

## **Drought and management effects on biophysicochemistry in Lake Austin, Texas**

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### **Abstract**

*During a recent drought period, a municipal reservoir was affected by increased duration and magnitude of cyanobacterial blooms, deep water hypoxia, and abundant Hydrilla verticillata growth. We analyzed over 20 years of water quality data from sources monitoring Lake Austin, Texas, USA, to relate changing hydraulic pulsing rates during periods of water surplus and deficit to system biophysicochemistry. During the drought period, average daily discharges through the reservoir system were significantly reduced to conserve water supply. In response, growing season hydraulic residence times in Lake Austin, typically a run-of-river system with residence of about 10 d, increased 2–8 fold. Nutrient loadings to Lake Austin decreased during low discharge, but only total phosphorus (TP) concentrations declined appreciably in the reservoir. Late summer lower flows across years created epilimnetic conditions conducive to cyanobacteria blooms because of stable stratification, water temperatures exceeding 25 C, and nitrate concentration minima with TP and ammonia maxima. Drought-induced discharge reductions promoted earlier onset of warm, stratified waters, and lower hypolimnetic oxygen concentrations. The physicochemical changes coupled with longer hydraulic residence altered the timing of algal bloom initiation, duration, and peak, and quadrupled the probability of cyanobacterial bloom occurrence. The largest cyanobacteria blooms were observed when growing season discharges were consistently below 20 m<sup>3</sup>/s (i.e., hydraulic residence time > 2 weeks). Concomitantly, the non-native macrophyte hydrilla rapidly spread in the warm, reduced flow conditions. By linking flushing rates to reservoir stratification and potential phytoplankton bloom magnitude, our results will allow managers to anticipate summer water quality conditions.*

## 23 **Introduction**

24 Water demand in many regions is frequently augmented and satisfied through construction of  
25 reservoirs along rivers (Cooke et al. 1993). Reservoir water quantity and quality are vulnerable to  
26 climatic changes that alter patterns of inflow and flushing rates, withdrawals, nutrient loading  
27 and cycling, and stratification (Lewis 2000; Mosley 2015; Yasarer and Sturm 2016). For  
28 example, discharge-driven hydraulic residence time in a reservoir influences nutrient mixing and  
29 processing rates, and can structure phytoplankton species composition and biomass (Thornton et  
30 al. 1990; Rangel et al. 2012; Cha et al. 2014). Reservoir biochemical responses to changing  
31 external drivers can negatively impact the ecosystem services the reservoir provides including  
32 provisioning of drinking water, recreational opportunities, and water for commercial and  
33 agricultural uses (Elliott et al. 2006; Yuan and Pollard 2015).

34 Reservoir managers need to understand chemical and biological responses to flow regime shifts  
35 due to regional climatic anomalies in order to sustain the delivery of ecosystem services  
36 (National Research Council 2012; Brooks et al. 2016). In this study, we evaluated over twenty  
37 years of monitoring data from a regionally important reservoir, Lake Austin, Texas, USA. Our  
38 objective was to compare periods of normal and drought-influenced reservoir water provisioning  
39 changes on dam discharge rates and hydraulic residence times, vertical temperature and oxygen  
40 profiles, nutrient concentrations, and phytoplankton abundances. We then estimated how  
41 flushing rates impacted the timing, probability of occurrence, and magnitude of phytoplankton  
42 bloom events. Our model findings should help managers to anticipate potential drinking water  
43 problems for mitigation strategy development.

## 44 **Methods**

## 45 ***Water management plans***

46 The LCRA's first water management plan for the lower Colorado River basin was published in  
47 1989 and is reviewed annually. Revisions approved in 1991, 1992, 1999, 2010, and 2015 have  
48 reflected improved hydrologic modeling capabilities, changes in annual demand coupled to  
49 current system storage and inflows, forecasted future needs, and modifications to the criteria and  
50 thresholds determining the provision (both volume and intra-annual recurrence) of water for  
51 customer and environmental flows while ensuring adequate storage. For the purposes of this  
52 study, we divided our data summaries and analyses into the periods of 1990–1998, 1999–2010,  
53 and 2011–2014. The first two delineations represent times of water supply adequacy and  
54 satisfaction of all downriver customer demands each year. We kept the periods separate in most  
55 analyses in order to visualize if changes in water provisioning, increasing demands by a growing  
56 region, or invasion of Lake Austin by hydrilla (*Hydrilla verticillata* [L.f.] Royle) in 1999  
57 appreciably impacted water quality. The last period encompassed a time of record low inflows  
58 into the basin ([http://www.lcra.org/water/water-supply/highland-lakes-  
59 overview/Documents/Fact-Sheet-Drought-by-the-Numbers.pdf](http://www.lcra.org/water/water-supply/highland-lakes-overview/Documents/Fact-Sheet-Drought-by-the-Numbers.pdf)). In order to protect water supply  
60 volume, beginning in 2012 LCRA ceased water releases to agricultural customers, which  
61 typically represent 70% of annual water demand.

## 62 ***Data sources***

63 The LCRA provided daily reservoir elevations (meters above mean sea level [m amsl]) and  
64 average daily discharge volume (cubic meters per second [ $\text{m}^3/\text{s}$ ]) at Mansfield Dam and Tom  
65 Miller Dam for the period from 1 January 1990 through 31 December 2014 ( $n = 9131$  for each  
66 dam). Hydraulic residence time was estimated daily by dividing Lake Austin's volume by the  
67 daily discharge from the Tom Miller dam. We also computed a moving average for discharge  
68 and hydraulic residence time for the previous 15 d period.

69 Water quality data was also provided by the LCRA (<http://waterquality.lcra.org/>). Reservoirs  
70 were typically sampled bi-monthly. Because Lake Travis is a bottom-release reservoir (~15 m  
71 above Lake Austin's surface elevation), we focused on hypolimnetic water chemistry (collected  
72 ~1 m above bottom) from the most down-river monitoring site (Fig. 1). Though disparate,  
73 hypolimnion thickness was typically in excess of 30 m, with the bottom ~20 m being anoxic in  
74 summer, so we assumed discharge was more reflective of hypolimnetic rather than epilimnetic  
75 (i.e., top 1 m) water chemistry. Lake Austin surface (~0.3 m) and bottom (~1 m above bottom)  
76 samples were collected from three sites (Fig. 1). Depth profiles for temperature (C) and dissolved  
77 oxygen (DO; milligram per liter [mg/L]) were collected using a Hydrolab MS5 Datasonde (OTT  
78 Hydromet, Loveland, Colorado, USA), and Secchi disk transparency (m) was determined. Water  
79 chemistry constituents analyzed in this study were ammonia-nitrogen ( $\text{NH}_3\text{-N}$ ; mg/L),  
80 nitrate+nitrite-N ( $\text{NO}_x\text{-N}$ ; mg/L), total Kjeldahl N (TKN; mg/L), total phosphorus (TP; mg/L),  
81 and chlorophyll *a* ( $\mu\text{g/L}$ ). Total N (TN; mg/L) was derived by summing TKN and  $\text{NO}_x\text{-N}$   
82 concentrations. Water chemistry analyses followed Environmental Protection Agency standard  
83 methods and were carried out by the LCRA-Environmental Laboratory Services (Austin, TX,  
84 USA; <http://www.lcra.org/services/els/Pages/default.aspx>).

85 Austin Water, the City of Austin's drinking water and wastewater utility, provided total algal and  
86 taxonomic group (flagellates, green algae, diatoms, and cyanobacteria) abundances (organisms  
87 per mL [org/mL]) beginning 12 January 1992. Due to homeland security concerns, sampling

88 frequency is classified. Final sample size for total algae, green algae, diatoms, and cyanobacteria  
89 equaled 7082, 7033, 7025, and 7012, respectively. Winsborough Consulting (Leander, TX) and  
90 Austin Water also provided soft alga genera identifications. While dinoflagellates may also  
91 develop nuisance bloom conditions (Paerl 1988), we excluded them from our analyses as  
92 abundances were relatively homogeneous within and across years (mean  
93  $\pm$  95% confidence interval [CI] =  $3427 \pm 50$  org/mL). Austin Water has established reporting  
94 threshold triggers (i.e., bloom events) of 10,000 org/mL for total algae and 300 org/mL for  
95 cyanobacteria. Above those abundance thresholds, Austin Water has noted increased potential  
96 for taste-and-odor problems in the drinking water, necessitating additional treatment.

97 Texas Parks and Wildlife Department provided submerged aquatic vegetation (SAV) survey  
98 data. Surveys were conducted on Lake Austin approximately every two years prior to invasion  
99 by hydrilla in 1999. After 2002, Texas Parks and Wildlife typically conducted three surveys per  
100 year to evaluate the spread of hydrilla and control efficacy of stocked triploid grass carp  
101 (*Ctenopharyngodon idella* [Valenciennes in Cuvier and Valenciennes]). Surveys were carried  
102 out by boat, and the SAV bed boundaries identified by sight and sonar. Bed area (ha) was  
103 interpolated with geographic information system (GIS) software. Rake samples were randomly  
104 collected during surveys to identify species present and the proportional abundance within the  
105 SAV beds.

## 106 **Data analysis**

### 107 **Data summaries**

108 We utilized a retrospective study approach to summarize and investigate parameters of interest  
109 and to visualize differences between hydrologic periods and relationships among parameters. In  
110 order to compare periods, we calculated means and their 95% CIs for dam discharges, hydraulic  
111 residence time, water chemistry, dissolved oxygen (DO), temperature, Secchi disk transparency,  
112 and chlorophyll-*a* (chl-*a*). Data were summarized by periods, sites, and months, and were also  
113 combined across sites by month and period, across months by site and period, and by period  
114 across sites and months. Linear regression was used to evaluate similarities in discharges from  
115 Mansfield and Tom Miller dams. Nutrient loading rates from Lake Travis to Lake Austin were  
116 estimated by taking the summation of the hypolimnetic concentrations from Lake Travis  
117 multiplied by the average daily discharge from Mansfield Dam on the day of water quality  
118 sampling, and extrapolating that load annually based on the total volume released in each year  
119 against the volume released on days of water quality sampling.

### 120 **Change point analysis**

121 We applied change point analysis to identify when a stochastic process changed between algal  
122 abundance and either Julian Day of the year or the 15 d moving average hydraulic residence time  
123 and 15 d moving average discharge from Tom Miller Dam. For the former, we separated data  
124 into only two periods, 1992–2010 and 2011–2014, and looked at when algal abundance began  
125 significantly increasing (change point 1) and peaked (change point 2). The two non-drought  
126 periods were combined due to overlap in phytoplankton patterns. Change points between  
127 phytoplankton abundance and hydraulic residence time or discharge were determined for the  
128 entire period of observation (1992–2014) only during the growing season (1 May through 30  
129 November) to eliminate seasonality effects (i.e., low winter flows and algal abundances). We

130 used the phytoplankton abundances at the end of the 15 d moving average period. Quantile  
 131 regression was used to estimate the 95<sup>th</sup> percentile of the distribution of the response variable  
 132 (phytoplankton abundance) conditional on residence time or discharge (Cade et al. 1999;  
 133 Koenker and Machado 1999). The 95<sup>th</sup> quantile was chosen because of its relatedness to the  
 134 highest algal abundances and it has been previously shown to better detect trends in algae counts  
 135 than other quantiles (Porras and Richter 2015).

136 The following model was used to find the Julian day change points for total, diatom, and green  
 137 algal abundances:

$$138 \quad \mu = \begin{cases} \alpha & jday < cp1 \\ \alpha + \beta1 \times (jday - cp1) & jday < cp2 \text{ but } \geq cp1 \\ \alpha + \beta2(cp2 - cp1) + \delta1 \times (jday - cp2) & jday \geq cp2 \end{cases} \quad (1)$$

139 Where  $\mu = 95^{\text{th}}$  quantile,  $jday = \text{Julian day}$ ,  $cp1 = \text{change point 1}$ , and  $cp2 = \text{change point 2}$ . For  
 140 cyanobacteria, the second change point fixated at a local peak within the data, necessitating an  
 141 additional change point in the model:

$$142 \quad \mu = \begin{cases} \alpha & jday < cp1 \\ \alpha + \beta1 * (jday - cp1) & jday < cp2 \text{ but } \geq cp1 \\ \alpha + \beta2 * (cp2 - cp1) + \delta1 * (jday - cp2) & jday < cp3 \text{ but } \geq cp2 \\ \alpha + \beta3 * (cp3 - cp2) + \delta2 * (jday - cp3) & jday \geq cp3 \end{cases} \quad (2)$$

143 All  $\alpha$ ,  $\beta$ , and  $\delta$  parameters were modeled using normal distributions with large variance as priors.  
 144 Change points were modeled using uniform distributions ranging from 0 to 365 for  $cp1$ ,  $cp1$  to  
 145 365 for  $cp2$ , and  $cp2$  to 365 for  $cp3$ . Change points were found using Markov chain Monte Carlo  
 146 methods with an initial burn-in of 10,000 iterations and a sample size of 10,000 for estimating  
 147 parameters.

148 To find change points related between 95<sup>th</sup> quantile algal abundance and the moving 15 d  
 149 average residence time and moving 15 d average discharge, we used the approach depicted in  
 150 Killick et al. (2016). This method is useful in finding multiple changes within a series of ordered  
 151 points  $y_{1:n} = (y_1, \dots, y_n)$  by partitioning the sequence into segments with similar statistical  
 152 properties. The segments are defined by their position in the sequence,  $\tau_i$ , for  $i = 1$  to  $m$  change  
 153 points. Thus, one change point, for example, would partition the sequence into the segments  $y_{1:\tau}$   
 154 and  $y_{\tau+1:n}$ . For multiple change points, each position in the sequence,  $\tau_i$ , would be assessed. This  
 155 assessment would be based on minimizing the function:

$$156 \quad \sum_{i=1}^{m+1} [C(y_{(\tau_{i-1}+1):\tau_i})] + \beta f(m) \quad (3)$$

157 Where  $C[\cdot]$  is a cost function for each segment partitioned by some change point position,  $\tau_i$ , and  
 158 the cost function used is a negative log-likelihood. The term  $\beta f(m)$  reflects a penalty function to  
 159 prevent overfitting of change points to the sequence of data.

160 **Bayes' theorem**

161 Bayesian credible intervals were developed for the mean number of days in the growing season  
162 in which total algae or cyanobacteria counts exceeded 10,000 org/mL and 300 org/mL,  
163 respectively. These credible intervals were developed using the OpenBUGS software that  
164 modelled the counts as realization coming from a Poisson distribution with an unknown mean  
165 count. The posterior distribution of mean counts was determined by assuming an uninformative  
166 prior distribution, and computing the likelihood of a Poisson distribution given the data of  
167 seasonal counts. For both total algae and cyanobacteria mean number of days, the counts were  
168 partitioned into two data sets, 1992–2010 and 2011–2014.

169 In order to estimate the probability of a cyanobacterial bloom occurrence based on 15 d average  
170 hydraulic residence time, we applied Bayes' Theorem (Gotelli and Ellison 2004; McCarthy  
171 2007). Mathematically, Bayes' Theorem is:

$$172 \quad Pr(A|B) = (Pr(B|A) \times Pr(A))/Pr(B) \quad (4)$$

173 Where  $Pr(A)$  = probability of event  $A$  (i.e., a bloom event >300 cyanobacteria org/mL),  $Pr(B)$  =  
174 probability of event  $B$  (i.e., a hydraulic residence time greater than an a priori threshold),  
175  $Pr(B|A)$  is the probability of event  $B$  happening given event  $A$ , and  $Pr(A|B)$  = the probability of  
176 event  $A$  happening given event  $B$ . The term  $Pr(B)$  required further derivation for occurrences of  
177 a discharge event when there was not an algae bloom with:

$$178 \quad Pr(B) = Pr(B|A) \times Pr(A) + Pr(B|not A) \times Pr(not A) \quad (5)$$

179 Where  $Pr(not A)$  = the probability of event  $A$  not happening (i.e., cyanobacteria counts < 300  
180 org/mL), and  $Pr(B|not A)$  = the probability of a hydraulic residence time/discharge given no  
181 bloom event.

182 The probabilities for each term in Equations 4 and 5 were estimated from daily records of counts  
183 of cyanobacteria at the end of the 15 d moving hydraulic residence time. For instance, the  
184 percentage of days when bloom events occurred served as proxies for  $Pr(A)$ . Similarly, the  
185 percentage of days when Lake Austin had an estimated hydraulic residence time less than some  
186 threshold for those days that there was a bloom event were substituted into  $Pr(B|A)$ . Within the  
187 growing season across periods 1990–1998, 1999–2010, and 2010–2014,  $n = 965, 1670,$  and  $585,$   
188 respectively.

189 As a check for statistical independence of the data, roughly one-third of the data was randomly  
190 sampled, and the same probabilities were estimated with the smaller set. The result was  
191 differences of  $\pm 3\%, \pm 3\%,$  and  $\pm 10\%$  for 1990–1998, 1999–2010, and 2010–2014, respectively. It  
192 should be noted that the large differences in percent for 2010–2014 was due to the small sample  
193 size when the hydraulic residence time was <30 d. When the probabilities were estimated for an  
194 average residence time >45 d, the difference in probabilities decreased to about  $\pm 2\%$ . Statistical  
195 analyses were run with SAS v. 9.4 (SAS Institute, Inc, Cary, NC, USA) or the  $R$  package  
196 “*change*point” (Killick et al. 2016).

## 197 **Results**

### 198 ***Reservoir levels and discharges***

199 From 1990 to 2010, the Lake Travis water level and stored volume ranged from approximately  
200 45% to >100% of desired capacity (Fig. 2A). Lake Austin elevations were kept relatively  
201 constant ( $\pm 0.3$  m of 150.2 m amsl) aside from periodic draw-downs to help control SAV (Suppl.  
202 Fig. 1A). Prior to onset of the 2011 drought, multi-year declines in Lake Travis' surface water  
203 level were observed in 1995–1997, 1999–2001, 2005–2007, and 2008–2010 (Fig. 2A). After  
204 2011, Lake Travis levels rapidly declined, reaching a low of less than 190 m amsl (~35% of  
205 capacity) in 2013.

206 Mansfield Dam average daily discharge varied substantially within and across months during the  
207 first two periods. Lake Travis discharges averaged within a month across each of the first two  
208 periods followed a seasonal pattern with lowest ( $<10$  m<sup>3</sup>/s) average daily discharges between  
209 October and March. June and July typically had the highest average discharges due to rainfall  
210 and to meet downriver agricultural demands. Despite the inter-annual variability in daily  
211 discharge, average rates in March through September were similar between the first two periods;  
212 rates were higher in October through February in the 1999–2010 period (Fig. 2B). For the period  
213 of drought (2011–2014), average monthly discharges from Lake Travis were significantly lower  
214 than the previous twenty years in May through September, averaging  $<20$  m<sup>3</sup>/s per day. Average  
215 monthly discharges from Lake Austin were to Lake Travis prior to 2011, but between 2011 and  
216 2014 discharges averaged  $<10$  m<sup>3</sup>/s across months (Suppl. Fig. 1B).

217 Between 1990 and 2010, Tom Miller Dam daily discharges tracked discharges from Mansfield  
218 Dam (Tom Miller discharge =  $1.01 * \text{Mansfield discharge} - 2.12$ ;  $r^2 = 0.97$ ). During the  
219 drought, discharges from Lake Travis through Lake Austin were primarily to meet  
220 environmental flow requirements and maintain Lake Austin's elevation, resulting in Lake Austin  
221 discharge rates falling below the 1:1 line (Tom Miller discharge =  $0.793 * \text{Mansfield discharge} -$   
222  $2.398$ ;  $r^2 = 0.82$ ; Fig. 2C).

223 As a result of the lower discharges during the drought period from the Tom Miller dam, average  
224 monthly hydraulic residence time in Lake Austin increased significantly from the previous two  
225 periods (Fig. 2D). Through the first two periods, monthly average hydraulic residence time  
226 during the growing season was typically less than 14 d; the 1999–2010 period averaged only  
227 about one week. During the drought period, monthly estimates of residence time increased two-  
228 to eight- fold (Fig. 2D).

## 229 *Lake Travis hypolimnetic water chemistry and loading rates*

230 Among months across time periods, average concentrations of NH<sub>3</sub>-N (Fig. 3A) and NO<sub>x</sub>-N (Fig.  
231 3B) were highest (0.27–0.50 mg/L) and lowest (<0.02 mg/L), respectively, in October. Total  
232 Kjeldahl N concentrations constituted approximately 80% of the TN concentration across  
233 months and years (not shown). Within a period, monthly concentrations of TN (means: 0.50–  
234 0.91 mg/L; Fig. 3C) and TP (means: 0.02–0.12 mg/L; Fig. 3D) were similar. However, TP  
235 concentrations were generally higher during the non-drought periods, contributing to lower  
236 average molar N:P ratios (10–30) relative to the drought period (30–60; Fig. 3E).

237 Despite similar average monthly discharges between the periods of 1990–1998 and 1999–2010  
238 (Fig. 2C), estimated average annual nutrient loading estimates during the latter period declined  
239 by between 5x (TN) to over 100x (TP; Fig. 3F). During the drought, estimated average annual  
240 nutrient loading rates declined approximately an order of magnitude for NH<sub>3</sub>-N, NO<sub>x</sub>-N, and TN,  
241 and nearly two orders of magnitude for TP, relative to the previous two decadal periods.

## 242 *Lake Austin surface water quality*

243 Between sites, for all months and periods, excluding Chl-*a*, water chemistry did not consistently  
244 differ (Fig. 4A–F). For all sites and periods, bi-monthly NH<sub>3</sub>-N concentrations were greatest in  
245 summer and fall (>0.05 mg/L; Fig. 4G) whereas NO<sub>x</sub>-N concentrations were lowest in August  
246 and October (<0.1 mg/L; Fig. 4H). Total N concentrations were similar across months (~0.5  
247 mg/L; Fig. 4I). Total P concentrations typically peaked in October (0.08 mg/L; Fig. 4J),  
248 contributing to the lowest seasonally observed molar N:P (~50; Fig. 4K) and NO<sub>x</sub>-N:P (~2; Fig.  
249 4L) ratios. Among periods for all sites and months, average concentrations of NH<sub>3</sub>-N for 1999–  
250 2010 and 2011–2014 were similar, but both periods were lower than 1992–1998 (Fig. 4M).  
251 Concentrations of NO<sub>3</sub>-N (Fig. 4N), TN (Fig. 4O), and TP (Fig. 4P) were lowest during the  
252 drought period resulting in the greatest observed molar N:P (> 100; Fig. 4Q) and NO<sub>x</sub>-N:P (>20;  
253 Fig. 4R) ratios.

254 Chlorophyll *a* concentrations increased moving down-river, though the mid-reservoir site 12297  
255 had greatest Secchi disk transparency (Fig. 5A and B, respectively). Across months, Chl-*a*  
256 concentration peaked at ~3.5 µg/L in October (Fig. 5C) whereas bi-monthly Secchi disk  
257 transparency generally remained similar (Fig. 5D). Drought-period chl-*a* concentrations were  
258 more than double the non-drought periods (Fig. 5E). Greatest Secchi disk transparency depths  
259 (approx. 3.5 m) generally occurred during the period of 1992–1998 (Fig. 5F). See supplement  
260 Fig. 2 and 3 for additional details.

## 261 *Vertical profiles and hypolimnetic chemistry*

262 Between 1992 and 2010 at site 12294, April vertical temperature differences were minimal (Fig.  
263 6A). Stratification of the water column was typically evident by June (Fig. 6B). By August,  
264 epilimnetic water temperatures were typically greater than 25 C (Fig. 6C). By the October  
265 sampling, the water column was isothermal (Fig. 6D). During the drought, stratification became  
266 established earlier (Fig. 6A), and epilimnetic depth was greater during the drought period (Fig.  
267 6B & C). Water temperatures across all months were warmer, with the largest difference, 7 C,  
268 observed in June (Fig. 6B), historically when hypolimnetic discharges from Lake Travis into  
269 Lake Austin were largest (Fig. 2B).

270 Between 1992 and 2010, April and June vertical DO profiles were relatively homogenous (Fig.  
271 6E & F, respectively). In August, a hypoxic hypolimnion was typically observed (Fig. 6G), but  
272 waters were oxygenated and well mixed again by the October sampling (Fig. 6H). During the  
273 drought period, a distinctive oxicleine was evident in April (Fig. 6E). June (Fig. 6F) and August  
274 (Fig. 6G) hypolimnetic DO was typically less than 2 mg/L below 6 m depth; DO concentrations  
275 >5 mg/L were limited to the epilimnion. Despite warmer water temperatures and low discharges,  
276 turnover and oxygenation of the water column was still evident by October (Fig. 6H).

277 With the exception of NO<sub>x</sub>, there were no seasonal or drought and non-drought period patterns in  
278 water chemistry patterns near the bottom (~14 m) of site 12294 (not shown). Seasonal changes in  
279 NO<sub>x</sub> concentrations matched those of the epilimnion.

## 280 *Phytoplankton abundance*

281 Total phytoplankton abundance between 1992 and 2010 exceeded the reporting bloom threshold  
282 of 10,000 org/mL on only 1.1% of the measurement days (Fig. 7A). We generally observed  
283 greater total algal biomass in years of lower Lake Travis elevation and average daily discharges.  
284 For the non-drought period of high discharges, we estimated an increasing likelihood of there  
285 being less than 8 d experiencing a bloom in the growing season (Suppl. Fig. 4A). Conversely,  
286 between 2011 and 2014, nearly 30% of the growing season days with counts experienced a  
287 bloom, and four of the five greatest phytoplankton cell abundances were recorded (Fig. 7A).  
288 During the drought we estimated an increased likelihood of over 30 d during the growing season  
289 experiencing a bloom (Suppl. Fig. 4A). We estimated the timing of total phytoplankton  
290 abundance increase and peak in non-drought years at Julian d 236 and 270, respectively (Fig.  
291 8A). Between 2011 and 2014, total phytoplankton abundance initiation and peak (Julian d 170  
292 and 199, respectively) were earlier than the non-drought period.

293 Green algal abundance before 2011 typically peaked below 1500 org/mL, with periodic spikes  
294 (Fig. 7B). The two largest green algal biomass events occurred in 2013 and 2014. Green algal  
295 abundance during the drought period began increasing later in the year than under non-drought  
296 conditions (Julian d 113 vs 73, respectively), but peaked earlier during the drought (Julian d 182)  
297 than non-drought (Julian d 204) period (Fig. 8B).

298 Diatom abundance infrequently exceeded 1000 org/mL prior to 2011, but peak abundances  
299 increased up to an order of magnitude during the drought period (Fig. 7C). We estimated diatom  
300 abundance between 1992 and 2010 began increasing very early in the year (Julian d 68) and on  
301 average did not peak until early August (Julian d 222; Fig. 8C). However, between 2011 and  
302 2014, biomass increased later (Julian d 167) and quickly peaked (Julian d 202).

303 Cyanobacteria abundance prior to 2011 generally spiked coincident with years when Lake Travis  
304 elevation and discharge declined (e.g., 1996, 2006–2007, 2009; Fig. 7D). However, the largest  
305 recorded abundance, 20,000 org/mL, occurred in 2013. Between 2011 and 2014, cyanobacteria  
306 were considered “blooming” (i.e., >300 org/mL) on 356 d, compared with only 374 d between  
307 1992 and 2010. In non-drought years, we estimated their potentially being between 10 and 40  
308 growing season days exceeding Austin Water’s reporting criteria (Suppl. Fig. 4B). Conversely,  
309 under sustained low discharges, upwards of 120 d may exceed reporting thresholds. Prior to  
310 2011, cyanobacteria began blooming in early August (Julian d 222) and peaked in early October  
311 (Julian d 277; Fig. 8D). During the drought, cyanobacteria bloom initiation was almost two

312 months earlier and peaked ~20 d later than the non-drought period. The dominant  
313 cyanobacterium during blooms was *Oscillatoria* spp.

### 314 ***Flushing rates and phytoplankton abundance***

315 Across all years we found significant change points in algal abundance associated with hydraulic  
316 residence (Fig. 9) and discharge volumes (Suppl. Fig. 5; a full list of change points identified can  
317 be found in supplemental Table 1). Below a hydraulic residence time of 8 d (or average  
318 discharges of ~26 m<sup>3</sup>/s), total algal abundance records for Lake Austin were largely below the  
319 reporting threshold of 10,000 org/mL (Fig. 9A). We observed the largest phytoplankton  
320 abundances when residence times exceeded four weeks (discharges below ~6 m<sup>3</sup>/s; supplemental  
321 Fig. 5A).

322 Green algae abundance records indicated a significant change when hydraulic residence times  
323 were above one week, and greatest cell volumes were observed when flushing rates exceeded  
324 two weeks (average discharges below 23 m<sup>3</sup>/s; Fig. 9B and supplemental Fig. 5B).

325 Diatom abundances were most responsive to moderate hydraulic residence times (e.g., 21 to 68  
326 d; Fig. 9C). Abundance records rapidly declined when average hydraulic residence times were  
327 less than 21 d (>~13 m<sup>3</sup>/s) but also when flushing rates exceeded ~3 months (or discharges <~3  
328 m<sup>3</sup>/s; supplemental Fig. 5C). Diatom abundance records peaked when the 15 d average hydraulic  
329 residence times were ~40 d (Fig. 9C).

330 Cyanobacteria records indicated rapid increases above 450 org/mL when average hydraulic  
331 residence times in Lake Austin exceeded 6 d (Fig. 9D). Average discharge rates below 8 m<sup>3</sup>/s,  
332 were associated with the largest recorded abundances (supplemental Fig. 5D; supplemental Table  
333 1). Across all years, the probability of a bloom event increased as the average hydraulic  
334 residence time exceeded three weeks, and peaked at an ~20% probability (supplemental Fig. 6).  
335 When partitioning by periods, the probability of a bloom was generally less than 15% in non-  
336 drought years, regardless of residence time. During the drought period, the probability of a  
337 bloom more than doubled as residence time increased from 15 to 45 d, and the probability  
338 peaked at approximately 60% (Suppl. Fig. 6).

### 339 ***Submerged aquatic vegetation***

340 Prior to 1999, SAV was primarily comprised of Eurasian watermilfoil (*Myriophyllum spicatum*  
341 L.) with a limited extent in the upriver third of the reservoir (Fig. 10). After 1999, hydrilla  
342 periodically dominated the SAV community; coverage was highly variable after 2002 due to  
343 triploid grass carp stocking efforts. After 2011, areal coverage of hydrilla rapidly increased and  
344 peaked at over 250 ha in 2013. Herbivores eventually achieved total control of hydrilla, and no  
345 SAV has been observed in Lake Austin after Jun 2014 (Fig. 10).

### 346 **Discussion**

347 In the Lake Austin reservoir, we showed how water chemistry constituents and phytoplankton  
348 succession and biomass dynamics changed due to seasonality and drought-induced discharge  
349 reductions. We determined that growing season low discharge rates were one of the most  
350 important factors influencing reservoir physicochemical dynamics which altered the timing,

351 probability, and magnitude of phytoplankton blooms. From this, based on spring and summer  
352 flushing rates managers will be better able to anticipate the timing and potential magnitude of  
353 hypoxia and phytoplankton blooms that can negatively impact municipal drinking water quality.

354 Across hydrologic periods, Lake Travis hypolimnetic water chemistry seasonal patterns were  
355 consistent with anoxic and microbial nutrient cycling dynamics. The late-summer peak of TP  
356 concentration is expected from liberation of sediment P during anoxia, and NH<sub>3</sub>-rich and NO<sub>3</sub><sup>-</sup>-  
357 poor waters are likely to be driven by microbial ammonification and denitrification, respectively  
358 (Wetzel 2001). Concomitant with lower discharges from Lake Travis during the drought period,  
359 nutrient loading to Lake Austin significantly declined. Bi-monthly surface chemistry patterns  
360 reflected Lake Travis hypolimnetic chemistry. However, we found Lake Austin surface water  
361 nutrient chemistry varied little longitudinally despite the upriver loading and 33 km reservoir  
362 course, likely due to typically rapid pulsing and narrow channel morphology (Thornton et al.  
363 1990). Lake Austin's longitudinal dynamics were likely also confounded by sediment  
364 resuspension dynamics affected by grass carp grazing, SAV distribution/elimination, and  
365 increasing recreational activities (Maceina et al. 1992; Cunha et al. 2014). Lake Austin  
366 hypolimnetic water chemistry patterns could have been obfuscated by bottom waters being  
367 hypoxic rather than anoxic, coupled with potentially rapid cycling and uptake that would not be  
368 captured by the bi-monthly sampling regime (Bormans et al. 2015).

369 Increased water residence times contribute to warmer waters which enhance vertical water  
370 column stability (Mitrovic et al. 2011; Huber et al. 2012). In Lake Austin, drought period waters  
371 were warmer earlier, increasing the duration of stratification and deep water hypoxia (e.g., DO  
372 concentrations <2 mg/L). Greater hypolimnetic thickness could promote the accumulation of  
373 reduced compounds that contribute to taste-and-odor problems, and increases the likelihood of  
374 intake by a drinking water utility (Beutel 2003; Davis J, Austin Water, Feb. 2017, pers. comm.).  
375 Greater diatom abundances observed during the drought may have also played a role in  
376 enhancing deep water hypoxia earlier in the summer as greater cell sedimentation over a short  
377 period of time would increase biological oxygen demand (Reavie et al. 2016a).

378 Diatom abundance increased early in the growing season due to cool water temperatures,  
379 abundant nutrients (and presumably silicates), and turbulence maintaining cells in the photic  
380 zone (Sommer et al. 1986; Jacquet et al. 2005; Domingues and Galvão 2007). During the  
381 drought, the duration of diatom blooms was short and abundances were up to an order of  
382 magnitude greater relative to non-drought years, indicating that flushing negatively impacted  
383 diatoms biomass in non-drought years. Salmaso and Zignin (2010) also documented a negative  
384 association between diatom abundances and river discharge rates. Extended stratification and  
385 warmer waters can significantly enhance small diatom taxa biomass (Reavie et al. 2016b);  
386 however, without species identifications, we do not know if community composition shifted  
387 between periods. However, we also found that extended periods of hydraulic stability (i.e., >68  
388 d) negatively impacted diatom abundances, likely due to cell settling in the prolonged period  
389 absent of vertical turbulence (Rühland et al. 2015). Green algal dynamics generally matched  
390 diatoms in timing of biomass accumulation and peak, but were a small component of the  
391 phytoplankton community, likely due to P-limitations (see below; Sommer et al. 1986; Wetzel  
392 2001).

393 Cyanobacteria abundance peaked in late summer across years. In lentic systems, conditions  
394 conducive to cyanobacteria bloom development, including N-limitations and P-enrichment,

395 water temperatures exceeding 25 C, and low mixing and flushing rates that promote stratification  
396 and vertical stability, typically occur in late summer (Soranno 1997; Ferber et al. 2004; Huber et  
397 al. 2012; Reichwaldt et al. 2015; Yasarer and Sturm 2016). For example, as N becomes limiting  
398 cyanobacteria, including *Oscillatoria* which was prominent in Lake Austin, may fix atmospheric  
399 N and migrate vertically to scavenge  $\text{PO}_4^-$  and  $\text{NH}_4^+$  from the metalimnetic entrainment zone  
400 (Paerl 1988; Gallon et al. 1991; Bormans et al. 1999; Jacquet et al. 2005). In Lake Austin TN did  
401 not vary seasonally, but when  $\text{NO}_x\text{-N}$  concentrations were near their minimum and TP  
402 concentrations were maximal, cyanobacteria peaked. The number of “low nitrate days” (e.g.,  
403  $<0.1 \text{ mg NO}_3\text{-N/L}$ ; Nürnberg 2007) and a molar  $\text{NO}_x\text{-N:P}$  ratio below 11 (with water  
404 temperatures  $>21 \text{ C}$ ; Pick and Lean 1987) have been linked to cyanobacteria bloom magnitude.  
405 Lake Austin epilimnetic  $\text{NO}_x\text{-N}$  concentrations and molar  $\text{NO}_x\text{-N:P}$  were typically at those  
406 thresholds by the August sampling. Lower  $\text{NO}_x\text{-N}$  loadings during the drought period from Lake  
407 Travis could have helped promote the observed earlier bloom onset; higher resolution monitoring  
408 is needed to better link  $\text{NO}_x\text{-N}$  dynamics with cyanobacteria abundances.

409 Epilimnetic TN and TP concentrations and ratios can also influence cyanobacteria bloom  
410 development and toxicity (Nõges et al. 2008; Grantz et al. 2014; Paerl et al. 2014; Persaud et al.  
411 2014). Regardless of hydrologic period, Lake Austin TN concentrations generally fell within the  
412 thresholds associated with a low bloom risk (e.g.,  $300\text{--}700 \mu\text{g/L}$ ; Yuan and Pollard 2015). Prior  
413 to the drought, TP concentrations typically exceeded  $20 \mu\text{g/L}$  which has also been linked to  
414 increased risk of cyanobacterial blooms (Paerl et al. 2014; Yuan and Pollard 2015; Pick 2016).  
415 However, blooms were infrequent in non-drought years, further substantiating the importance of  
416 discharge rates in the inhibition of cyanobacteria biomass accumulation. Fortuitously, during the  
417 drought when environmental conditions were favorable for cyanobacteria blooms, TP  
418 concentrations in Lake Austin declined ( $\bar{x} \sim 13 \mu\text{g/L}$ ), and molar N:P ratios remained P-limited  
419 (i.e.,  $>22$ ; Guildford and Hecky 2000), though still favorable to cyanobacteria (Pick and Lean  
420 1987; Nõges et al. 2008). The increasing P-limitation may have been a boon to *Oscillatoria*  
421 (Jacquet et al. 2005), a known toxin producer (Hitzfeld et al. 2000). Toxin production is typically  
422 observed in eutrophic systems (i.e., both N- and P- rich; Paerl et al. 2016), and Lake Austin  
423 blooms thus far have not been found to be toxic (Maddox C, Austin Water, Oct 2016, pers.  
424 comm.; Bellinger B, Watershed Protection, Dec 2016, unpubl. data). The peak biomass  
425 experienced to date for this study ( $20,000 \text{ org/mL}$ ) is considered a low health risk to recreational  
426 users (WHO 2003). We hypothesize that the absence of toxic cyanobacteria from Lake Austin,  
427 notably during the drought, was largely driven by P-limitations.

428 In non-drought years the typically small magnitude and probability of cyanobacteria blooms  
429 were reflective of the short period of favorable physicochemical conditions coupled with water  
430 chemistry (Bowes et al. 2016). For example, water temperatures did not exceed  $25 \text{ C}$  until late  
431 summer, and growing season hydraulic residence times were typically much less than three  
432 weeks which will limit cyanobacteria biomass accumulation (Thornton et al. 1990; WHO 2011;  
433 Huber et al. 2012; Londe et al. 2016). After bloom initiation, cyanobacteria had a relatively short  
434 growth period before cooling of waters and vertical mixing of the water column. The limited  
435 duration of optimal growth conditions driven by high flushing rates fits our estimation that in  
436 normal water provisioning years there would be  $<40 \text{ d}$  experiencing a cyanobacterial bloom, with  
437 only a 10–15% probability of occurrence on a given day, even at lower flushing rates.

438 Conversely, maximum observed cyanobacteria abundances occurred during the drought period  
439 when hydraulic residence times significantly increased. The absence of sustained flushing  
440 discharges led to development of favorable physicochemical conditions earlier in the summer,  
441 contributing to bloom initiation being nearly two months earlier than during the non-drought  
442 period. As the 15 day moving average hydraulic residence time exceeded one month, we  
443 estimated there being over a 50% chance of a given day being in a bloom, with upwards of a  
444 total of 120 d exceeding bloom thresholds. Infrequent large discharges would be unable to  
445 sufficiently cool water temperatures, de-stratify the water column, or flush biomass (Cooke et al.  
446 1993; Huber et al. 2012; Yasarer and Sturm 2016). Ample water volume is therefore required for  
447 managers to both achieve and continuously maintain (e.g., >14 d) critical discharge velocities in  
448 order to disrupt a cyanobacterial bloom (Mitrovic et al. 2011; Cha et al. 2014; Bowes et al.  
449 2016). If flow-regulation was not an option due to drought, disruption of water column stability  
450 could be achieved through alternative methods (Bormans et al. 2015). For example, artificial  
451 mixing of Nieuwe Meer, Netherlands, with compressed air bubble plumes for two weeks  
452 eliminated lake stratification, aerated the anoxic hypolimnion, and prevented noxious  
453 *Microcystis* blooms (Visser et al. 1996).

454 While SAV re-establishment is important to the fisheries and water clarity of Lake Austin, it  
455 remains unresolved if during low discharge conditions a robust SAV community would  
456 substantially inhibit large phytoplankton blooms in the lower section of the reservoir. Prior to the  
457 drought, spikes in algal biomass and decreased Secchi disk transparency were observed,  
458 primarily between 1999 and 2010 relative to 1992–1998, despite generally greater SAV  
459 abundances after the invasion by hydrilla. A confounding factor is grass carp grazing which can  
460 negatively influence transparency (Pípalová 2006), and promote phytoplankton biomass  
461 accumulation (Maceina et al. 1992). During the drought period Chl-*a* concentration in Lake  
462 Austin increased at all sites regardless of SAV coverage. And, in 2012, the largest hydrilla extent  
463 in the upper reservoir was observed (i.e., >250 ha or ~31% of the total reservoir area), while the  
464 second highest total algae and cyanobacteria abundances to date also were measured. Lake  
465 Austin's bathymetry and internal connectivity driven by its' narrow channel make it possible that  
466 during periods of low flows, alternative stable states, i.e., an SAV-dominated clear-water and  
467 phytoplankton-dominated turbid-water state, can occur side-by-side (Scheffer and Van Nes  
468 2007; Janssen et al. 2014).

469 Management plans developed by water provisioning authorities will continue to evolve to ensure  
470 demands can be met in the face of uncertain climatic conditions and basin inflows. However,  
471 from the modeling methods and outputs demonstrated in this study (e.g., Fig. 9, supplemental  
472 Fig. 5 & 6), consideration of the ecological impacts of modified water flows could be  
473 incorporated into management plans towards the protection of water quality. For example, in the  
474 Texas Colorado River basin, beyond conservation strategies the LCRA is constructing two new  
475 off-channel reservoirs in the lower watershed that will provide water storage for agricultural  
476 customers ([http://www.lcra.org/about/newsroom/news-releases/Pages/LCRA-receives-8-million-  
477 grant-for-second-new-reservoir.aspx](http://www.lcra.org/about/newsroom/news-releases/Pages/LCRA-receives-8-million-grant-for-second-new-reservoir.aspx)). As a result, even in the absence of drought conditions,  
478 lower summer discharges should be anticipated for the Highland Lakes. Wastewater reuse is also  
479 becoming more widespread. While that can further reduce demands on drinking water supplies,  
480 land application within riparian corridors could increase nutrient loadings to receiving systems  
481 (Porras et al. 2016). Additionally, zebra mussels (*Dreissena polymorpha*) have recently invaded  
482 the Highland Lakes, and their presence in low nutrient lakes (TP < 25 ug/L) has been found to

483 enhance prevalence and toxicity of the cyanobacterium *Microcystis aeruginosa* (Raikow et al.,  
484 2004; Knoll et al. 2008). Continued monitoring with expansion of high frequency sampling is  
485 needed to further improve phytoplankton bloom predictions to better inform adaptive  
486 management strategies necessary to ensure protection of water quality (Dodds et al. 2012; Marcé  
487 et al. 2016).  
488

## 489 **Recommendations and future research**

- 490 • During the growing season, collect higher resolution (e.g., daily, weekly) water chemistry  
491 (e.g., temperature, nitrate, ammonium, total phosphorus) data
- 492 • Expand and refine phytoplankton bloom models incorporating more potential drivers  
493 (e.g., temperature, nutrients)
- 494 • Vertical sampling of the epilimnion and metalimnion for algal biomass to determine  
495 vertical heterogeneity. It is possible that surface (0.3 m) samples are biasing  
496 understanding of phytoplankton biomass and bloom dynamics
- 497 • Diatom species ID to determine if the species composition and sizes change under  
498 drought and non-drought conditions
- 499 • Expand understanding of the relationship between aquatic vegetation, clarity, and  
500 phytoplankton biomass as plant biomass recovers in Lake Austin
- 501 • Continue monitoring cyanobacterial blooms for toxins
- 502 • Expansion of collaborations with LCRA and Austin Water to enhance management,  
503 monitoring, and modeling of future water quality conditions based on changing flow  
504 regulations, climate, and development in the watershed
- 505 • Determine how much nutrient loading to Lake Austin is contributed by Lake Travis and  
506 how much is contributed from other sources such as wastewater reuse, septic tank leaks,  
507 atmospheric deposition, or inflow from Austin creeks
- 508 • Determine rates of N and P release from lake sediments
- 509 • Determine how changes in Lake Austin impact Lady Bird Lake
- 510 • Monitor for zebra mussel population increase and determine if a future infestation will  
511 affect Lake Austin's phytoplankton biomass

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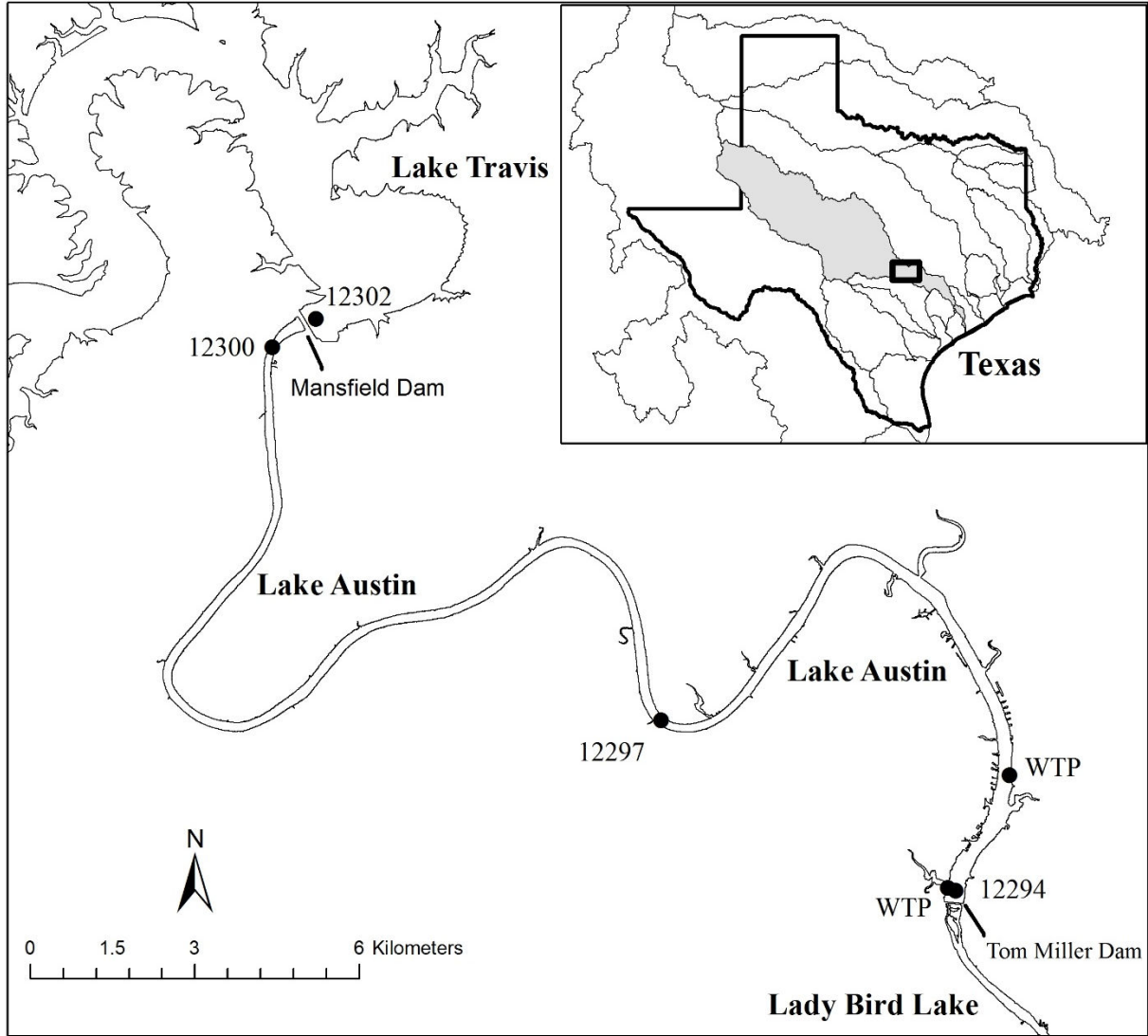
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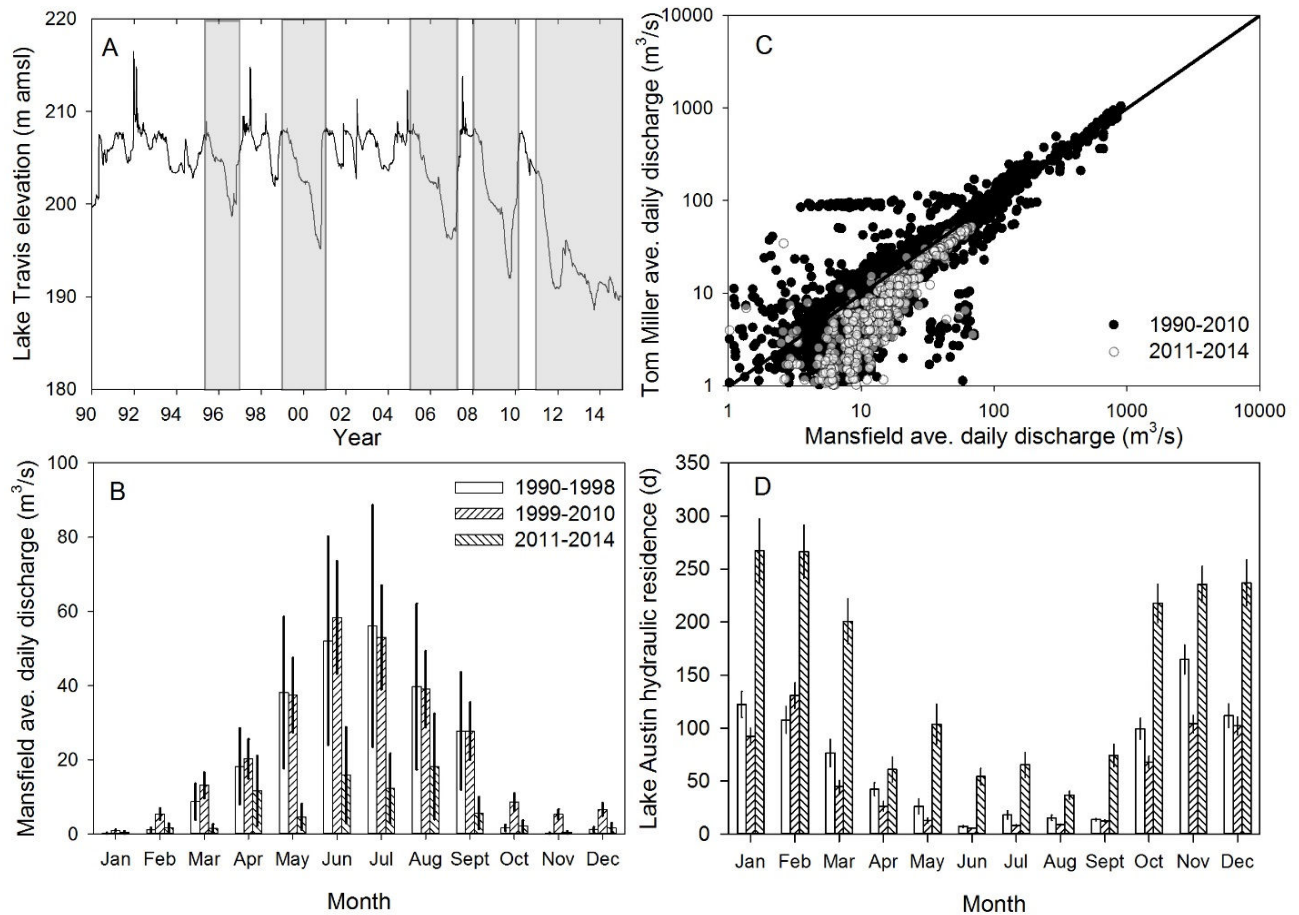
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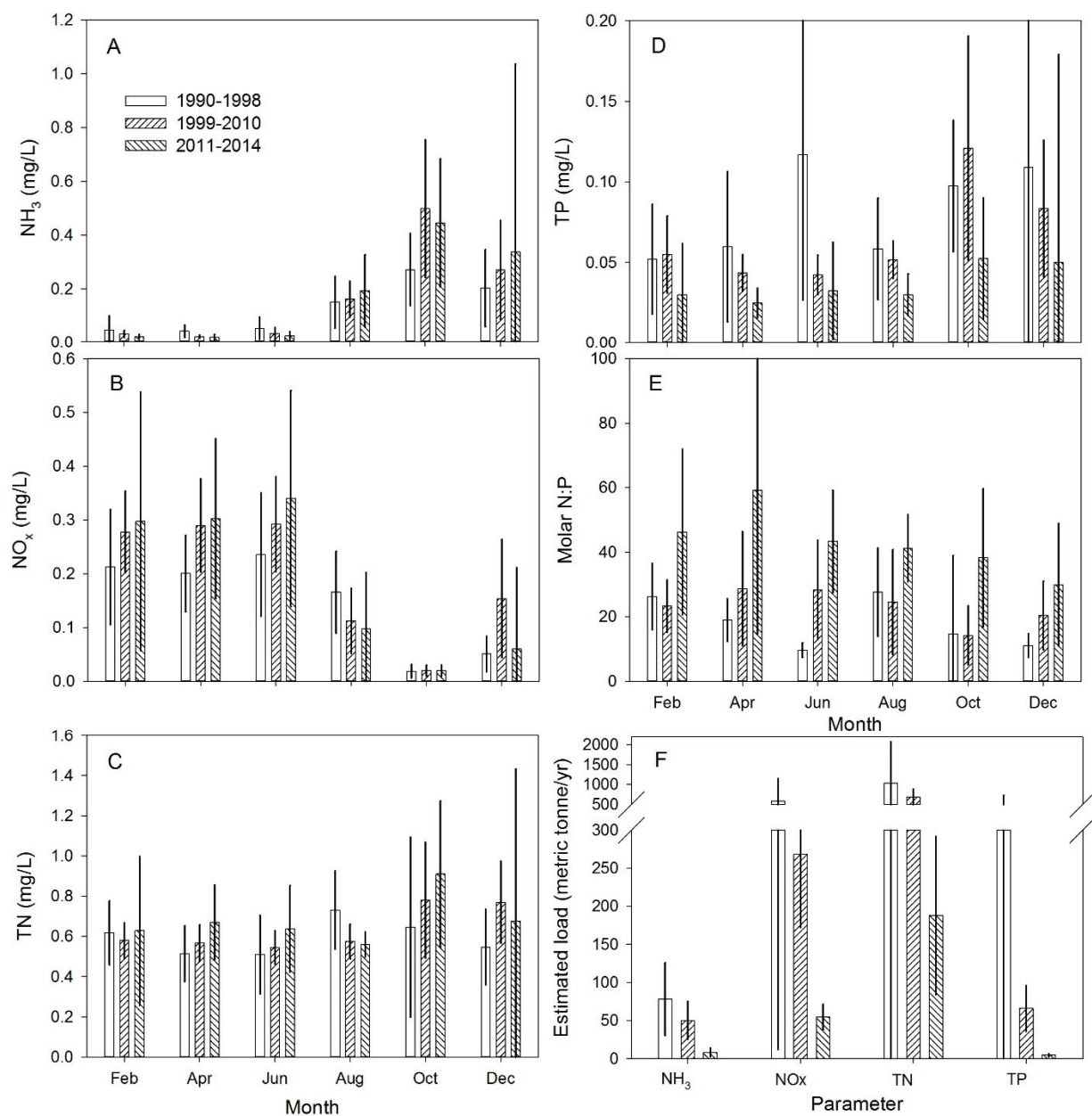
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Figure 1. Location of sampling sites in Lake Travis (12302) and Lake Austin (12300, 12297, 12294), and the primary drinking water treatment plants (WTPs) for the city of Austin. Inset map shows basin location (gray) and location of the reservoirs (box) within the state of Texas, USA.



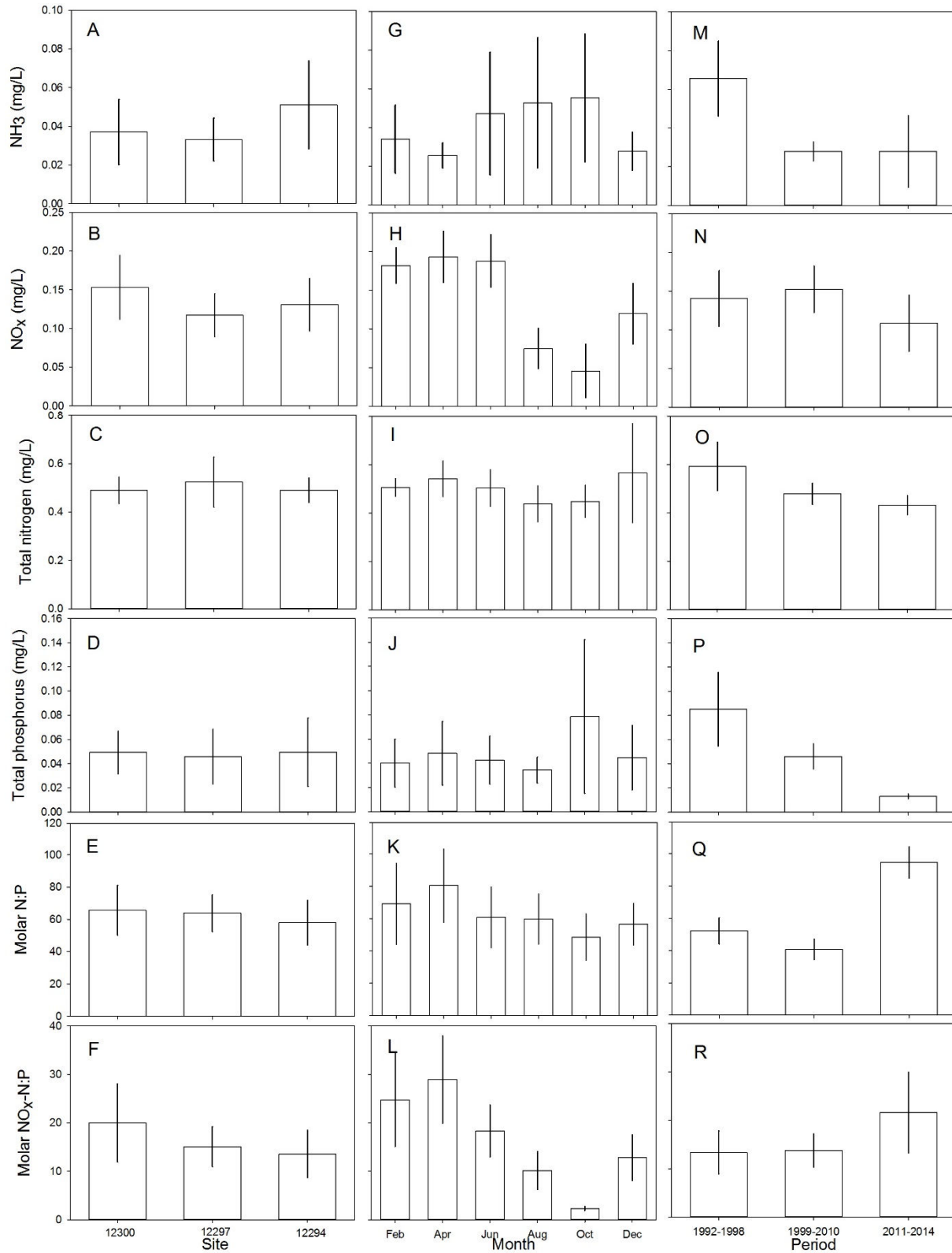
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Figure 2. A) Lake Travis daily elevations at Mansfield dam (meters above means sea level [m amsl]). Gray areas represent multi-year periods of Lake Travis elevation declines and lower average daily discharges. B) Monthly mean  $\pm$  95% confidence intervals of daily discharge volumes from the Mansfield dam, grouped into periods. The periods of 1990–1998 (open bars) and 1999–2010 (bars with forward slash) represent periods of normal reservoir management; 2011–2014 (bars with back slash) was a period of drought-induced water conservation. C) Comparison of  $\log_{10}$  daily average discharge volumes from the Mansfield Dam (x-axis) and Tom Miller Dam (y-axis) during years of normal reservoir management (closed circles; 1990–2010) and drought-induced water conservation (open circles; 2011–2014); a 1:1 line is shown for reference. D) Monthly mean  $\pm$  95% confidence intervals of daily hydraulic residence time (days) in Lake Austin, grouped into periods. Bar fills same as in panel B.



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Figure 3. Bi-monthly Lake Travis hypolimnetic (site 12302, ~1 m above bottom) mean  $\pm$  95% confidence intervals for nutrient chemistry (A–E). F) estimated loading (metric tonnes/yr) of nutrients to Lake Austin. The periods of 1990–1998 (open bars) and 1999–2010 (bars with forward slash) represent periods of normal reservoir management; 2011–2014 (bars with back slash) was a period of drought-induced water conservation.

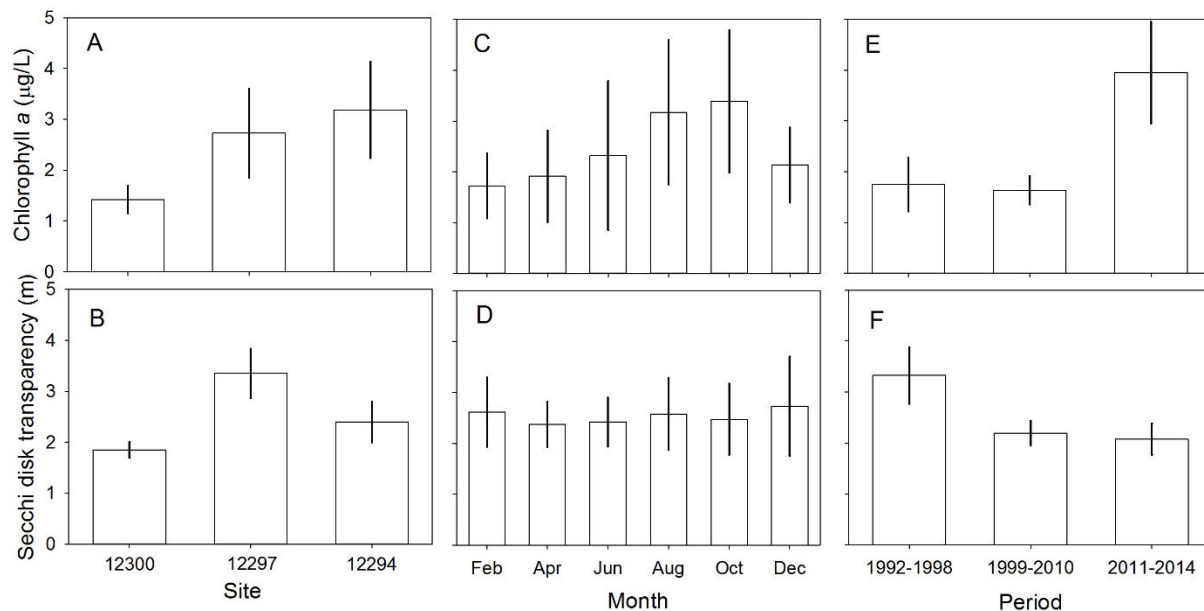


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Figure 4. Lake Austin surface (0.3 m depth) mean  $\pm$  95% confidence intervals for nutrient chemistry parameters comparing: sites, with means pooled across all months and periods (left

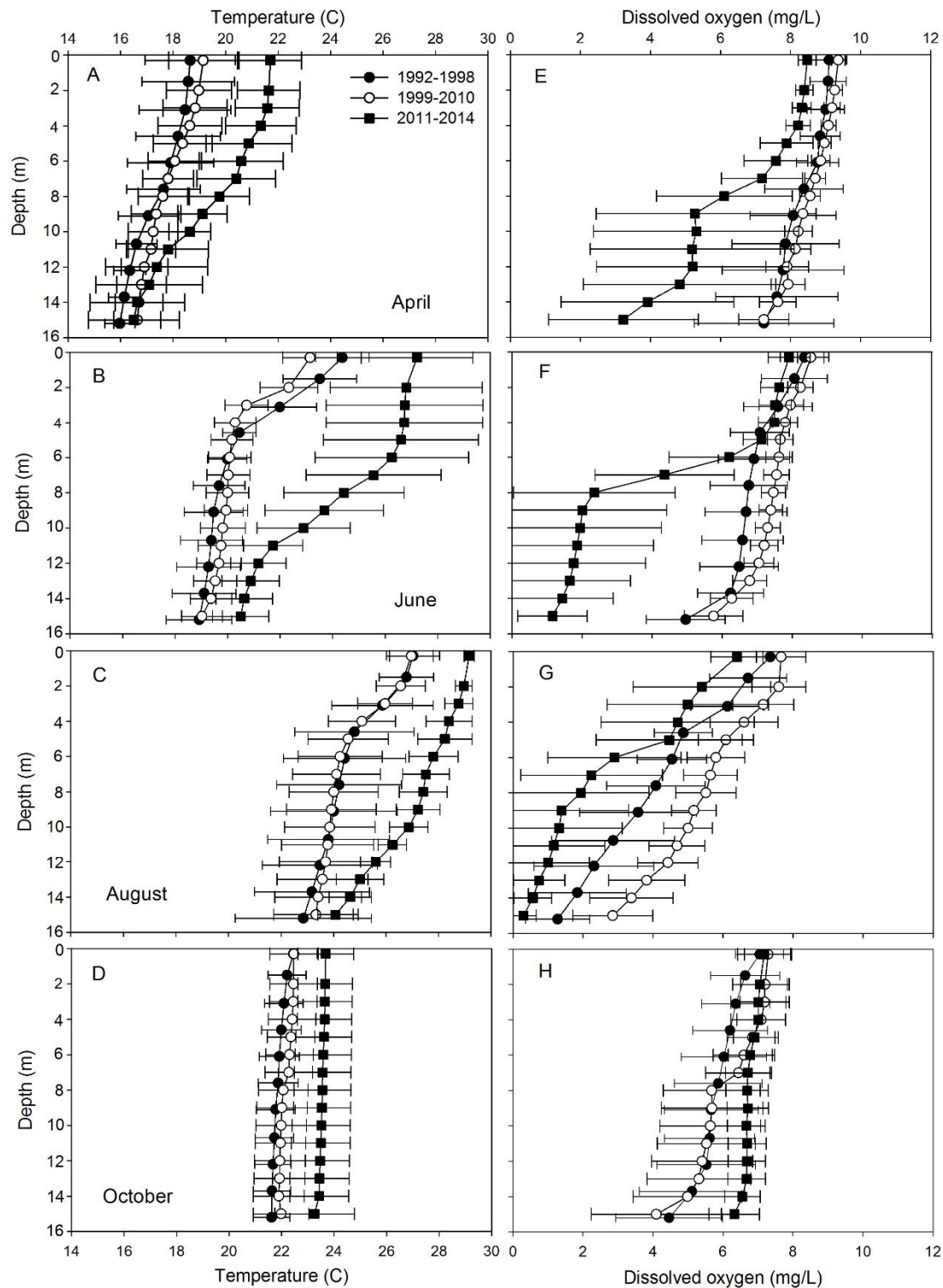
741 column); months, with means pooled across all sites and periods (center column); and periods,  
742 with means pooled across all sites and months (right column). The periods of 1992–1998 and  
743 1999–2010 represent periods of normal reservoir management; 2011–2014 (bars with back slash)  
744 was a period of drought-induced water conservation. Mean period comparison at each site from  
745 bi-monthly samplings can be found in supplemental figure 2.

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Figure 5. Lake Austin mean 95% ± confidence intervals for surface (0.3 m depth) chlorophyll *a* (µg/L; top row) and Secchi disk transparency (m; bottom row) comparing: sites, with means pooled across all months and periods (left column); months, with means pooled across all sites and periods (center column); and periods, with means pooled across all sites and months (right column). The periods of 1992–1998 and 1999–2010 represent periods of normal reservoir management; 2011–2014 was a period of drought-induced water conservation. Mean period comparison at each site from bi-monthly samplings can be found in supplemental figure 3.

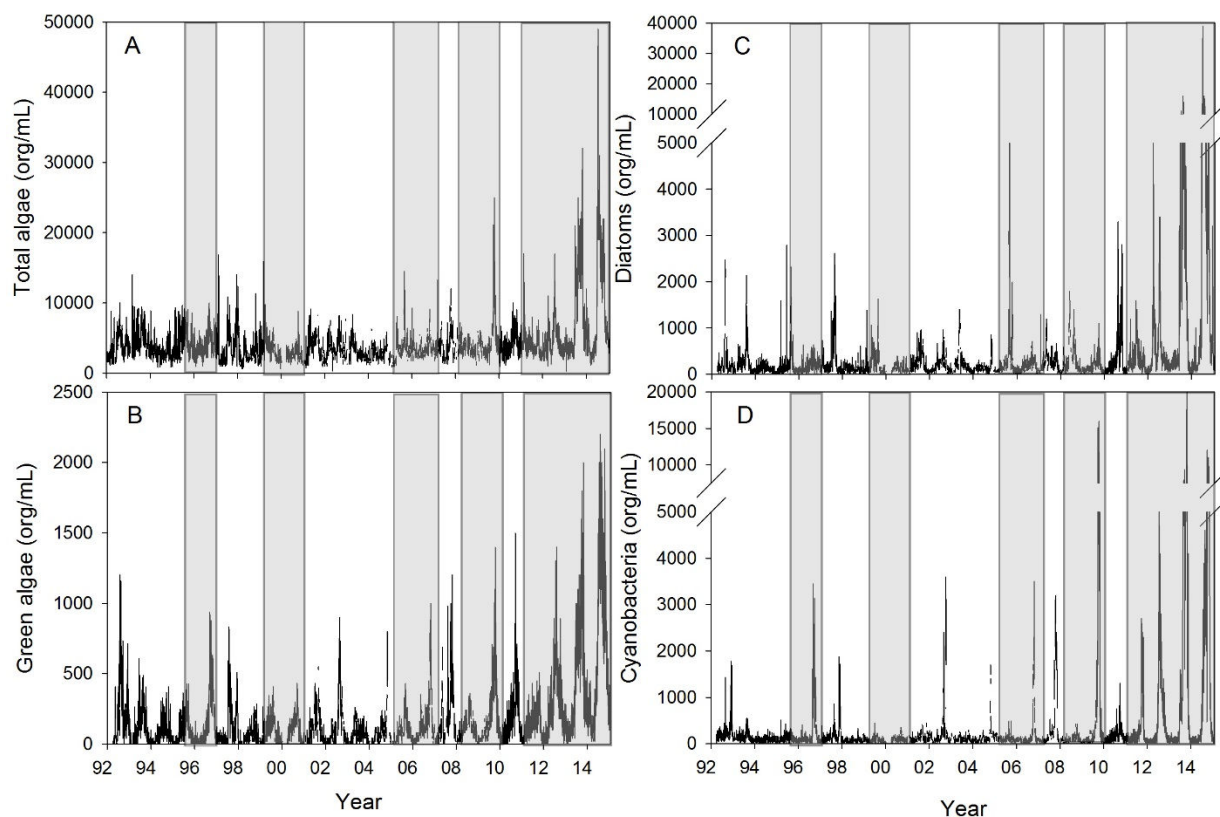


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Figure 6. Lake Austin site 12294 mean  $\pm$  95% confidence intervals for bimonthly vertical profiles of temperature (C; left column) and dissolved oxygen concentrations (mg/L; right column) grouped into periods (1992–1998, closed circles; 1999–2010, open circles; and 2011–

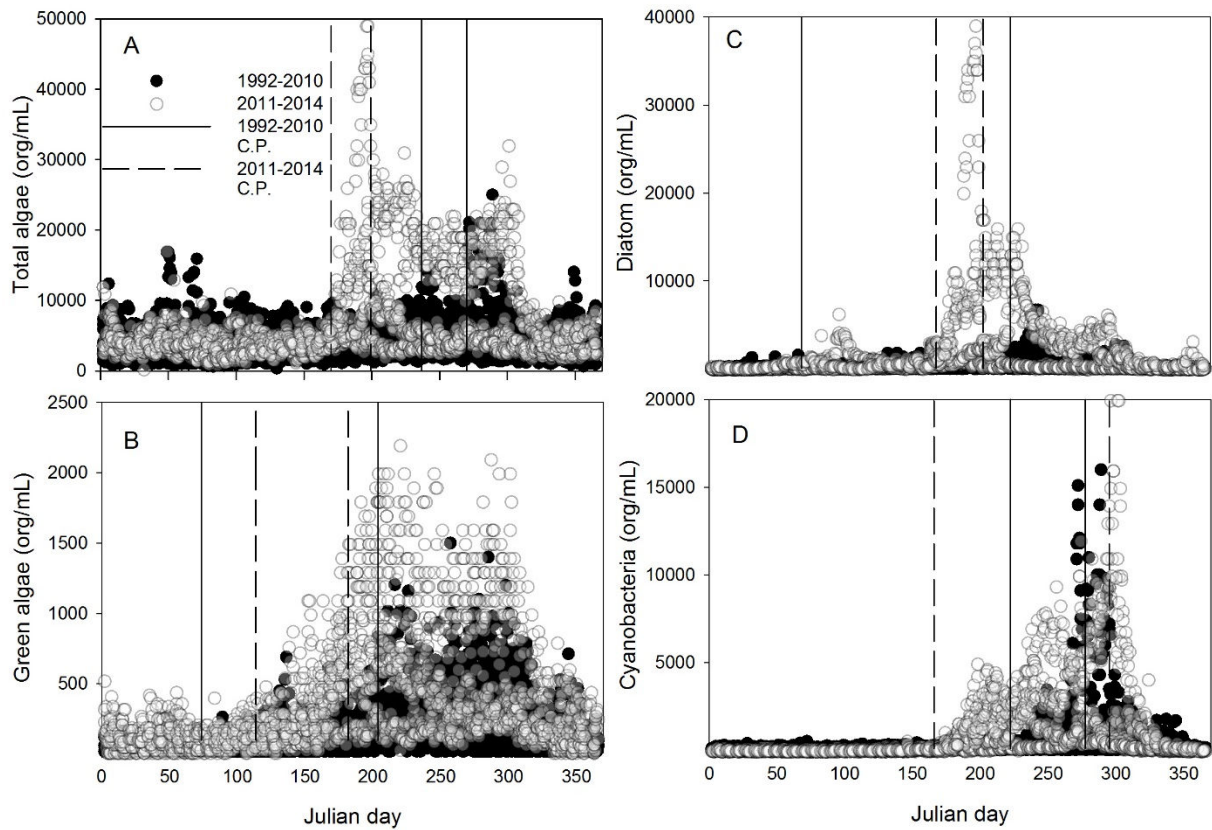
822 2014, closed squares). The periods of 1992–1998 and 1999–2010 represent periods of normal  
823 reservoir management; 2011–2014 was a period of drought-induced water conservation.

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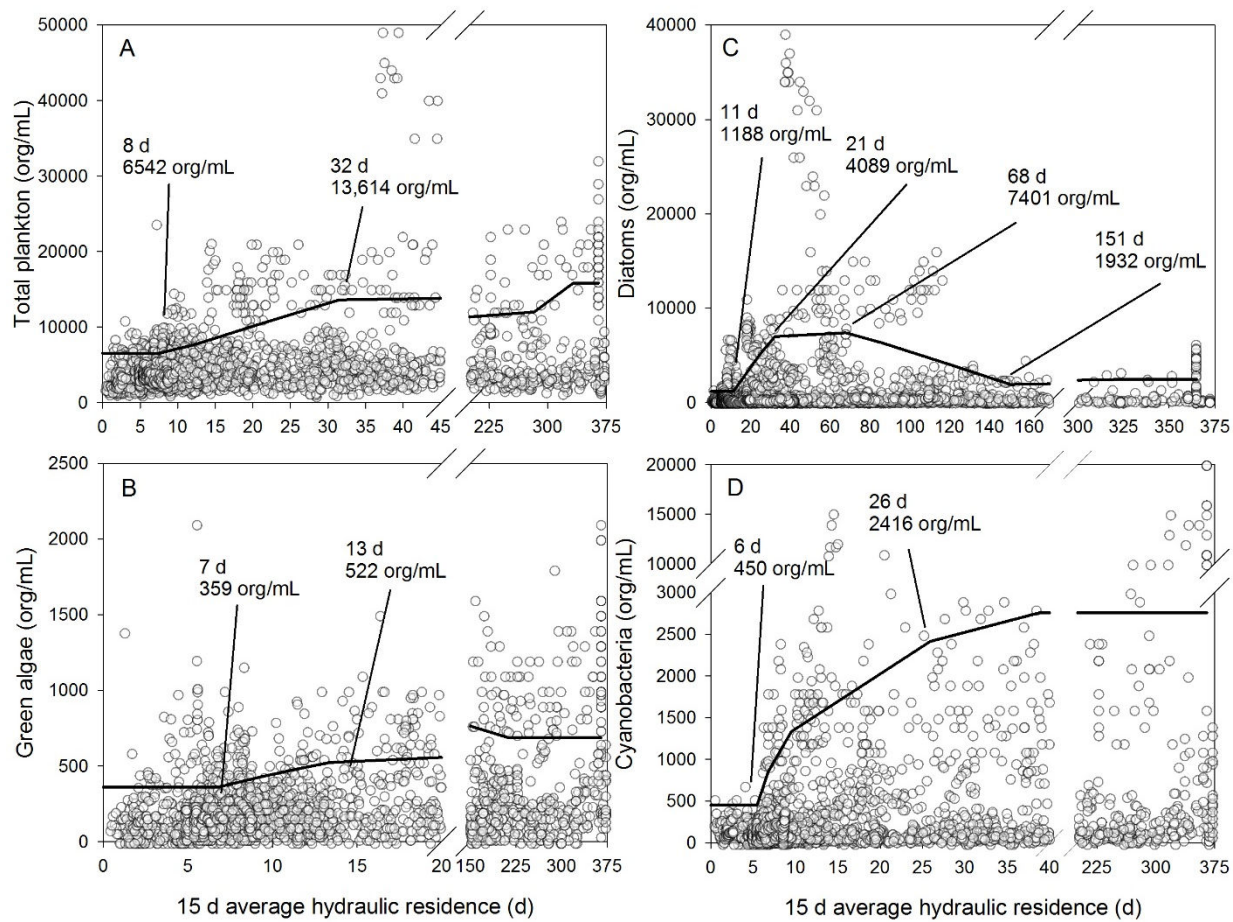
870 Figure 7. Abundances (org/mL) of A) total algae (including flagellates); B) green algae; C)  
871 diatoms; and D) cyanobacteria measured at the drinking water treatment plants near site 12294 in  
872 Lake Austin. Gray areas represent multi-year periods of Lake Travis elevation declines and  
873 lower average daily discharges (from Figure 2A).



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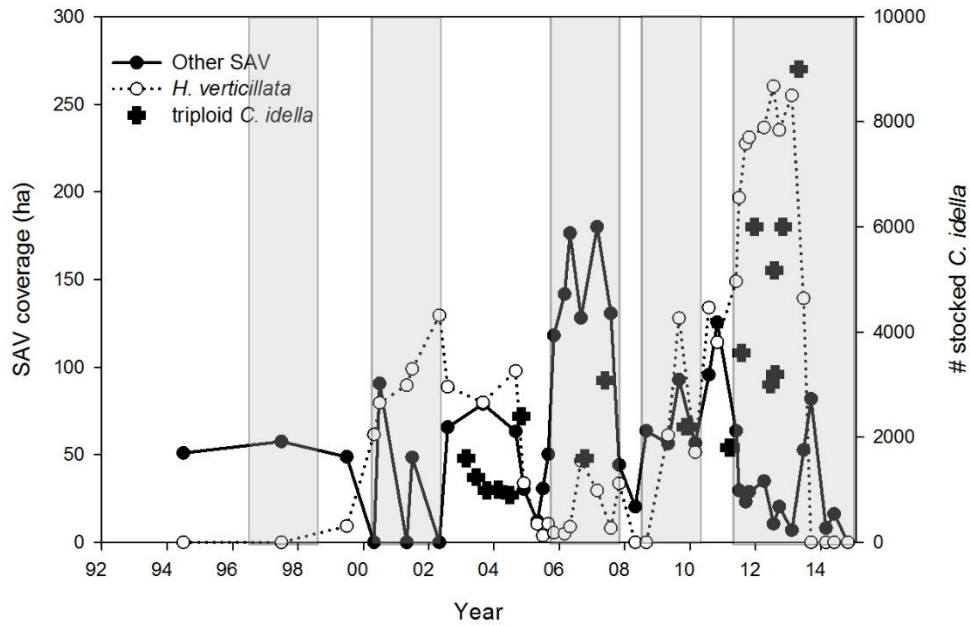
876 Figure 8. Julian day-normalized (x-axis) abundances (org/mL; y-axis) of A) total algae  
877 (including flagellates); B) green algae; C) diatoms; and D) cyanobacteria, for the period 1992–  
878 2010 of normal reservoir management (closed circles) and the 2011–2014 period of drought-  
879 induced water conservation (open circles). Significant change points (C.P.) indicating bloom  
880 initiation and peak for each period are shown with solid lines and dashed lines, respectively.

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Figure 9. Relationship between the 15 d moving average hydraulic residence time of Lake Austin (d) and abundance (org/mL) of A) total algae (including flagellates); B) green algae; C) diatoms; and D) cyanobacteria during the growing season (1 May–30 November) across all years (1992–2014). The solid black line depicts the 95<sup>th</sup> quantile regression of algal abundance with hydraulic residence time. Only select change points are shown in the figure; a full listing of all significant change points can be found in supplemental table 1.

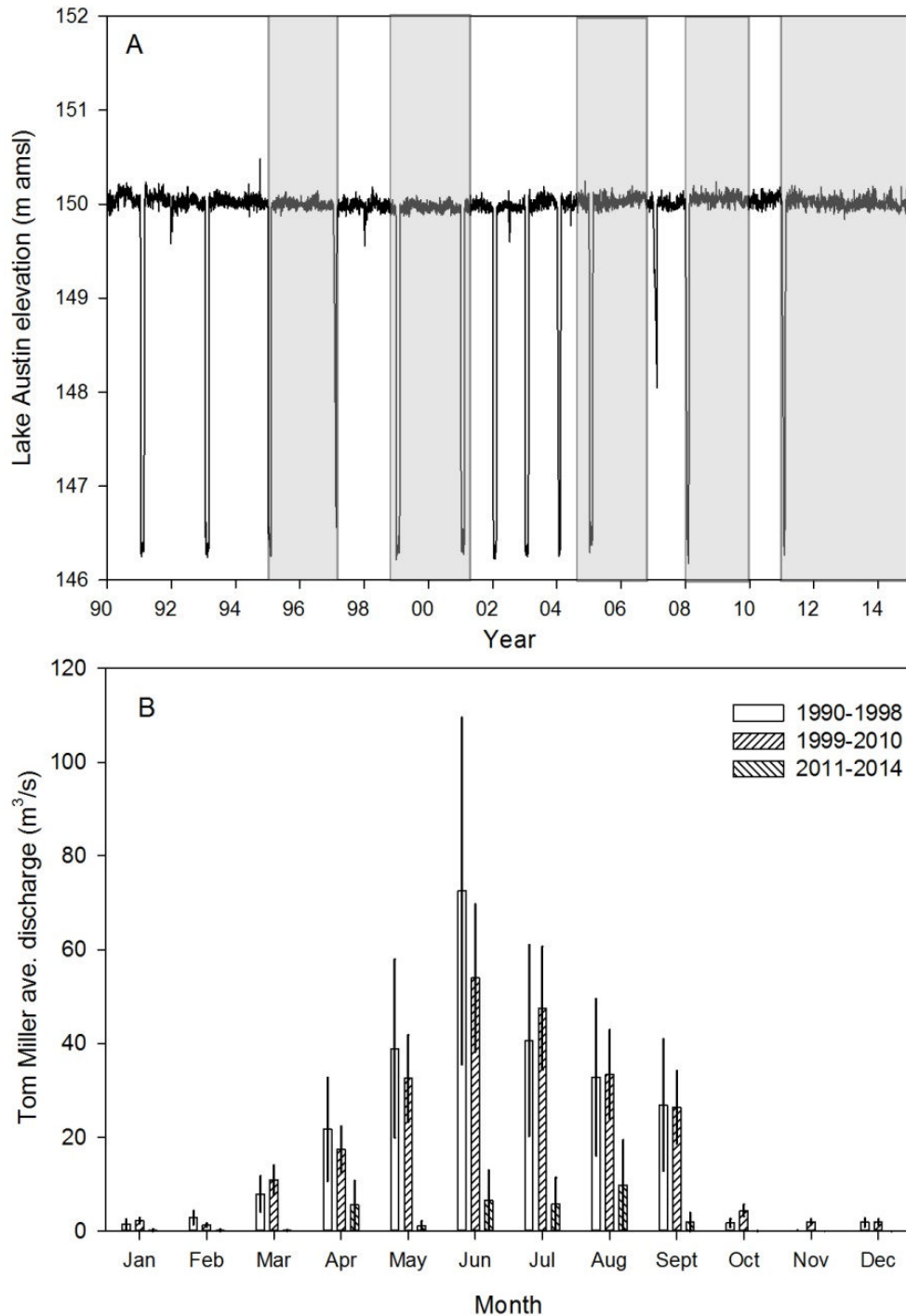


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920 Figure 10. Temporal changes (x-axis) in non-hydrilla aquatic vegetation (closed circles) and  
 921 hydrilla (open circles) coverage (ha; left y-axis) and number of stocked triploid grass carp (#  
 922 fish; crosses, right y-axis). Gray areas represent multi-year periods of Lake Travis elevation  
 923 declines and lower average daily discharges (from Figure 2A).

924 Supplemental table 1. Significant change points identified from the 95<sup>th</sup> quantile regression line fitted to phytoplankton group  
 925 abundances (org/mL) and 15 d moving average hydraulic residence time (see: Fig. 9) or the 15 d moving average Tom Miller Dam  
 926 discharge (m<sup>3</sup>/s; see: Suppl. Fig. 6).  
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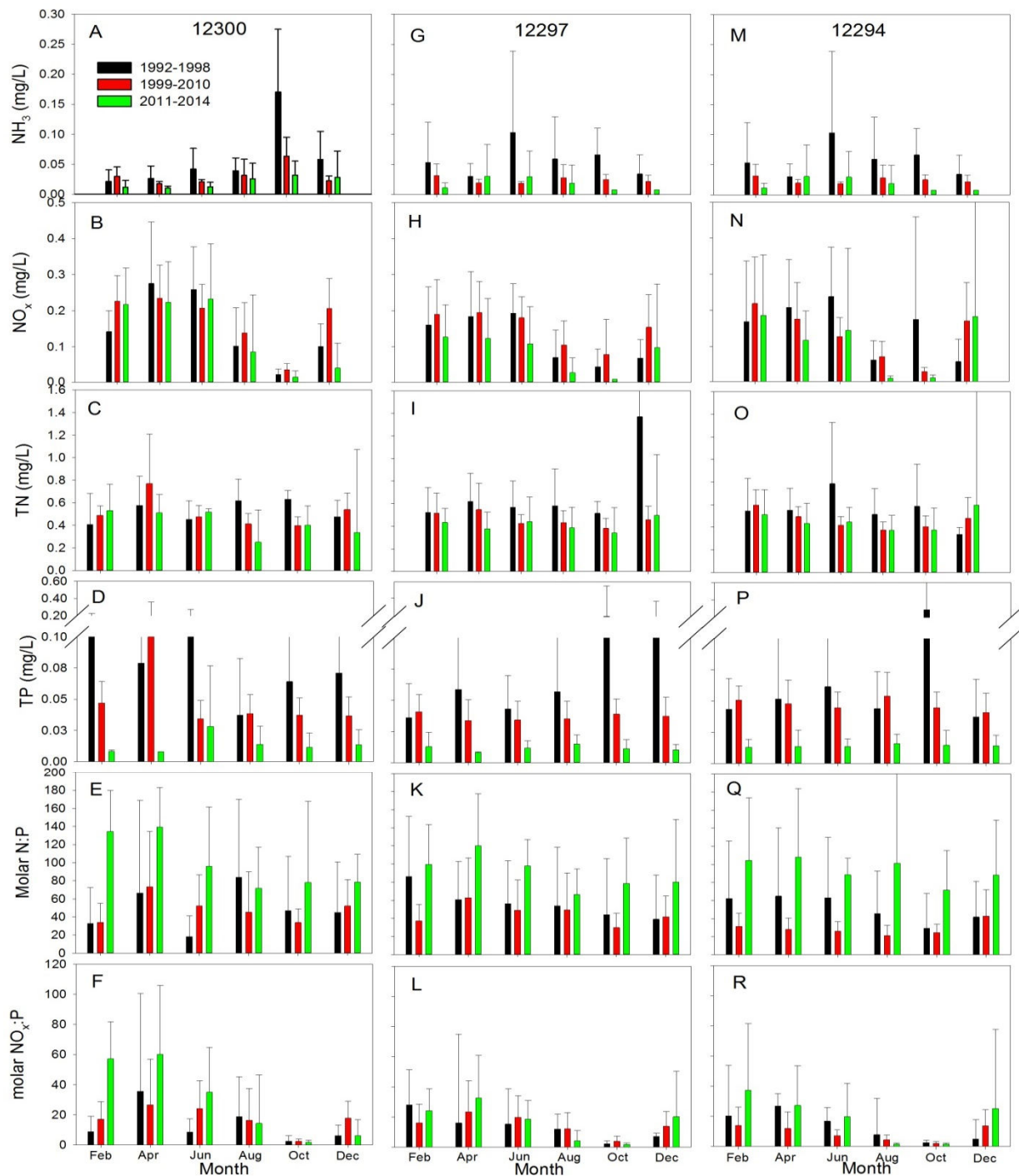
Group	15 d average hydraulic residence (d)	95th quantile biomass (org/mL)	15 d average discharge (m <sup>3</sup> /s)	95th quantile biomass (org/mL)	Group	15 d average hydraulic residence (d)	95th quantile biomass (org/mL)	15 d average discharge (m <sup>3</sup> /s)	95th quantile biomass (org/mL)
Total algae	7.5	6542	5.8	24,675	Diatoms	11.2	1188	2.2	11,360
	11.7	7548	10.7	22,796		16.7	2799	3.7	11,848
	13.1	7927	15.2	20,310		21.0	4089	12.8	11,430
	20.4	10246	18.6	17,301		26.0	5488	16.3	9892
	31.5	13614	23.2	12,430		32.2	7001	18.4	8125
	92.9	14572	26.4	10,079		67.8	7401	20.4	5982
	116.4	12717	37.3	9175		86.0	6339	21.7	4331
	140.9	10876	45.3	10,314		118.8	4128	22.9	2415
	282.7	12018				150.5	1932	23.4	1167
	332.7	15834				335.9	2496		
Green algae	6.8	359	5.7	1624	Cyanobacteria	5.5	450	1.7	9892
	8.6	408	10.5	1498		6.8	856	3.6	8960
	9.5	432	14.9	1332		9.5	1333	5.6	8066
	10.3	452	18.1	1128		25.9	2416	7.7	7176
	11.0	469	22.6	763		38.8	2758	12.9	5619
	12.1	493	48.9	636				24.3	4155
	13.4	522	72.8	844				28.4	2891
	96.2	952	113.1	844				32.2	1469
	142.6	773						34.2	601
	212.8	686							



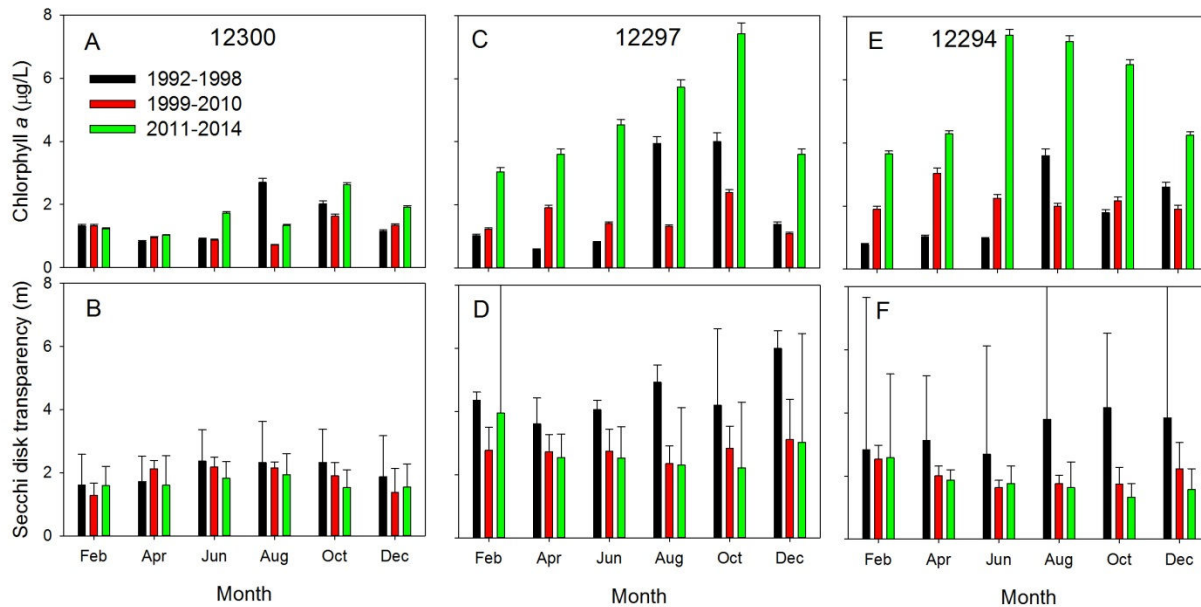
930 Supplemental figure 1. A) Lake Austin daily elevations at Tom Miller Dam (meters above mean  
 931 sea level [m amsl]). Gray areas represent multi-year periods of Lake Travis elevation declines  
 932 and lower average daily discharges. B) Monthly mean  $\pm$  95% confidence intervals of daily  
 933 discharge (m<sup>3</sup>/s) from Tom Miller Dam grouped into periods. The periods of 1990–1998 (open  
 934 bars) and 1999–2010 (bars with forward slash) represent periods of normal reservoir  
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936 management; 2011–2014 (bars with back slash) was a period of drought-induced water  
937 conservation.  
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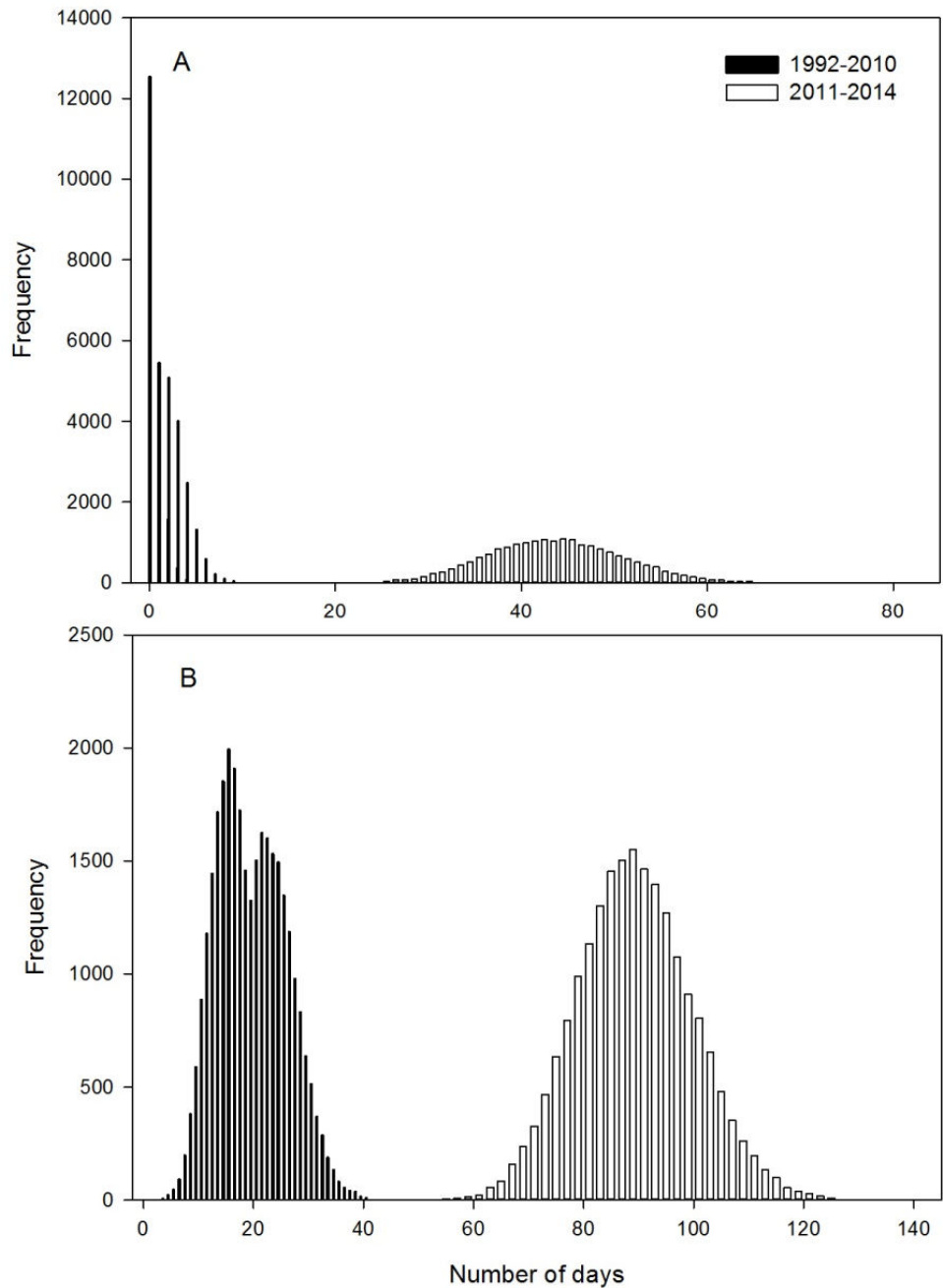


982 Supplemental figure 2. Lake Austin surface water (0.3 m depth) mean  $\pm$  95% confidence  
 983 intervals for nutrient chemistry parameters at Lake Austin sites 12300 (left column), 12297  
 984 (center column), and 12294 (right column) summarized for each period (black fill: 1992–1998;  
 985 red fill: 1999–2010; green fill: 2011–2014) for each bi-monthly sampling. The periods of 1992–  
 986 1998 and 1999–2010 represent periods of normal reservoir management; 2011–2014 was a  
 987 period of drought-induced water conservation.  
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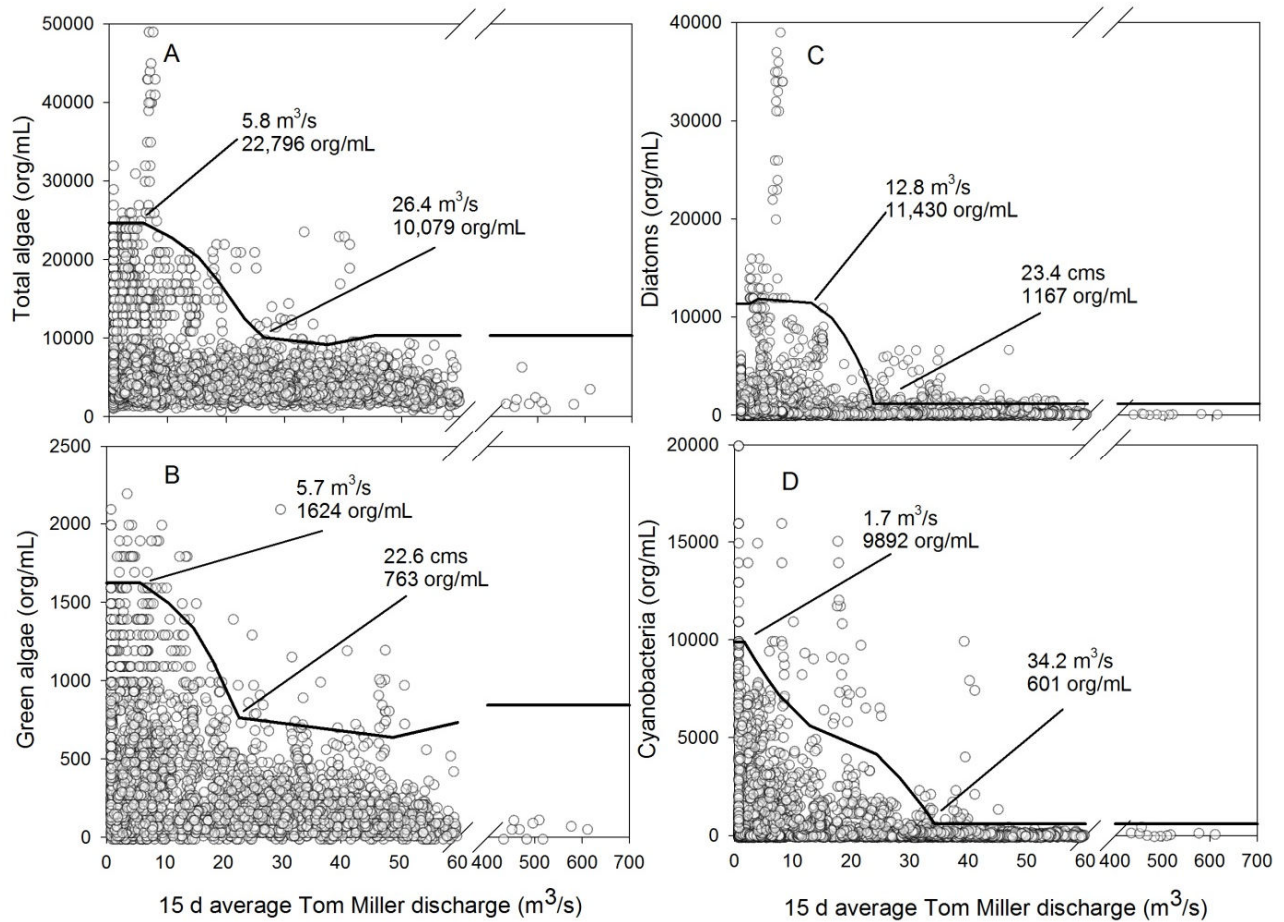


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 991 Supplemental figure 3. Lake Austin surface water (0.3 m depth) mean ± 95% confidence  
 992 intervals for chlorophyll *a* (µg/L; top row) and Secchi disk transparency (m; bottom row) at sites  
 993 12300 (left column), 12297 (center column), and 12294 (right column) summarized for each  
 994 period (black fill: 1992–1998; red fill: 1999–2010; green fill: 2011–2014) for each bi-monthly  
 995 sampling. The periods of 1992–1998 and 1999–2010 represent periods of normal reservoir  
 996 management; 2011–2014 was a period of drought-induced water conservation.  
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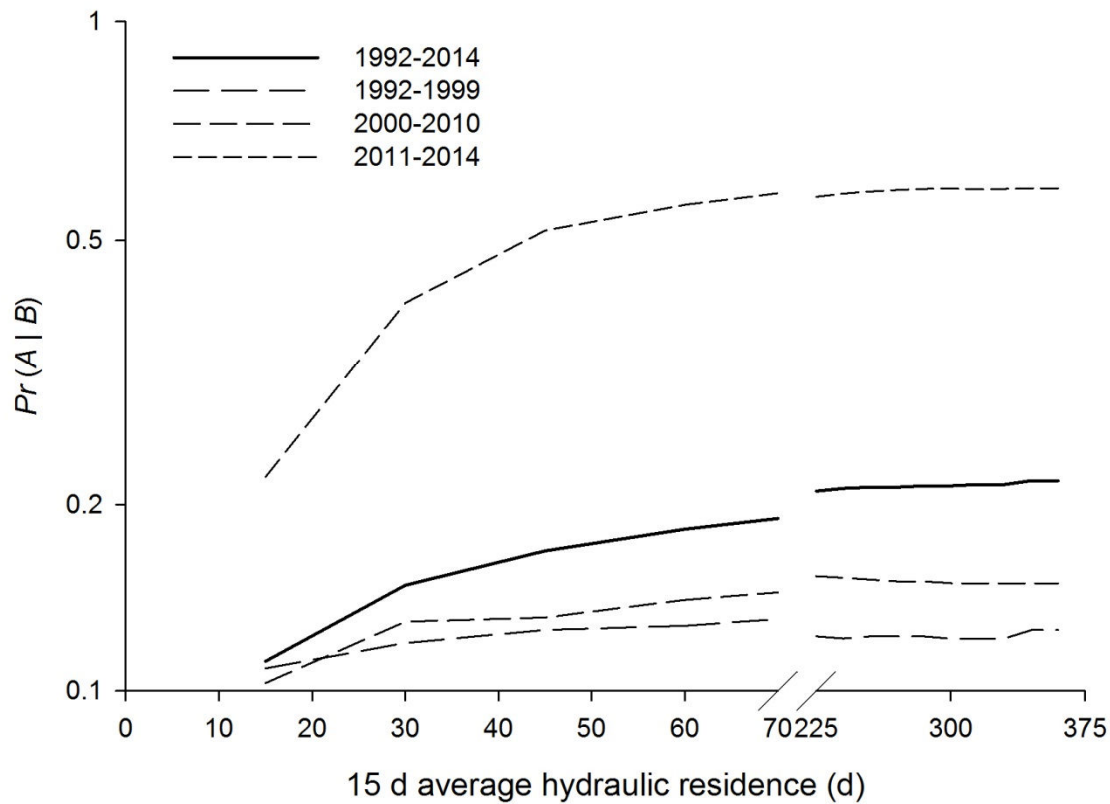


1020 Supplemental figure 4. Bayesian posterior frequency distribution (unitless) of the mean number  
 1021 of growing season days (1 May–30 November) that Lake Austin could be in exceedance of  
 1022 Austin Water bloom cell abundance thresholds for A) total algae (i.e., >10,000 org/mL) and, B)  
 1023 cyanobacteria (i.e., >300 org/mL) during normal reservoir management (solid bars) and drought-  
 1024 induced water conservation (open bars) years.  
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Supplemental figure 5. Relationship between the 15 d moving average daily discharge (m<sup>3</sup>/s) from the Tom Miller Dam and abundance (org/mL) of A) total algae; B) green algae; C) diatoms; and D) cyanobacteria during the growing season (1 May–30 November) across all years (1992–2014). The solid black line depicts the 95<sup>th</sup> quantile regression of algal abundance measurements with discharge. Only two significant change points are shown on the figure; all of the change points identified are given in supplemental table 1.



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 1050 Supplemental figure 6. Probability predictions,  $Pr(A|B)$ , of cyanobacteria abundance exceeding  
 1051 300 org/mL ( $A$ ) at different 15 d average hydraulic residence times (d) in Lake Austin ( $B$ )  
 1052 derived from Bayes' theorem (see methods for full equation). Probabilities determined for the  
 1053 entire period of record (1992–2014; solid line), and the periods of 1992–1998 (long dash line)  
 1054 and 1999–2010 (medium dash line) which represent periods of normal reservoir management;  
 1055 2011–2014 (short dash line) was a period of drought-induced water conservation.