escherichia coli and Dissolved Oxygen Trends in the Upper Llano River Watershed, Texas (2001-2016)

Texas Water Resources Institute | College Station, Texas
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Abstract

Trends in *Escherichia coli* (*E. coli*) and dissolved oxygen at water quality monitoring stations in the Upper Llano watershed were evaluated for calendar years 2001-2016. Trends in *E. coli* concentrations and loads were evaluated with linear regressions and Weighted Regressions on Time, Discharge, and Season (WRTDS) method. Trends in dissolved oxygen concentrations were evaluated with linear regression and generalized additive mixed models (GAMMs). Linear regressions provide insight on instream concentration trends. WRTDS and GAMMs were also used to remove effects of year to year variations of flow on *E. coli* and annual variations in mean temperature on dissolved oxygen. Linear regression indicated a statistically significant decrease in *E. coli* on the main stem of the Llano River immediately downstream of the the North and South Llano River confluence. Linear regression did not indicate statistically significant changes in DO at any sites in the the watershed.

Keywords: Llano River, *E. coli*, Dissolved Oxygen, Water Quality

Introduction

The Upper Llano River Watershed includes the North and South Llano rivers and is located in the central Texas hill country (Figure 1). Both rivers have high water quality are supportive of diverse aquatic communities and sustain locally important recreational opportunities. Although both rivers meet existing water quality standards, local stakeholders voiced concern about changing land uses, proliferation of invasive species, and pressures on water supply as local populations grow. These concerns lead to the formation of local stakeholder organizations and the development of the Upper Llano River Watershed Protection Plan (ULRWPP). The URLWPP, accepted by the Environmental Protection Agency in Summer 2016, identified and recorded those key concerns raised by local stakeholders. Among the surface water quality issues identified in the plan were concerns about low dissolved oxygen (DO) and the elevated indicator bacteria, *Escherichia coli* (*E. coli*). Since 2016, implementation efforts in the watershed to reduce *E. coli* bacteria loads and potential contributors to depressed DO have expanded. Amongst these efforts are enrolling agricultural producers into Natural Resource Conservation Service (NRCS) conservation plans or prescribed grazing plans intended to reduce runoff, protect riparian areas, increase soil health, and reduce livestock time spent in streams. Although limited water quality sampling has occurred, this report intends to provide an understanding of bacteria and dissolved oxygen trends through 2016. The purpose of this report is to evaluate changes in water quality concentrations over time using simple statistical regression methods and to evaluate potential correlations with dissolved oxygen concentrations.

Methods

Study Location and Data Sources

The Upper Llano Watershed includes the catchments of both the North Llano and South Llano River (Figure 1). For purposes of this report, the watershed was extended downstream to capture water quality trends of the combined flows from both watershed. The project watershed was 4,809 km² (1,188,330 acres). We obtained mean daily streamflow from three gages in the study area from the USGS National Water Information System (NWIS) Figure (1, Figure 2, and Table 1). The Texas Commission on Environmental Quality (TCEQ) water quality data was obtained through the Clean Rivers Program data tool. Although a number of water quality monitoring stations occur within the watershed, stations with less than one year of data were omitted because they could not provide meaningful insight on temporal concentration trends. Records from five water quality stations were included in the analysis (Figure 1, Figure 2, and Table 2). Although NL2 includes a limited number of available sample events, it was included because sampling at the long-term station NL1 was recently discontinued due to site access issues. Water quality monitoring data used in this report were collected by TCEQ, Lower
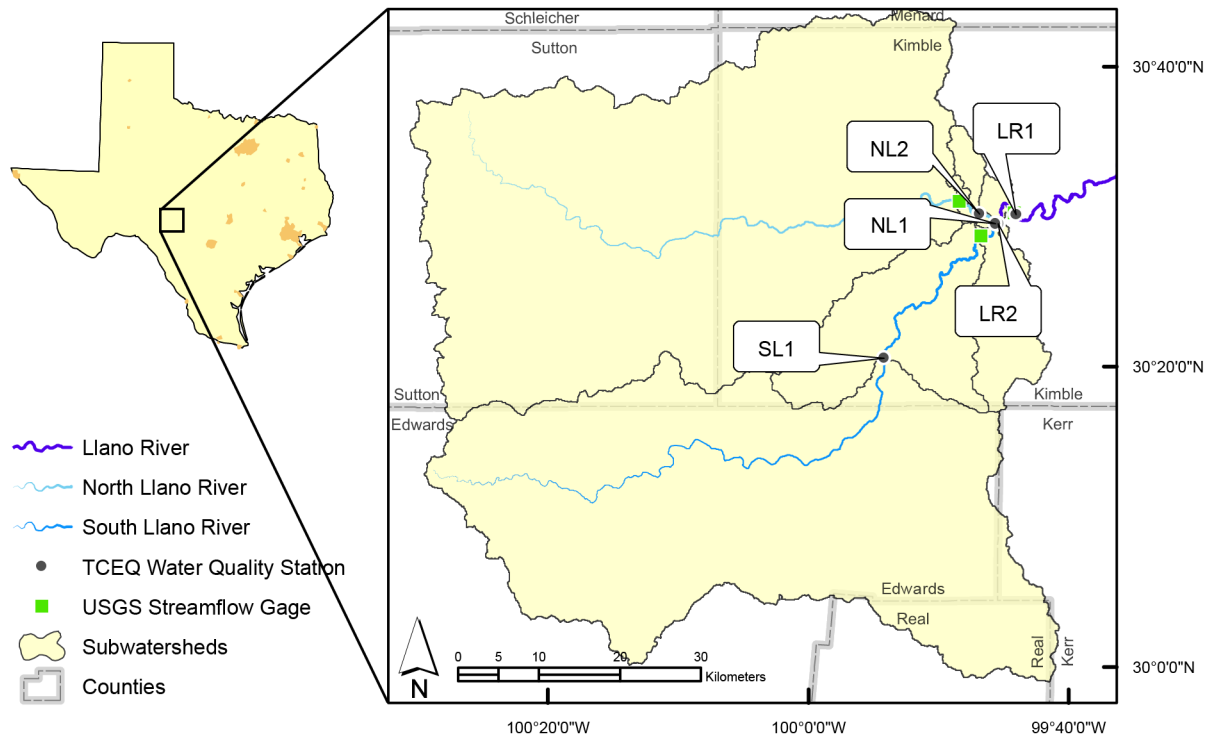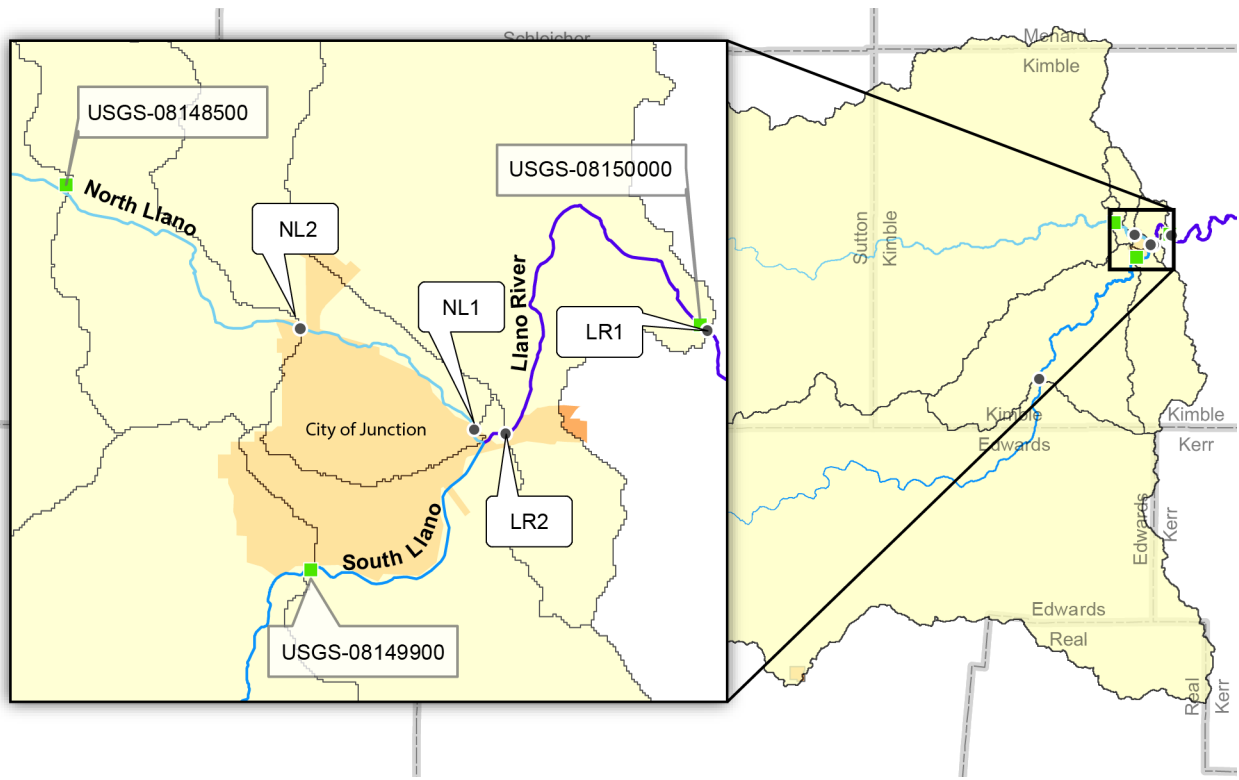
Figure 1: Study area

Figure 2: Detailed map of stream gage and monitoring station locations near Junction, Tx

Table 1: USGS gage summary information

USGS Site No.	Q_{min}*	Q_{max}†	$\bar{Q}$‡	$\tilde{Q}$§	Drainage Area (km^2)	Name	Period of Record
08148500	0.0	13000	37.39	16.0	2332	North Llano River near Junction, TX	2001-2016
08149900	23.4	4160	70.88	49.1	2277	South Llano River at Flat Rock Ln at Junction, TX	2012-2016
08150000	31.8	45900	146.11	95.6	4809	Llano River Near Junction TX	2001-2016

* Minimum flow
† Maximum flow
‡ Mean Flow
§ Median Flow

Table 2: TCEQ water quality monitoring station summary

TCEQ Station Number	Reference Name	Number of Samples	Date Range	Drainage Area	$\frac{A_y}{A_x}$
17471	LR1	16	2011-10-03 : 2015-08-10	4809	NA
21489	LR2	41	2001-08-06 : 2015-02-06	4789	0.996
17425	NL1	35	2001-09-11 : 2014-08-18	2372	1.017
21548	NL2	8	2014-10-13 : 2016-10-18	2342	1.004
18197	SL1	13	2011-04-20 : 2016-10-18	1941	0.852

Colorado River Authority, Texas State University, and other Clean Rivers Program Partners. The collection of these data are conducted under TCEQ and EPA approved Quality Assurance Project Plans.

Streamflow Estimation

Mean daily streamflows for ungaged water quality monitoring sites (LR2, NL1, NL2, SL1) were developed using the Drainage-Area Ratio (DAR) method. DAR provides a simple method of estimating streamflows at ungaged sites, in particular for nested subwatersheds (Hirsch, 1979; Ries and Friesz, 2000; Emerson et al., 2005). DAR calculates streamflow discharge as:

$$Y_i = X_i(\frac{A_y}{A_x})^\phi$$

where Y_i is the calculated discharge at the ungaged site at time i, X_i is discharge measured at the gaged site at time i, while $(\frac{A_y}{A_x})$ is the ratio of ungaged and gaged catchment areas. Values used for $(\frac{A_y}{A_x})$ are indicated in Table 2.

In most applications, ϕ is assumed to equal one (Asquith et al., 2006). However, when applied to Texas streams, ϕ = 1 results in substantial over- or under-estimation at the largest magnitude streamflows (Asquith et al., 2006). Therefore, we applied values for ϕ based on streamflow percentile values as indicated by Asquith et al. (2006) (Table 3). To facilitate interpretation of hydrologic conditions as each station, flow duration curves were plotted and converted to load duration curves overlaid with water quality measurements (Morrison and Bonta, 2008). The load duration curves were developed using Texas's primary contact recreation criterion for *E. coli* bacteria in surface freshwater bodies of 126 cfu/100 mL. The duration curves provide a visual analysis of flow conditions at stations, as well as an understanding of what flow conditions water quality concentrations and loads exceed established water quality standards.

Table 3: Reccomended ϕ values for DAR method applied in Texas streams (Asquith2006)

Streamflow percentile	ϕ	Streamflow percentile	ϕ	Streamflow percentile	ϕ
0	0.885	38	0.893	76	0.934
2	0.885	40	0.894	78	0.934
4	0.885	42	0.895	80	0.933
6	0.886	44	0.897	82	0.933
8	0.886	46	0.899	84	0.931
10	0.887	48	0.902	86	0.927
12	0.887	50	0.905	88	0.920
14	0.887	52	0.908	90	0.906
16	0.887	54	0.912	92	0.890
18	0.888	56	0.916	94	0.865
20	0.888	57	0.920	95	0.850
22	0.888	60	0.924	96	0.830
24	0.889	62	0.927	97	0.806
26	0.889	64	0.930	98	0.773
28	0.889	66	0.932	99	0.737
30	0.890	68	0.934	100	0.700
32	0.890	70	0.935		
34	0.891	72	0.935		
36	0.892	74	0.935		

Correlation Analysis

Monotonic relationships between dissolved oxygen concentrations, log transformed streamflow, and stream temperature were evaluated graphically and with Kendall's Tau (Helsel and Hirsch, 2002). Censored values and outliers were not filtered due to the resistance of rank-methods such as Kendall's Tau to these values. Correlations were considered significant at the $p < 0.05$ level. All statistical analysis and plots were completed in R, versions 3.4.0 (R Core Team, 2016).

Concentration and Load Trends

We evaluated trends in bacteria concentrations collected at each station with simple linear regressions of log transformed concentrations against date (Helsel and Hirsch, 2002). Dissolved oxygen concentrations were evaluated using untransformed values regressed against date. The null hypothesis of zero slope over time was rejected at $p < 0.05$. Scatter plots of concentrations and dates are provided to aid result interpretation. We verified linear regression assumptions of normality with qqplots and by plotting model residuals against dates. LR2 and NL1 were the only stations with enough data to estimate concentrations and loads using regression based techniques. We estimated instream mean daily bacteria loads using the Weighted Regressions on Time, Discharge and Season (WRTDS) approach with the EGRET package in R (Hirsch et al., 2010; Hirsch and De Cicco, 2015). We used WRTDS outputs to describe trends in concentration and loads over time. WRTDS is a regression-based approach that models log transformed water quality constituents concentrations as a function of time, flow, and season. The functional model is:

$$ln(E.coli) = \beta_0 + \beta_1 T + \beta_2 ln(Q) + \beta_3 cos(2\pi t)$$

Where T is decimal time and included in the model as an annual term (β_1) or seasonal term (β_3, β_4), and Q is flow. WRTDS differs from other regression based estimators, such as LOADEST, by using a weighted regression to estimate coefficients for each combination of Q and t. The coefficients are used to estimate daily concentrations, daily loads, and flow-normalized loads and concentrations. Flow-normalized values are derived from removal of

variation in concentration based on random variations in discharge. Further details of WRTDS are outlined in Hirsch et al. (2010). These flow normalized values are useful to understand how changes in land practices over time affect instream constituent concentrations and loads. In particular flow normalized values are less influenced by increases in loads solely attributed to increases in stormflow runoff. Flow-normalized values provide a useful insights that are not reflected by regression on concentration values alone, which can be influenced by changes in precipitation and runoff and might not reflect progress in land management activities.

We estimated instream dissolved oxygen concentrations at LR and NL1 using generalized additive mixed models (GAMMs) (Wood, 2008). GAMMs are regression models that use non-linear smooth functions on covariates. Dissolved oxygen concentrations were not expected to be a function of flow, but a function of stream temperature, season, long-term trend, and instream metabolic processes. Therefore, we utilized GAMMs to describe dissolved oxygen concentration because it allows the inclusion of specified terms and random effects. The generalized GAMM model used to fit dissolved oxygen concentrations is:

$$DO = \beta_0 + \beta_1 ln(Q) + \beta_2 Year + \beta_3 Month + \beta_4 Temperature$$

Where Q is flow, $Year$ is year term, $Month$ is the seasonal month term, and $Temperature$ is the water temperature term. Appropriateness of GAMM model fits were evaluated by inspecting model residuals following procedures described by Zuur et al. (2009). GAMMS were developed using the `mgcv` and `nlme` packages in `R`.

Results

Flow Duration Curves

The flow duration curves (Figure 3) indicate the Llano River at stations LR1 and LR2 maintained consistent flows through the period of record (2001-2015). Compared to LR1 and LR2, the flow duration curves of the North Llano at stations NL1 and NL2 depict a flashier stream with periods of no flow. Flows at SL1 indicate a more consistent flow, likely contributing to consistent flows seen downstream at LR1 and LR2. Streamflow records for SL1 are limited because the USGS gage on the South Llano began data collection in 2012.

E. coli

E. coli Load Duration Curves

Load duration curves (Figure 4) indicate bacteria concentrations and loads remain below the allowable criterion across all flow conditions at each station. Isolated individual load exceedances, where measured values are above the load duration curve line, occur sporadically across the duration interval.

E. coli Concentration Trends

Linear regression of log transformed bacteria concentrations indicated a significant decrease in E. coli concentration at LR2 (T = -2.08, p <0.05) from 2001 through 2016 (Figure 5). The null hypothesis of no increasing or decreasing trend was not rejected at LR1, NL1, NL2, or SL1 over the respective available sample periods.

Figure 6 depicts the WRTDS estimated flow-normalized bacteria concentrations and loads at LR2 and NL1. From 2001 through 2016 annual mean flow-normalized *E. coli* concentration decreased 33 cfu/100 mL and flow normalized loads decreased 5.87×10^6 million colonies per day from 2001 through 2016 at LR2 (Table 4, Figure 6). *E. coli* concentration and load at NL1 decreased by 235 cfu/100mL and 1.27×10^5 million colonies per day from 2002 through 2016. Although substantial, the estimates for both stations exhibit considerable uncertainty with estimated bias statistic of 0.287 for LR2 and 0.138 for NL1. An inspection of model diagnostics indicate samples at both stations are considerably underrepresented at high flow events. Furthermore, temporal gaps (2009-2011) in monitoring data decrease confidence in these estimated concentration and loads.

Figure 7 and Figure 8 show the WRTDS estimates of *E. coli* concentration at LR2 and NL1 as a function of discharge over season and year. LR2 is characterized by increases in concentration with increases in discharge. However, in recent years, that relationship is less strong or even negative during spring and early summer months. Estimates at NL1 (Figure 8) indicate elevated concentrations at lower flows which generally decrease with increase discharges. There also appears to be a substantial shift in the concentration-discharge relationship during the June
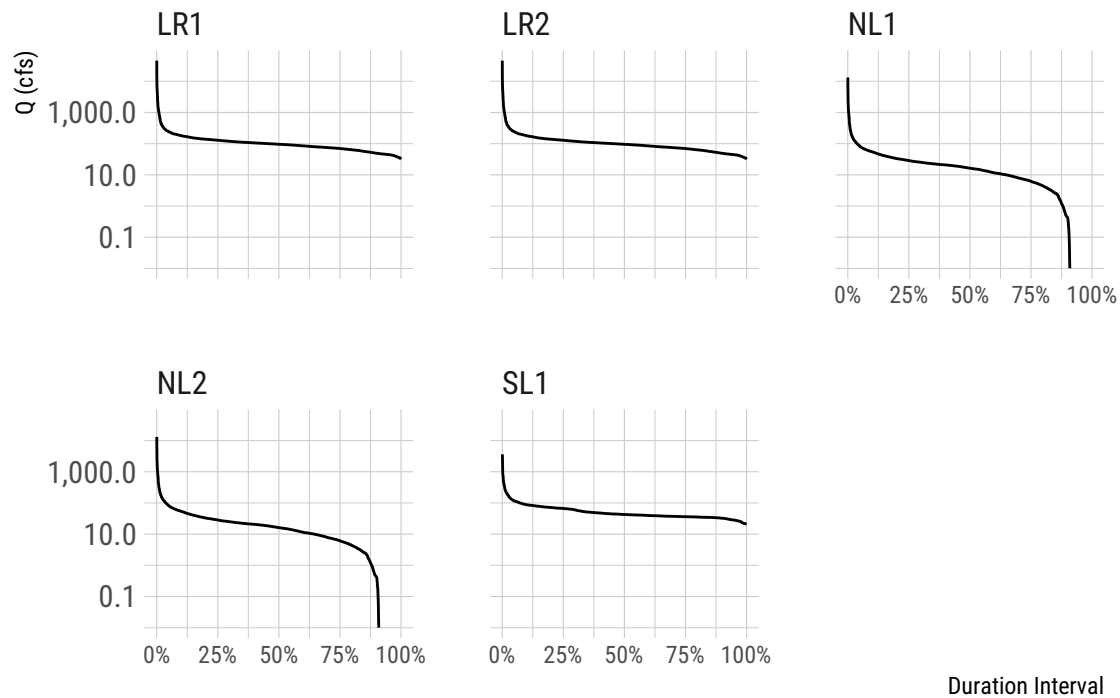
Figure 3: Flow duration curves of estimated mean daily flows at each station. Duration interval indicates the percent of days the mean daily flows is exceeded.

Table 4: WRTDS estimated changes in flow-normalized mean daily *E. coli* concentrations and loads.

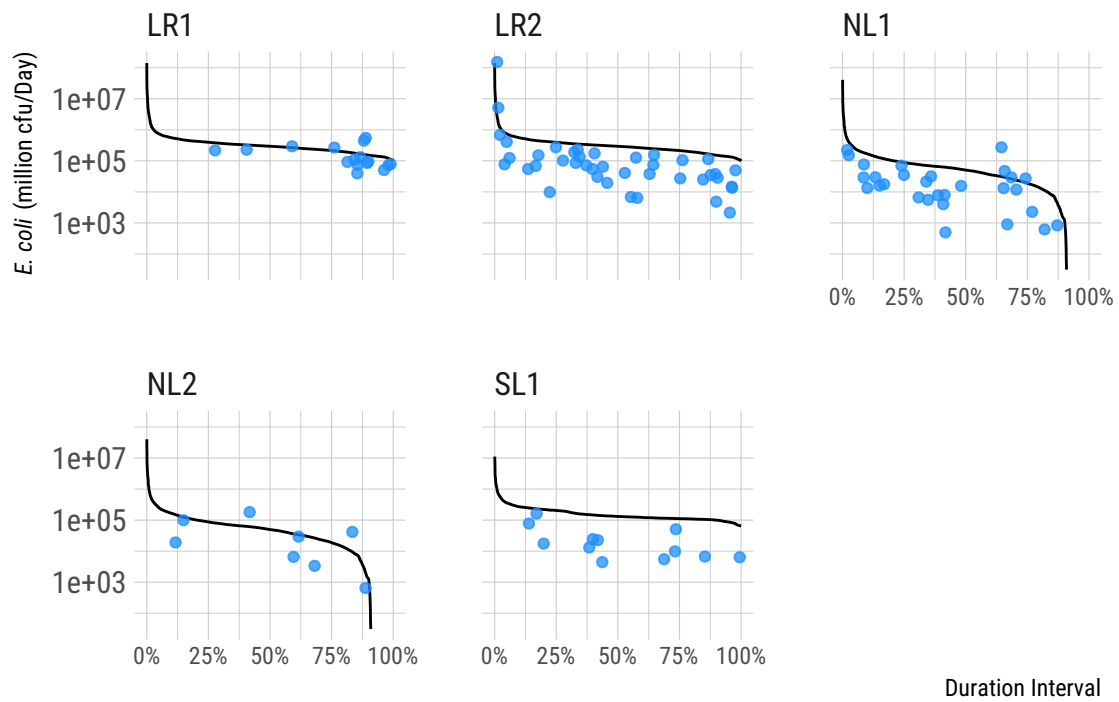| Station	Period	Change in Flow Normalized Mean Concentration (cfu/100mL)	Change in Flow Normalized Mean Load (million colonies per day)
LR2	2001-2016	-33.2	-5.869207×10^6
NL1	2002-2016	-235.0	-1.27081×10^5

Figure 4: Bacteria load duration curves. The black line indicates allowable bacteria loads based on the 126 colonies per 100 mL water quality criterion. The points indicate measured bacteria concentrations converted to loads based on estimated mean daily flows.
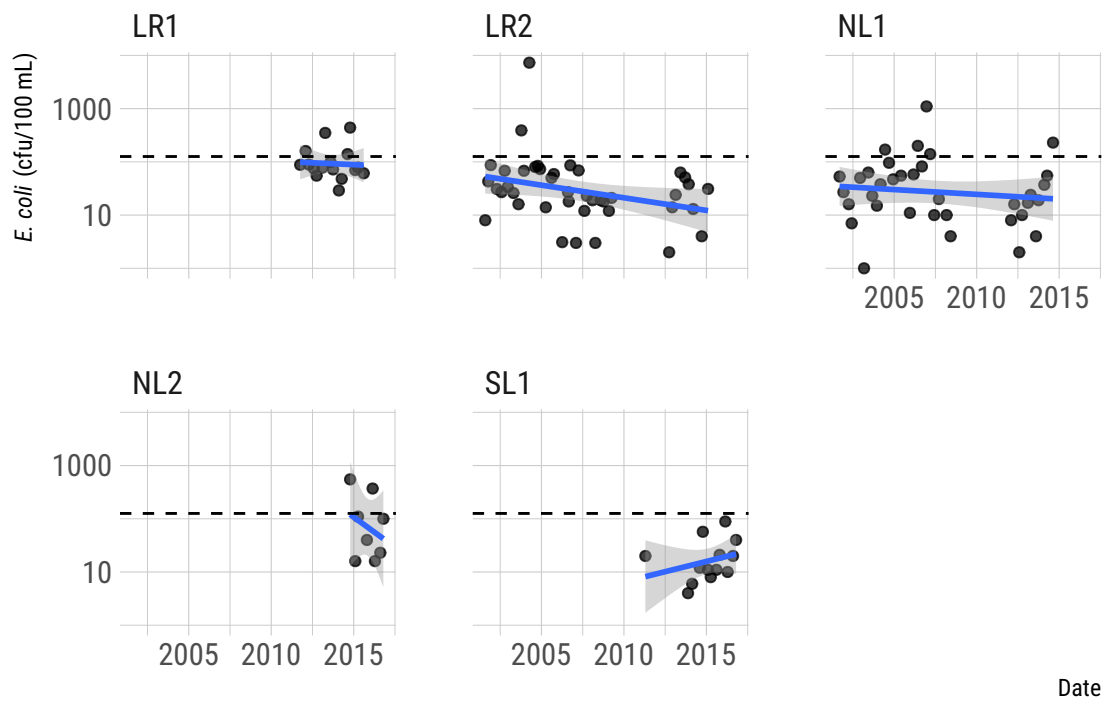
Figure 5: *E. coli* concentrations plotted against time for each station. The solid line is the linear regression through all points, and the shaded area indicates the 95% confidence intervals.
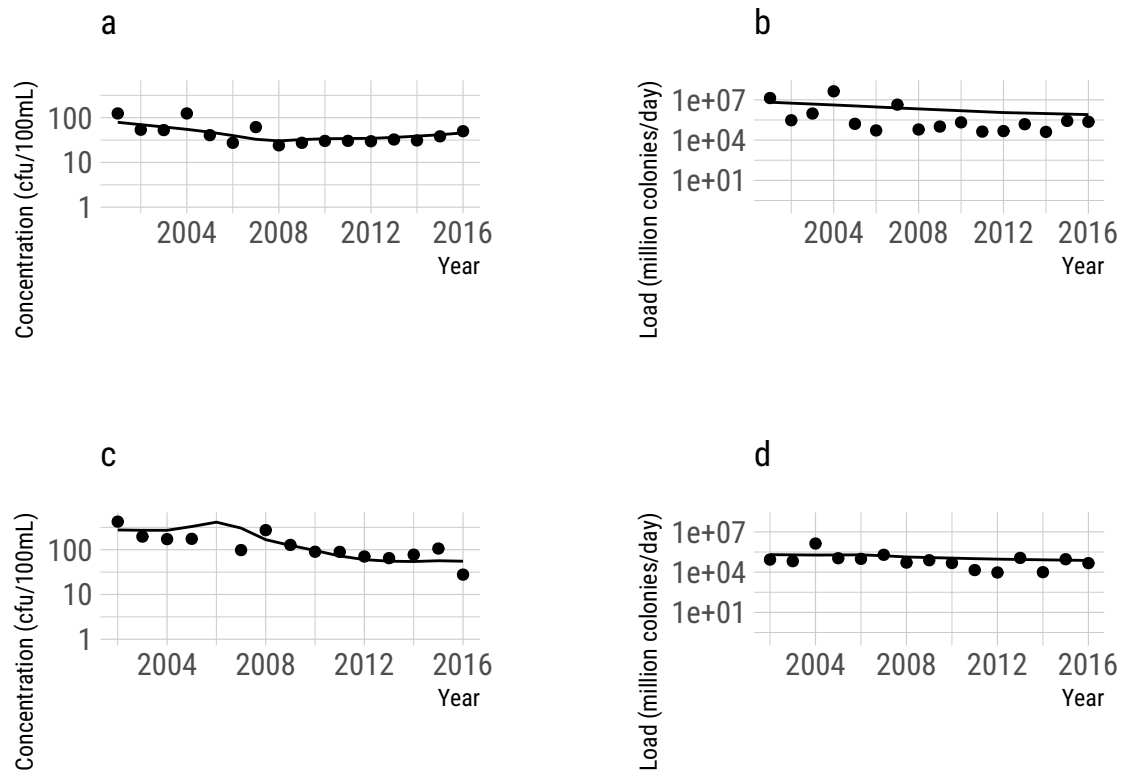
Figure 6: Annual mean estimated (dots) and flow-normalized (lines) *E. coli* concentrations at LR2 (a) and NL1 (b); annual mean estimated and flow normalized *E. coli* loads for LR2 (b) and NL1 (d).

through December time-period with *E. coli* concentration decreasing more rapidly with increased discharge in recent years.

Dissolved Oxygen

Linear regression did not indicate a statistically significant trend in DO concentration at any of the stations from 2001 through 2016 (Figure 9). Although a handful of samples fell below 5 mg/L (the current TCEQ established standard for minimum DO values in the Upper Llano watershed), long-term trends indicate grab DO concentrations remain well above the established limit.

The correlation matrix (Figure 10) depicts the relationship between DO, temperature, and log flow at each station in the watershed. DO and temperature exhibit a strong relationship at LR1 (τ = -0.81, $p < 0.001$), LR2 (τ = -0.38, $p < 0.001$), NL1 (τ = -0.44, $p < 0.001$), and NL2 (τ = -0.7, p < 0.01). A less strong but still significant relationship between DO and temperature was detected at SL1 (τ = -0.53, $p < 0.05$). No correlation appears between DO and log flow or temperature and log flow.

GAMMs indicate a small but significant decrease in DO concentration at LR2 (F = 5.990, p = 0.02) and NL1 (F = 7.636, $p < 0.001$) when controlled for temperature and season. Mean DO at LR2 decreased 1.07 mg/L or 8% between 2001 and 2016. The mean DO at NL1 decreased 2.71 mg/L or 8.8% between the maximum value at 2005 and minimum at 2016. The GAMMs fit to DO concentrations demonstrated reasonable certainty with adjusted R-squared values of 0.661 (LR1) and 0.677 (NL1). The adjusted R-squared values describe the amount of variance each model is expected to account for. Figure 11 shows the change in mean DO contributed by the long-term trend term, or what we could expect as a change in mean DO as a function of time, controlled for changes in flow, temperature, and season.

Figure 12, Figure 13 depict the GAMM estimated dissolved oxygen concentrations as function of water temperature, season (month of year), and year. Both stations reflect a decreasing dissolved oxygen concentration trend over time. Generally, a strong decrease in DO concentration with increasing water temperature occurs as water temperature increase up to 20 degrees Celsius. The GAM estimates a leveling off in concentration reductions, with an increase noted at higher temperature at NL1.

Discussion

E. coli

Linear regression indicated a significant decreasing trend in fecal indicator bacteria at station LR2 over the entire sample period. Linear regression did not indicate trends in fecal indicator bacteria at the other sites. Linear regression also did not indicate trends in dissolved oxygen at any sites. These stream concentrations are important to evaluate as indicators of compliance with water quality regulations and stream health. The duration curve and linear regression results indicate general water quality compliance. However, absence of recent sampling at sites with long-term datasets (LR2 and NL1) prevent extrapolation of results to current conditions. We used the WRTDS regression approach to describe bacteria loads and concentrations controlled for flow, as well as describe bacteria concentration as a function of flow and season. Although WRTDS indicates, substantial decreases in flow normalized E. coli concentration and load at LR2 and NL1, considerable bias indicates substantial uncertainty with the regressions. This uncertainty can be attributed to (1) relatively infrequent sampling for load estimation purposes, (2) a substantial data gap in the middle of the data set, (3) and sampling did not capture enough high flow events. Despite the uncertainty, we can be reasonably infer that flow normalized loads did not increase over the sample period. An inspection of *E. coli* concentration as a function of flow and time at LR2 (Figure 7), indicates E. coli concentration typically increases with flow, as is expected for nonpoint source influenced constituents. Importantly, the seasonal increases (September through December) in concentration that occurred early in the sampling period appear to be reduced at this site. NL2 (Figure 8) shows a completely different relationship, indicating possible flushing and decreased concentration of bacteria as flows increase. The elevated bacteria levels at low flows are muted in recent years.
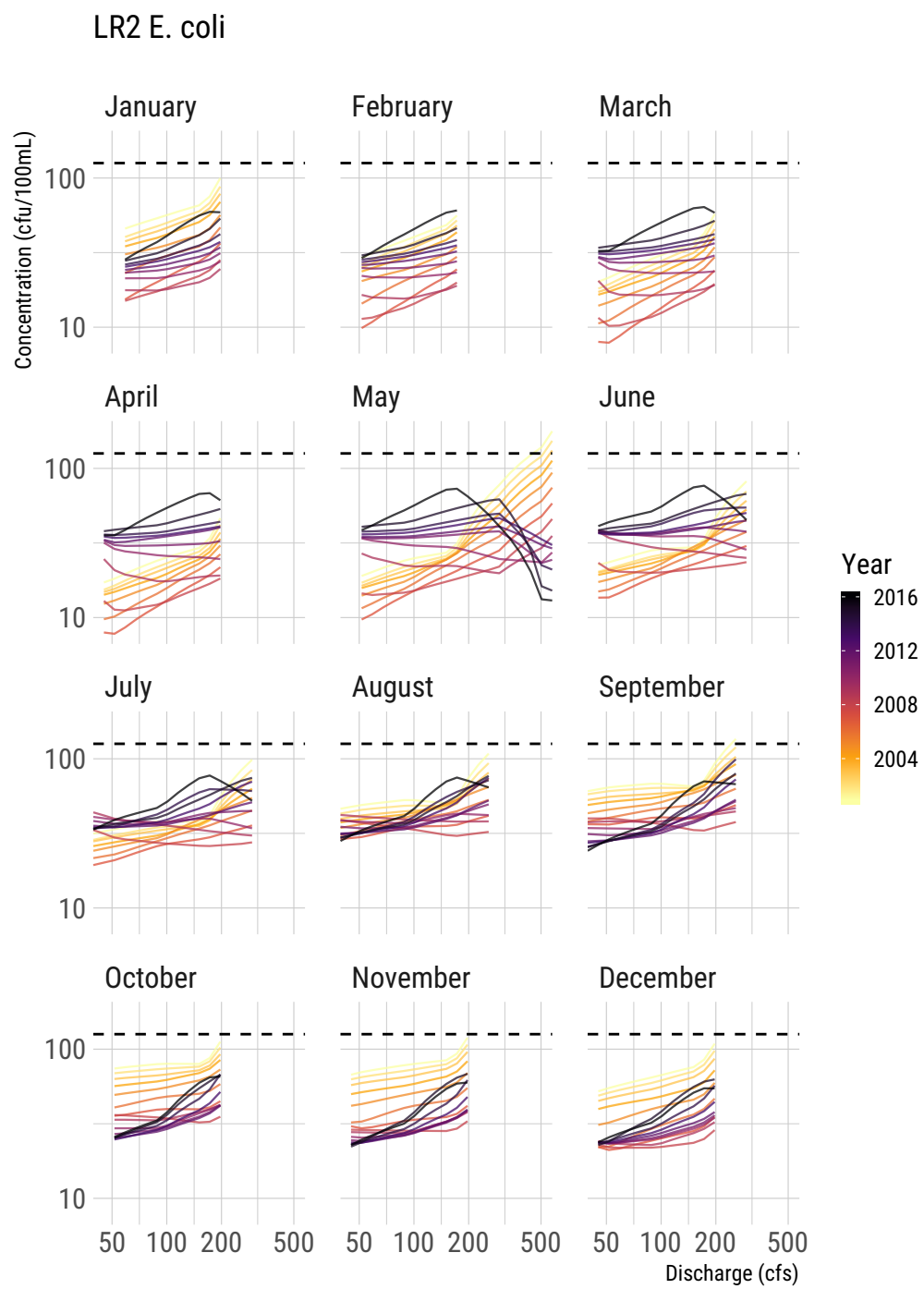
Figure 7: *E. coli* concentration and flow relationship over time and month at LR2.
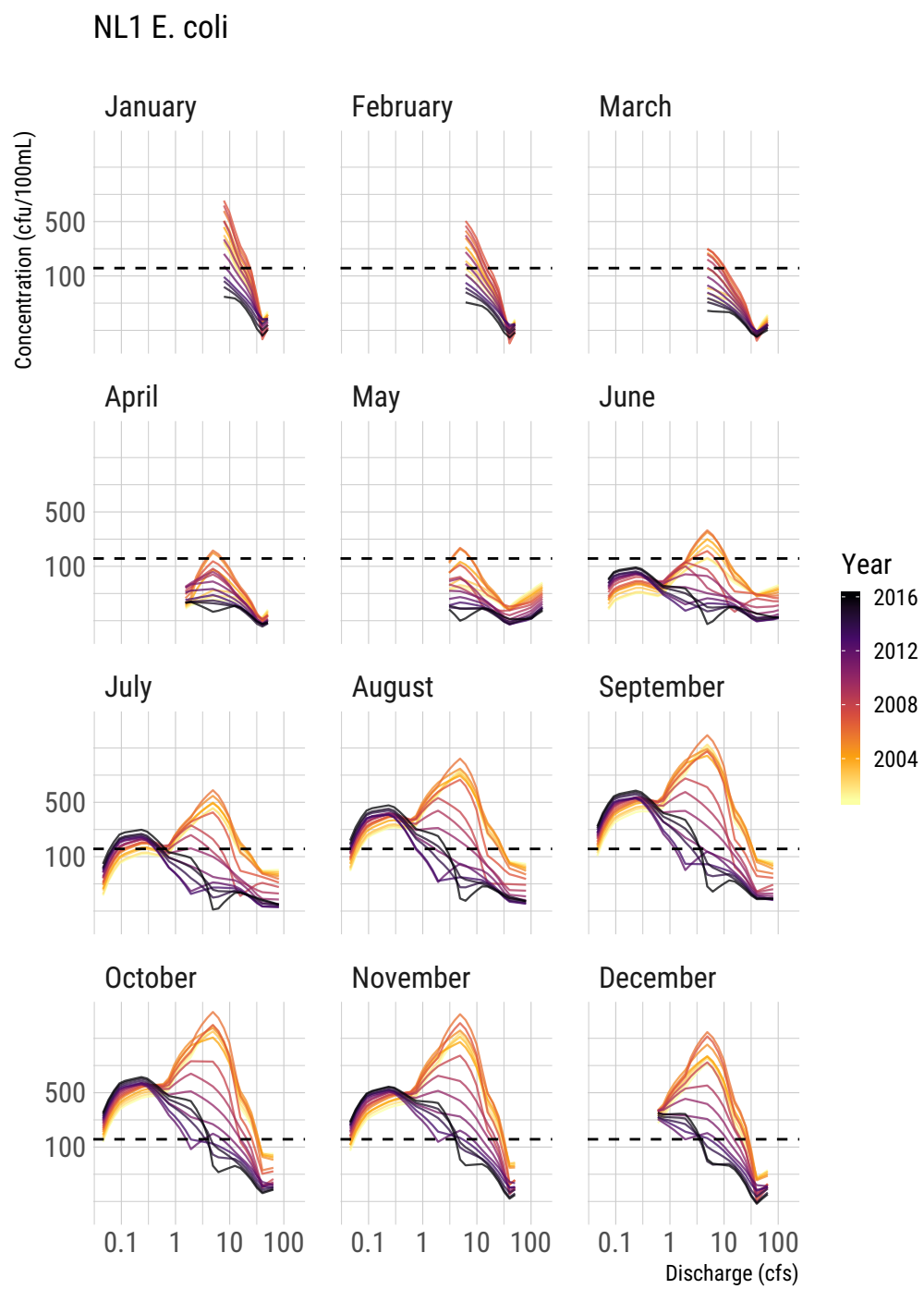
Figure 8: *E. coli* concentration and flow relationship over time and month at NL1.
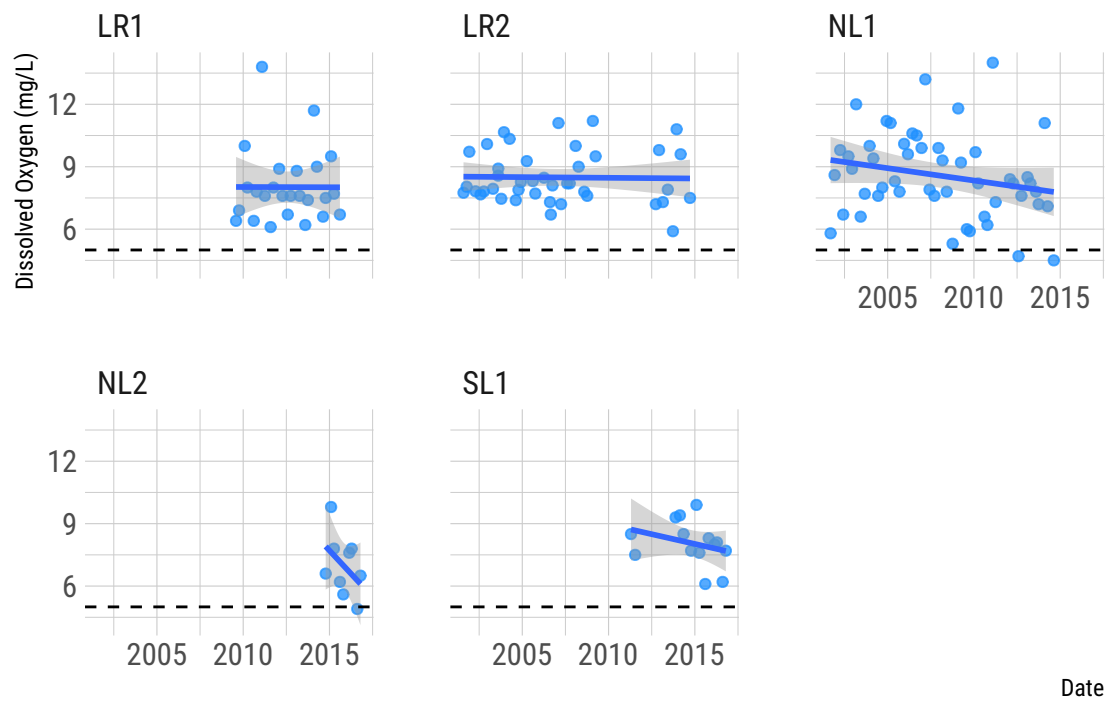
Figure 9: DO concentrations plotted against time for each station. The solid line is the linear regression through all points, and the shaded area indicates the 95% confidence interval.
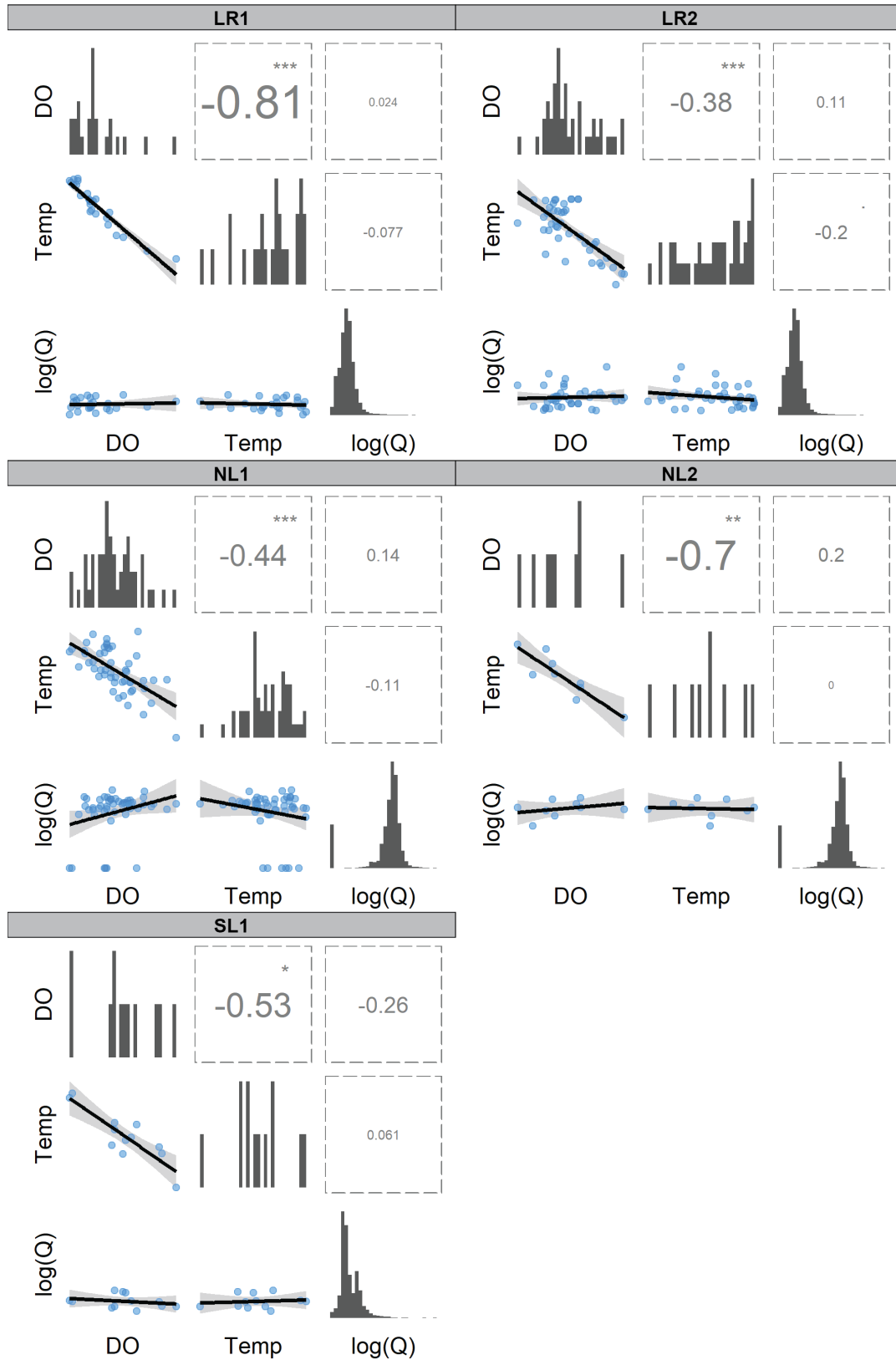
Figure 10: Correlation matrix for project stations. Kendall's Tau values are indicated in the upper right hand section of each matrix, a histogram of indicated values are included along the diagonal plots, and measured values are in the low lower left hand plots. Levels of significance are indicated as * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$
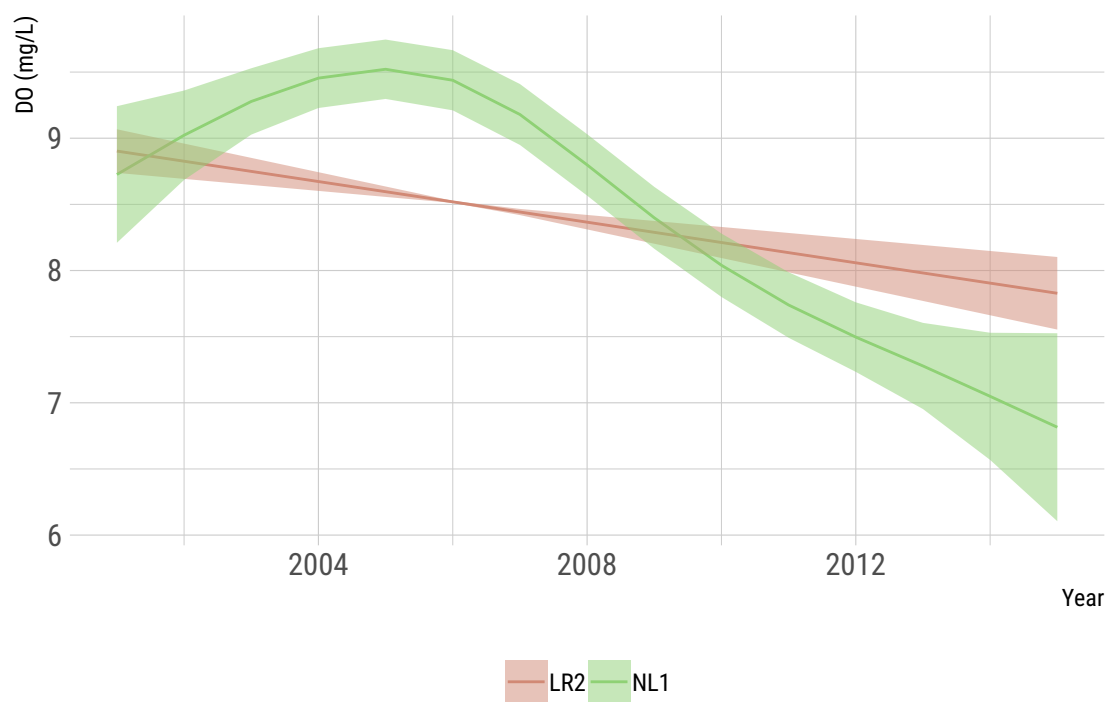
Figure 11: GAMM estimated changes in DO concentration from the long-term trend term. Solid lines are the estimated mean DO concentrations, shaded areas are the estimated standard errors

Figure 12: Estimated DO concentration and temperature relationships over time and season at NL1. Months without sampling data are not included in the figure.

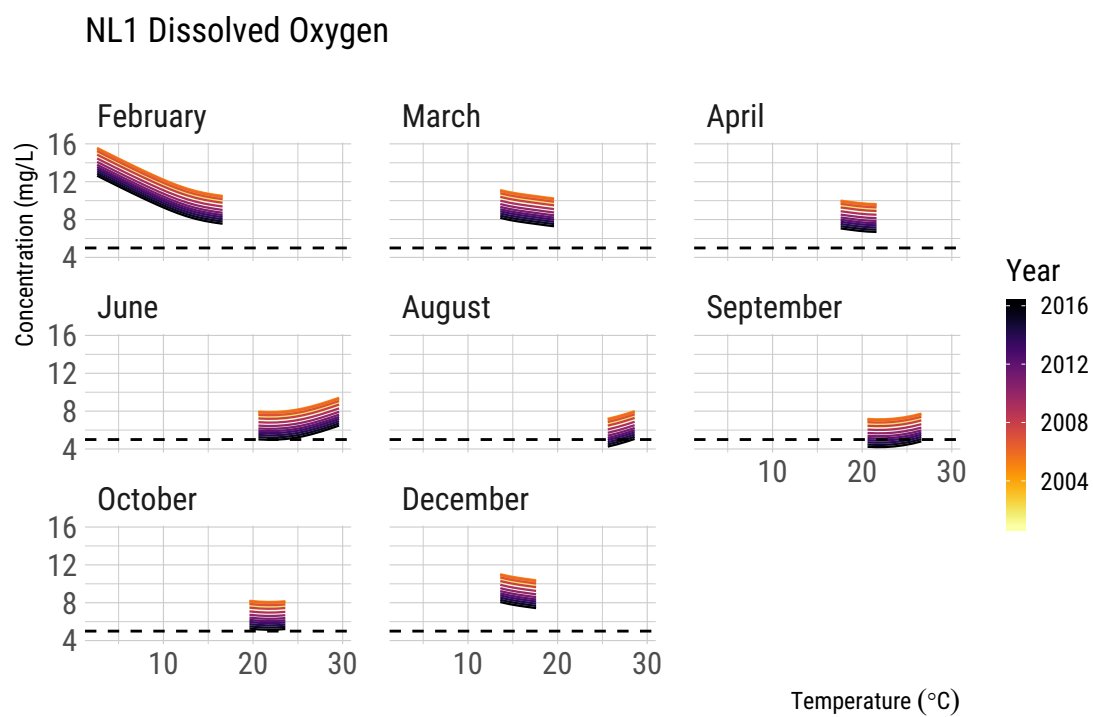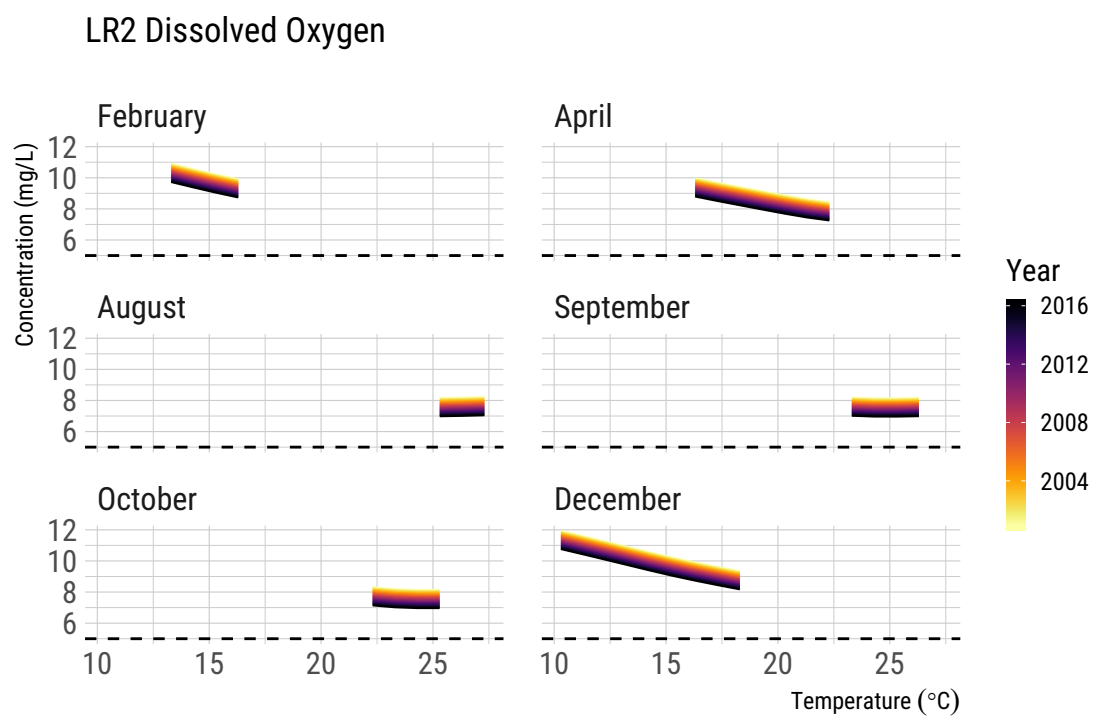Figure 13: Estimated DO concentration and temperature relationships over time and season at LR2. Months without sampling data are not included in the figure.

Dissolved Oxygen

While standard linear regression did not indicate a trend in dissolved oxygen concentrations, the GAMM regression indicated a small but significant decrease in dissolved oxygen as a function of time controlled for temperature and season. Overall dissolved oxygen concentration remained in compliance over the sampling period; however, the temperature and seasonal controlled trends point to possible underlying changes in mean dissolved oxygen concentration. Importantly, for NL1, periodic seasonal exceedances of the dissolved oxygen water quality standard appear more likely to occur based on the GAMM regression (Figure 13). Again, the lack of recent sampling data negatively impacts the confidence of regression results near the end of the sampling period, as reflected in the wide standard errors at NL1 (Figure 11). Increased monitoring frequency and continuity in site sampling effort would certainly increase confidence in regression results. While changes in DO concentrations are informative, understanding the potential drivers and mechanisms behind those changes would be valuable for land management decisions. Crucially, covariates such as nitrogen and phosphorous loads were not estimated because of the sparse sampling efforts associated with nutrients parameters thus far.

Conclusion

This report describes trends in *E. coli* and DO in the Upper Llano watershed using regression-based techniques (WRTDS and GAMM). Examination of the trends indicate that the parameters are within acceptable ranges for compliance with existing water quality standards. However, the GAMM approach identified a decreasing trend in dissolved oxygen concentrations (controlled for season and temperature) at NL1. Caveats to this report include substantial bias identified in the *E. coli* load estimations attributed to sparse data collection and lack of high flow sampling events. In order to provide increasingly reliable load estimates, future sampling efforts need to ensure representativeness of sampling conditions (flow and season). Furthermore, while the GAMM estimates of DO were reasonable, regressing dissolved oxygen changes with changes in nutrient loads would be informative for land management purposes. However, the sparseness of instream nutrient data collected to date prevents such analysis. Therefore additional efforts to fill this data gap are also recommended.

References

Asquith, W. H., Roussel, M. C., and Vrabel, J. (2006). Statewide Analysis of the Drainage-Area Ratio Method for 34 Streamflow Percentile Ranges in Texas. Technical Report Scientific Investigations Report 2006-5286, USGS.

Emerson, D., Vecchia, A., and Dahi, A. (2005). Evaluation of Drainage-Area Ratio Method Used to Estimate Streamflow for the Red River of the North Basin , North Dakota and Minnesota. Technical Report Scientific Investigations Report 2005-5017, USGS.

Helsel, D. R. and Hirsch, R. M. (2002). Statistical Methods in Water Resources. In *Techniques of Water-Resources Investigations of the United States Geological Survey. Book 4, Hydrological Analysis and Interpretation*, chapter Chapter A3, page 509. USGS, Reston, VA.

Hirsch, R. and De Cicco, L. (2015). User guide to Exploration and Graphics for RivEr Trends (EGRET) and dataRetrieval: R packages for hydrologic data (version 2.0, February 2015). Technical Report U.S. Geological Survey Techniques and Methods book 4, chap. A10, USGS.

Hirsch, R., Moyer, D., and Archfield, S. (2010). Weighted regressions on time, discharge, and seasons (WRTDS), with application to Chesapeake Bay River Inputs. *Journal of the American Water Resources Association*, 46(5):857–880.

Hirsch, R. M. (1979). An evaluation of some record reconstruction techniques. *Water Resources Research*, 15(6):1781–1790.

Morrison, M. and Bonta, J. (2008). Development of Duration-Curve Based Methods for Quantifying Variability and Change in Watershed Hydrology and Water Quality. Technical Report EPA/600/R-08/065, EPA.

R Core Team (2016). R Development Core Team.

Ries, K. G. and Friesz, P. J. (2000). Methods for estimating low-flow statistics for Massachusetts streams. Technical Report Water-Resources Investigations Report 00-4135, USGS.

Wood, S. N. (2008). Fast stable direct fitting and smoothness selection for generalized additive models. *Journal of the Royal Statistical Society. Series B: Statistical Methodology*, 70(3):495–518.

Zuur, A. F., Ieno, E. N., Walker, N. J., Saveliev, A. A., and Smith, G. M. (2009). *Mixed Effects Models and Extensions in Ecology with R.* Springer, New York.