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EXTERNAL NUTRIENT LOADING AND WATER QUALITY TRENDS IN THE UPPER OCKLAWAHA BASIN LAKES, 1980S THROUGH 2019-2020

by

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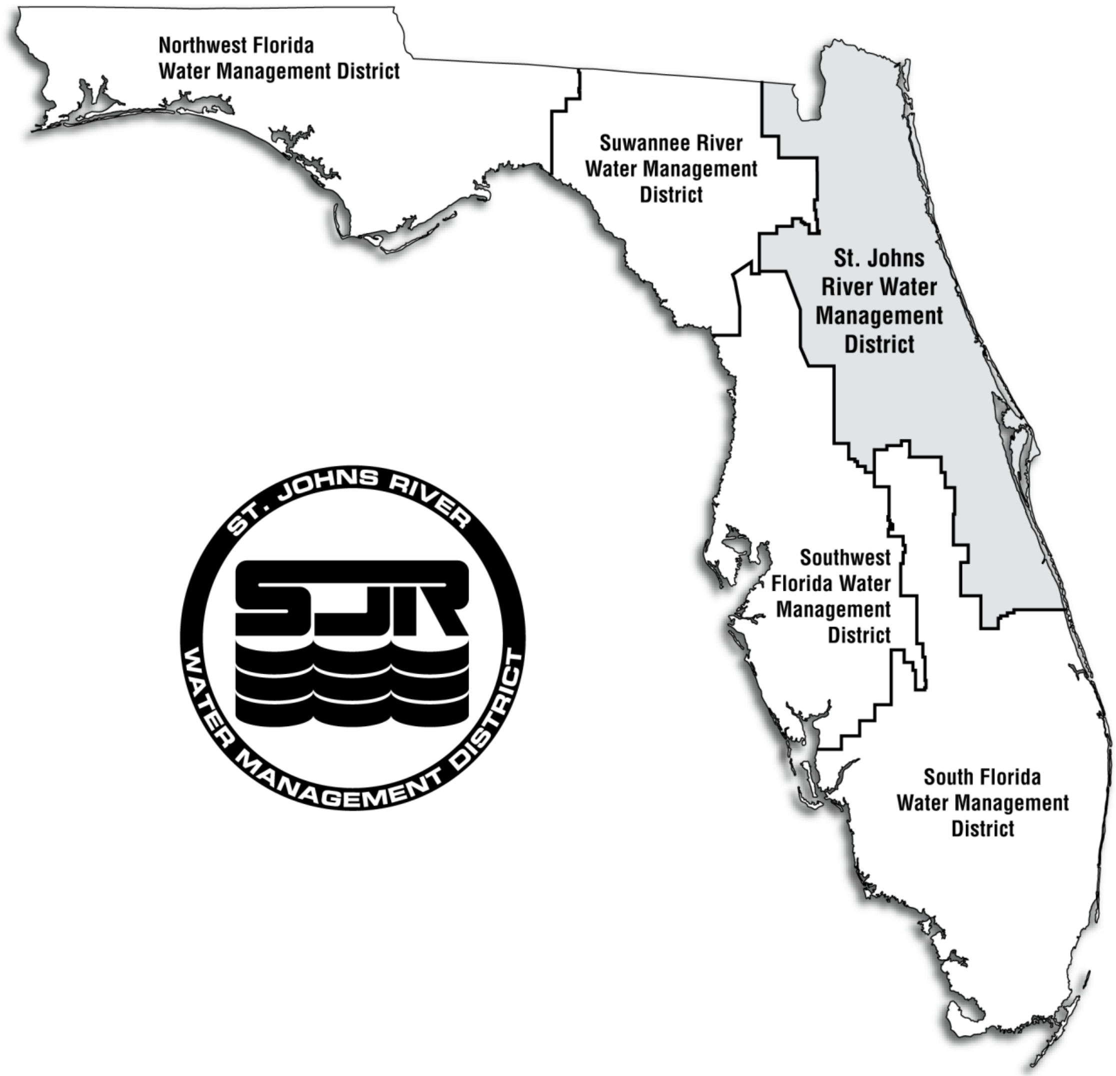

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EXECUTIVE SUMMARY

This report summarizes trends in external total phosphorus and total nitrogen loading and water quality through 2019–20 for eight major and three minor lakes in the Upper Ocklawaha River basin. The beginning of water quality data sets varies among the lakes; as early as 1983, in the late-1980s for most of the lakes, and after 2000 for two lakes. Major restoration projects in the Lake Apopka basin, upstream of Lake Beauclair, and in the Lake Griffin basin have led to decreases in total phosphorus loading and water quality improvements for six basin lakes (Apopka, Beauclair, Dora, Harris, Eustis, and Griffin), with the largest improvements generally in lakes Griffin, Beauclair, and Dora. There have also been water quality improvements in Lake Carlton, which benefits from the restoration projects upstream of Lake Beauclair. Many trophic state parameters have improved in the seven lakes, including decreases in concentrations of total phosphorus, total nitrogen, chlorophyll-*a*, and total suspended solids, and increases in Secchi depth transparency. On the other hand, water quality has deteriorated in lakes Yale and Weir, which are unaffected by basin restoration projects, and in Trout Lake, which has been influenced by the Pine Meadows wetland restoration. Most of the lakes had infrequent exceedances of the state water quality standard for ammonia. Lake Griffin often exhibited large winter increases in ammonia-nitrogen concentrations.

Water elevation changes appear to have had the most significant effects on water quality in Lake Apopka, where repeated multi-year drought periods since around 2000 have led to cyclic changes in water quality, and in lakes Weir and Yale, where part of the deterioration in water quality appears to be due to a substantial decrease in lake water elevations. In Lake Apopka, water quality deteriorated during the severe droughts, but substantially improved during periods of more normal water levels. The shallow depth, high dynamic ratio, and large changes in lake volumes of Lake Apopka make it susceptible to strong effects of water level fluctuations on water quality.

Estimates of potential nutrient loading to Lake Weir from gull populations indicate waterfowl excretion could be a significant source of phosphorus to that lake. Year-round surveys of the bird populations and their feeding and roosting activities are necessary to verify that conclusion.

Trends in phytoplankton biovolumes are not as apparent as seen for chlorophyll-*a*. There have been apparent decreases in peak phytoplankton biovolumes in Lake Griffin, although trends are not as clear in the other lakes. Chlorophyll is probably a better indicator of phytoplankton biomass than biovolumes. Reasons for this include: chlorophyll measurements are based on a much larger sample volume; biovolume estimates are approximations based on similarity of the cells to standard geometrical shapes; biovolume estimates may be particularly difficult for colonial or filamentous species; and there appear to be substantial differences in biovolume measurements by the different phytoplankton analysts used in this study (see General Discussion section). The phytoplankton in all the lakes have been dominated by cyanobacteria during the warm season. *Cylindrospermopsis*

usually has been the dominant or co-dominant cyanobacterial genus in all the lakes, except Apopka, where *Planktolyngbya* tends to be dominant, and in Trout Lake, where *Microcystis* is dominant.

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INTRODUCTION

This report summarizes trends in external total phosphorus (TP) and total nitrogen (TN) loading and in-lake water quality through 2019–20 for the major lakes in the Upper Ocklawaha River basin (UORB; Figure 1), including:

- Apopka
- Beauclair
- Dora
- Harris-Little Harris
- Eustis
- Griffin
- Yale
- Weir

Also, water quality trends are summarized for three minor lakes in the UORB, but complete nutrient loading estimates are not available for these lakes:

- Denham
- Carlton
- Trout

Lake Apopka is one of the headwaters of the chain of lakes. From there, water flows through lakes Beauclair, Dora, Eustis, and Griffin, which discharges to the Ocklawaha River. Water flows are controlled by dams between lakes Apopka and Beauclair, between lakes Eustis and Griffin, and downstream of Lake Griffin (Figure 1). Lake Harris receives flow from the Palatlakaha River, and discharges to Lake Eustis. In 2008, a flood-control connection was established through the Lake Harris Conservation Area (also called Harris Bayou) to allow discharges from Lake Harris to Lake Griffin, although this has been used only a few times during high-flow periods. Lake Yale discharges to Lake Griffin. Lake Denham is connected to Lake Harris, Lake Carlton is connected to Lake Beauclair, and Trout Lake is connected to Lake Eustis. During dry periods there can be some flow reversals among lakes Eustis, Harris, Dora and Beauclair; the magnitude of these flows is uncertain (net flows among these lakes over month-long periods are estimated for the nutrient loading calculations) but are likely small compared to the main flow direction. There may also be flow reversals between lakes Carlton, Denham, and Trout and their connected lakes. There are also rare occurrences of flow reversals between lakes Griffin and Yale. Lake Weir can discharge to the Ocklawaha River if its water level gets sufficiently high, but that has not happened during the period covered by this report. Numerous minor streams also enter the lakes from the surrounding watershed (Fulton et al. 1995; Hoge et al. 2003).

Morphometric and hydrologic data for the lakes are included in Table 1. Dynamic ratio is calculated as the square root of the lake surface area in km^2 divided by the mean depth in meters and has been used as an index of potential for wind-driven sediment resuspension

(Håkanson, 1982). Bachmann et al. (2000) concluded that the entire lakebed is subject to sediment resuspension if the dynamic ratio was >0.8. That would suggest most of the UORB lakes are subject to considerable sediment resuspension, with Lake Apopka most susceptible. Due to generally dry conditions in recent years, lake water elevations have usually been below the reference (typical) elevations, particularly for lakes Apopka, Yale, and Weir.

Most of the basin lakes receive mineralized groundwater, as well as surface inflows through nutrient-rich soils, and are considered naturally productive hard water lakes. However, Lake Weir has been included in an ecoregion of clear, low-nutrient lakes (Canfield 1981, Griffith et al. 1997). For decades, most of the major lakes in the basin have been characterized as eutrophic to hypereutrophic, while only lakes Weir and Yale have been classified as mesotrophic (Shannon and Brezonik 1972, Canfield 1981).

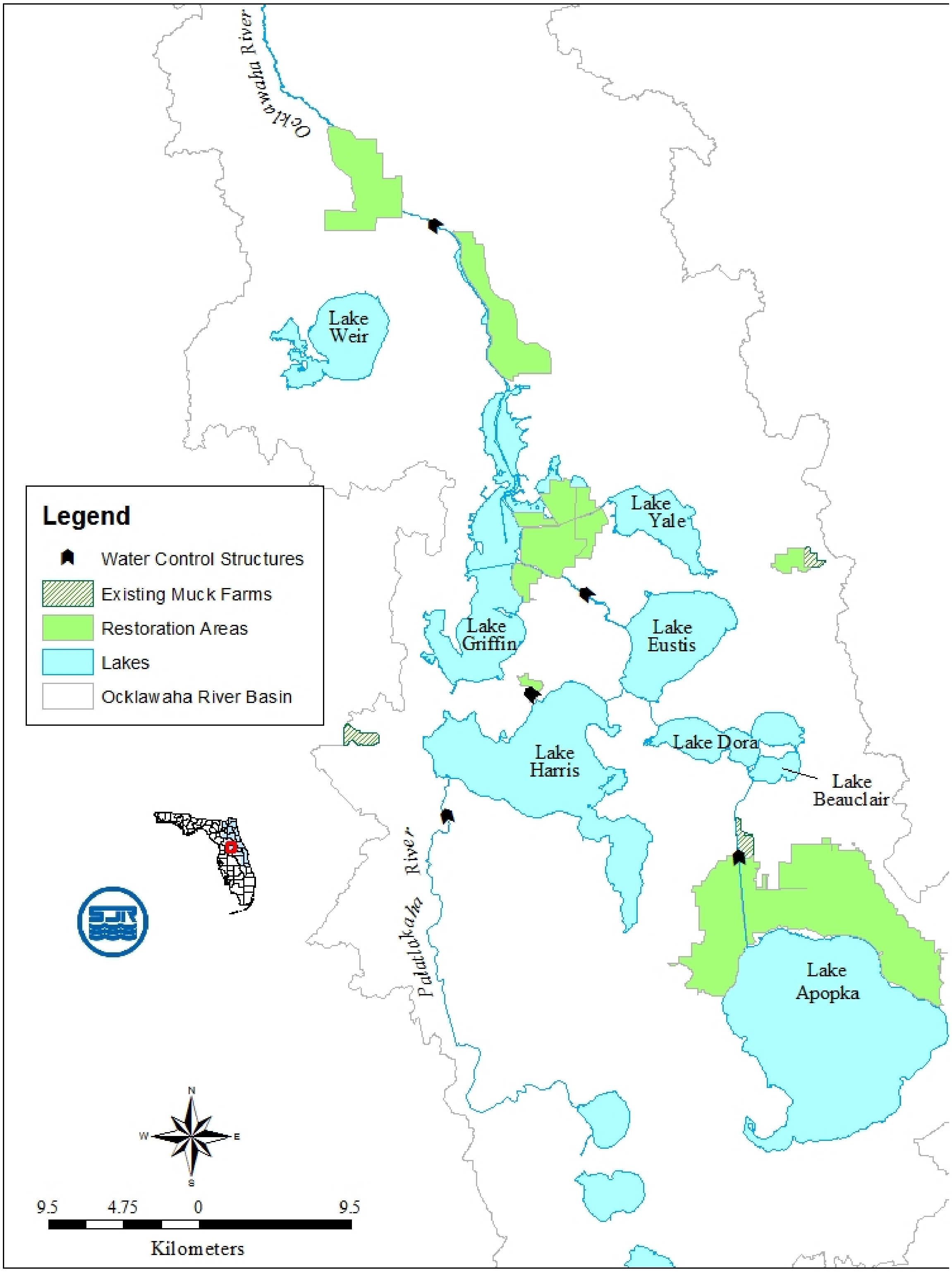


Figure 1. Upper Ocklawaha River basin.

Table 1. Morphometric and hydrologic data for the UORB lakes.

Morphometric data are for the reference elevations (typical water elevations for the lakes, within the regulation schedule for the regulated lakes).

Lake	Reference elevation (feet NAVD 1988)	Surface area (acres)	Mean depth (feet)	Maximum depth (feet)	Mean Dynamic ratio 2001-2019	Mean elevation 2001-2019 (feet NAVD 1988)[1]	Mean water residence time 2001-2019 (years)[1]
Apopka[1]	66	32,244	5.8	17.0	8.31	64.77	3.9
Beauclair[1,2]	62	1,218	6.2	14.4	1.14	61.25	0.20
Dora[1,2]	62	4,462	9.6	17.2	1.49	61.25	1.0
Harris-Little Harris[1,2]	62	19,079	11.7	31.9	2.49	61.19	3.2
Eustis[1,2]	62	8,315	10.5	21.7	1.78	61.17	0.82
Griffin[1,3]	58	13,837	5.8	20.1	2.82	57.21	0.64
Yale[1,2]	58	3,945	12.5	26.0	1.25	56.56	10.0
Weir[1,2]	56	6,028	18.0	31.8	0.99	51.70	22.3
Denham[1]	62	245	4.4	ND[4]	0.92	61.19	ND[4]
Carlton[1]	62	387	12.4	ND[4]	0.35	61.25	ND[4]
Trout[1]	61.7	103	7.7	ND[4]	0.27	61.17	ND[4]

Data Sources:

[1]Unpublished data

[2]Pachhai et al. 2013

[3]VanSickle 2013

[4]ND – No data

Beginning in the mid-1900s, large wetland areas in the basin were drained for agriculture, referred to as muck farms, particularly along the shores of lakes Apopka and Griffin. Discharges from the Apopka basin muck farms were major nutrient sources for that lake (Coveney et al. 2005). Recent decades have seen increasing urban and residential development in the drainage basin. For example, the populations of Lake and Marion Counties, which include most of the drainage basins for the lakes, more than sextupled from 1950 to 2000 (Florida Department of Environmental Protection [FDEP] 2001). Because of poor water quality and habitat loss, the UORB lakes were prioritized for development of Total Maximum Daily Loads (TMDLs). Phosphorus was identified as the key nutrient for development of TMDLs for the major UORB lakes and is a primary focus of this assessment. TMDLs for TP were adopted for all the major basin lakes, except for Lake Weir, by FDEP in 2003. The adopted TMDLs were based on Pollutant Load Reduction Goals (PLRGs) developed by St. Johns River Water Management District (SJRWMD) (Coveney 2000;

Fulton et al. 2004; Fulton & Smith 2008). FDEP also adopted a TMDL for TP for Lake Carlton in 2003 and adopted TMDLs for both TP and TN for Trout Lake in 2006 and for Lake Weir and Lake Denham in 2017.

Several significant restoration projects have been conducted in the basin that have influenced water quality. Beginning in the late 1980s, most of the basin's muck farms were purchased by SJRWMD, and these areas are being managed to reduce nutrient discharges and restore wetland and other aquatic habitat (Figure 1). These wetland restoration areas include:

- Lake Apopka North Shore (LANS) – a nearly 20,000-acre area on the northern shore of Lake Apopka acquired between 1988 and 1999
- Lake Harris Conservation Area (LHCA, also referred to as Harris Bayou) – a 500-acre area between lakes Harris and Griffin acquired between 1990 and 1992
- Pine Meadows Restoration Area (PMRA) – an 800-acre area in the watershed for Lake Eustis and Trout Lake acquired in 1992, and donated to Lake County in 2013
- Emeralda Marsh Conservation Area (EMCA) – a 7,000-acre area on the eastern shore of Lake Griffin acquired between 1991 and 1993

Included within the LANS is the Lake Apopka marsh flow-way project, which has operated since late 2003, other than some periods it was shut down for maintenance. This project circulates lake water through a wetland removing nutrients and suspended sediments (Coveney et al. 2002; Dunne et al. 2012). A pilot demonstration marsh flow-way project was operated for portions of 1990-1994 (Coveney et al. 2002). Water is discharged from the marsh flow-way into the Apopka-Beauclair Canal connecting lakes Apopka and Beauclair. Some of the discharge flows back into Lake Apopka and some flows downstream to Lake Beauclair. Another significant restoration project is the Nutrient Reduction Facility (NuRF), developed by Lake County Water Authority, and located near the dam on the Apopka-Beauclair Canal (Figure 1). This project has treated discharges from Lake Apopka with alum to reduce TP loading to the downstream lakes since early 2009. Another significant restoration activity is harvesting of rough fish (largely gizzard shad, *Dorosoma cepedianum* Le Sueur), primarily in lakes Denham, Apopka, and Griffin, to remove phosphorus and reduce recycling of nutrients from the bottom sediments (Godwin et al. 2011; Schaus et al. 2010; 2013; Fulton et al. 2015). The Basin Management Action Plan (BMAP) developed to implement the TMDLs has also included numerous stormwater projects to reduce TP loads to basin lakes (Florida Department of Environmental Protection 2014).

In this report, I will present annual estimates of external TP and TN loading, impacts of basin restoration projects on TP loading, analyses of trends in water quality, and summaries of phytoplankton biovolume and composition for eight major lakes in the UORB, through 2019-20. Water quality trends, but not nutrient loading estimates, are also summarized for three minor lakes in the basin. The trend analyses included both raw trends: "what the waterbody stakeholder sees", and influences of seasonality and changes in lake water elevations on those trends. This report is the fourth of a series of reports on nutrient loading and water quality trends in the UORB lakes (Fulton 2015, 2016, 2018). This report is the first of this series to include the minor lakes Denham, Carlton, and Trout.

METHODS

NUTRIENT LOADING

External nutrient loading to the major lakes is estimated through 2019 in this report. Methods used in estimating TP and TN loading are only briefly described here; more complete descriptions of methodology can be found in Fulton (1995) and Fulton et al. (2004). The methods described in those documents refer specifically to those for lakes other than Lake Apopka. For years prior to 2011, TP loads for Lake Apopka were developed by other staff using similar, but not identical methods.

In early years, rainfall estimates were developed from a network of SJRWMD and National Oceanic and Atmospheric Administration rainfall stations in the basin. A Thiessen polygon data layer (Chow et al. 1988) was developed from the rain station locations to define the areal extent of each station's rainfall. Beginning in 2002, I used Nexrad Doppler radar data (Fulton et al. 1998), which provided rainfall estimates for a 4 km^2 grid. Data on nutrient concentrations in rainfall and dry deposition were taken from wet/dry deposition collectors operated by SJRWMD near Lake Apopka. To reduce the influence of occasional outliers, I used annual median TP and TN concentrations in rainfall and dry deposition for calculations of direct atmospheric deposition to the lake surface.

Daily discharge volumes were obtained from USGS stations located at dams on major tributaries (Figure 1). Other major tributary discharges were estimated from a water budget for the upstream lake (discharges from lakes Harris, Beauclair, and Dora) or from rating curves developed using the U.S. Army Corps of Engineer's HEC-RAS model (discharges from Lake Yale). Tributary nutrient discharges were determined using annual flow-weighted mean TP or TN concentrations multiplied by annual discharge volumes (Galat 1990). In recent years, a different method was used to estimate nutrient discharges to Lake Beauclair through the Apopka-Beauclair Dam or the NuRF; daily discharge volumes were multiplied by TP or TN concentrations interpolated between measured values.

Stormwater runoff from surrounding watersheds was estimated using land use and soil maps and estimated basin rainfall. The land use data used for stormwater runoff estimates were derived from aerial photography taken in 1987, 1994–1995, 2000, 2004, 2009, and 2014. Runoff volumes were estimated using U.S. Soil Conservation Service (SCS) runoff methods (Suphunvorranop 1985, Soil Conservation Service 1986). Pandit and Gopalakrishnan (1996) found close correspondence between average watershed annual storm runoff coefficients estimated by the SCS methodology and measured values for another central Florida watershed. The overall runoff volume was then partitioned into land use and soil-specific runoff coefficients based on estimated pervious and impervious areas of different land uses and relative soil storage of different soil types. The resulting runoff coefficients were similar to coefficients developed for the central Florida area by Pandit and Gopalakrishnan (1996). Runoff TP and TN concentrations and sediment-associated fractions of these nutrients from different land use types were estimated from literature sources, primarily from a compilation of Florida studies by Harper (1994). Differences between the 1987 and later land use maps were used to determine development that occurred in the watershed after 1987. I assumed that there was stormwater treatment only for lands developed after the 1987 land use maps, because Florida did not require stormwater treatment for new development until after 1984. Based on the average treatment performance from 13 studies of Florida stormwater systems (Fulton et al. 2004), I assumed that 63% of the TP load and 42% of the TN load was removed by stormwater treatment. Losses of phosphorus and nitrogen in transport between the runoff source and the receiving water bodies were estimated using a relationship developed by Reckhow et al. (1989), which estimates losses of sediment-associated nutrients as a function of transport distance. No transport losses were assumed for the dissolved fraction of phosphorus or nitrogen in stormwater runoff. Appendix E provides a comparison of modeled estimates of stormwater runoff used in this study with runoff measurements or estimates from three recent runoff studies conducted in the basin.

Discharge volumes from operating muck farms were estimated by using a multiple regression equation developed in Fulton (1995), which related discharge volumes to area in production, rainfall, and evaporation. Nutrient concentrations in farm discharges were taken from monitoring data included in permit records or from data collected by SJRWMD after purchase of the properties. Discharge volumes from SJRWMD restoration areas were estimated from pump records or using the stormwater runoff methodology described above. Concentrations of TP and TN in restoration area discharges were taken from SJRWMD monitoring. I also estimated TP and TN loading for several point sources from data reported in FDEP permit files, including surface (weak waste) discharges from citrus processing plants, spills from citrus processing plants, runoff from waste disposal areas for citrus processing plants, runoff from waste disposal areas for municipal waste treatment plants and municipal waste spills. The same procedures used to estimate nutrient losses in transport of stormwater runoff were applied to estimate losses of nutrients in transport from muck farms, restoration areas, and point sources.

Total phosphorus and total nitrogen loading from septic tank effluents was estimated using methods described by Reckhow et al. (1980) and incorporated into the Eutromod watershed and lake modeling program (Henning and Reckhow 1990). I assumed only septic systems located within 200 m of the lakes, lakeshore wetlands, or canals connecting to the lakes contributed nutrients to the lakes, per capita releases of 1.48 kg TP/year and 4.75 kg TN/year, and soil retention of 90% for phosphorus and 45% for nitrogen. Septic tank nutrient loading estimates were not made for Lake Apopka by previous staff. I applied estimates based on a septic tank coverage for the Lake Apopka watershed in 2017 back through the year 2013.

A basin overview of nutrient loading was developed comparing averages for the baseline period used for TMDL development (1989-1994 for Lake Apopka, 1991-2000 for the other lakes), the most recent 5-year period 2015-2019, and loading targets developed for the TMDLs (or the PLRG for Lake Weir). For Lake Apopka, TN loading estimates have been developed only for the years 2012-2019. External loading was expressed both as average metric tons per year and adjusted for lake surface area (g/m^2/year).

For Lake Apopka, I estimated the reductions in TP due to the LANS restoration, the marsh flow-way, and the gizzard shad harvest. Load reductions due to the LANS restoration were estimated as the change in loads from the average during the baseline period for which the Lake Apopka TMDLs were developed (1989-1994, during which most of the farms were still in operation, although best management practices [BMPs] were beginning to developed to reduce farm discharges). TP removals from the lake were estimated from the beginning of shad harvesting in 1993 and of operations of the marsh flow-way in 2003. The small TP removals from the pilot demonstration marsh flow-way project were not included. The effect of shad harvesting only considered TP removal in the bodies of the fish, not changes in recycling of TP from the bottom sediments due to shad feeding activities.

For the downstream lakes Beauclair, Dora, Harris, Eustis, and Griffin, I estimated reductions in TP starting with the year 2000 due to:

1. Changes in Lake Apopka discharge TP concentrations from the average concentration during 1989-1999 (Figure 2). Discharge concentrations during this period were higher than lake concentrations due to direct discharges into the Apopka-Beauclair Canal from the operating farms and from the construction phase of the marsh flow-way.
2. Marsh flow-way TP removals from water discharged from Lake Apopka. The marsh flow-way discharges were partitioned between those that flow back into Lake Apopka vs. those that flow downstream to Lake Beauclair, based on daily discharge volumes from the flow-way and downstream discharge volumes. During the operational period from 2003-2019, only about 13% of the total flow-way removals flowed downstream to Lake Beauclair.
3. TP removals by the Lake County Water Authority's NuRF project, calculated from flow rates and inflow and outflow concentrations.
4. TP load reductions from Lake Harris Conservation Area, Pine Meadows Restoration Area, and Emeralda Marsh Conservation Area, based on estimates of TP discharges from the operating farms vs. those from these areas under restoration – these are the most uncertain load reduction estimates because we have limited information on actual discharges from the operating farms.
5. TP removals from Lake Griffin by the shad harvest in that lake, which occurred from 2002-2008. The shad harvest effects only considered TP removal in the bodies of the fish, not changes in recycling of TP from the bottom sediments due to shad feeding activities.

To propagate changes in estimated TP loading received by an upstream lake to the next lake downstream, I assumed that the TP discharged downstream from each lake would be proportional to the changes in TP loads to that lake. For example, if there had been no changes in Lake Apopka discharge TP concentrations, the TP discharge from Lake Beauclair (B) to Lake Dora without Lake Apopka discharge concentration changes was estimated as:

$$= \text{Existing B TP discharge} \times \frac{\text{Estimated B TP load without Apopka discharge conc changes}}{\text{Existing B TP load}}$$

This assumption may not be accurate for an individual year; for example, in some years the dams prevented downstream discharges from some of the lakes, so changes in nutrient loads to those lakes would not be propagated downstream in those years. However, over an extended time period it is reasonable to assume that downstream TP discharges from a lake will be proportional to the TP loads received by that lake. The changes in estimated TP loading will be presented as the cumulative reductions over the 20-yr period 2000-2019.

SJRWMD was approached by residents from the Lake Weir area who believed that flocks of seagulls that roost on the lake are feeding at Marion County's Baseline Landfill and may be potentially introducing phosphorus and nitrogen to the lake through their wastes. The landfill is about 14 km from Lake Weir, which is well within the flying range of foraging gulls (Patenaude-Monette et al. 2014). The only quantitative data available on bird populations on Lake Weir are the Christmas Bird Counts (CBC). CBC data from 2001-2012 were provided by John Stenberg. I estimated potential effects of gull excretion on Lake Weir nutrient concentrations, by using literature excretion rates to estimate nutrient excretion by a range of gull population sizes and comparing with the estimated external nutrient loading rates for the lake.

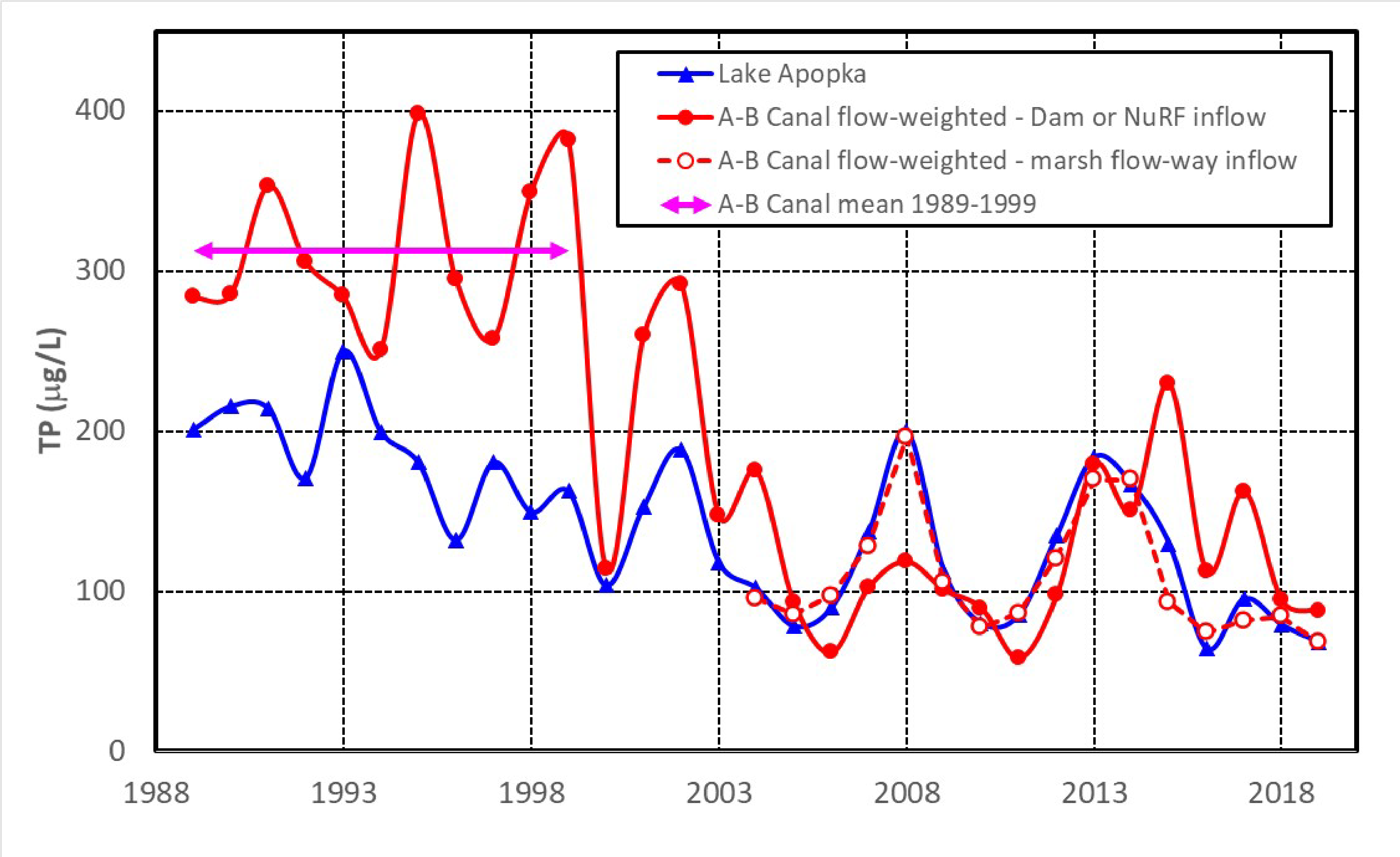


Figure 2. Average annual TP concentrations in Lake Apopka and in the Apopka-Beauclair Canal.

WATER QUALITY ANALYSES

Analyses of water quality in the lakes used only data collected by SJRWMD. Water quality data were combined from multiple stations in the lakes. Stations used for the water quality analyses are listed in Table 2. The beginning of water quality data sets varies among the lakes; as early as 1983 for Lake Griffin, 1984 for Lake Weir, in the late-1980s for most of the lakes, and after 2000 for lakes Carlton and Trout. Data with Data Qualifier codes including numbers, >, J, L, M, O, Q, S, V, and Y were excluded from analysis. Definitions of FDEP-approved and internal SJRWMD Data Qualifier Codes are in Appendix A. Some J-coded (Estimated value; see Appendix A) chlorophyll data from the 1990s were included because the samples were not filtered within 24 hours of collection. The time frame required for filtration was later changed to 48 hours, so those data would not have been J-coded under current QA standards.

Table 2. Water quality stations used for trend analyses for the UORB lakes.

Lake	Water quality stations
Apopka	CLA, NLA, SLA
Beauclair	BCE, BCS, BCW
Dora	DOR, DORE, DORW
Harris-Little Harris	HAR, 20020377, LLHARRIS, LHAR
Eustis	20020368, EUS
Griffin	20020381, LGCA, LGN, LGS, LGR, LGRC, LGRN, LGW
Yale	LYC, 20020371
Weir	CLW
Denham	DNEY
Carlton	CARL
Trout	TRTL

BASIN OVERVIEW

For a basin overview of water quality, I calculated annual and multiannual averages through 2020 for concentrations of TP, corrected chlorophyll-*a* (corrected for presence of pheopigments – chlorophyll-*a* degradation products, https://floridadep.gov/sites/default/files/application-chlorophyll-a-methods_0.pdf), TN, Total Suspended Solids (TSS), and Secchi transparency. There was wide variability in water quality sampling frequency over the period of record. To avoid over-weighting days, months, and years in which more samples were collected, daily averages for all stations in each lake were calculated, those averages were used to calculate monthly averages, the monthly averages were used to calculate annual averages, and multiannual averages were developed from the annual averages. This basin overview of water quality compared averages for SJRWMD water quality data for the baseline period used for TMDL development, the most recent 5-year period 2016-2020, and target concentrations developed for the TMDLs (or the PLRG for Lake Weir).

TREND ANALYSES

Trends in water quality were evaluated for two time periods, from the beginning of the period of record for SJRWMD water quality data for the lakes, and for the years 2001-2019. SJRWMD water quality sampling did not begin until 1999 in Lake Carlton and 2004 in Trout Lake, so for those lakes I did trend analyses for only the post-2001 period. Due to SJRWMD restoration projects, major reductions in external nutrient loading to some of the lakes took place during the 1990s. The assessment for the 2001-2019 period evaluated whether further changes in water quality have occurred after those external load reductions. Trend analyses were conducted for TP, corrected chlorophyll-*a*, TN, TSS, Secchi transparency, ammonia-nitrogen (NH_4), nitrate-nitrite nitrogen (NO_x), and orthophosphate (PO_4). TN was calculated as the sum of Total Kjeldahl Nitrogen (TKN) and NO_x. Trend analyses were also done on total water column masses of TP, corrected chlorophyll-*a*, TN, and TSS. Masses were calculated by multiplying concentrations by estimated lake volumes, calculated from water elevations and morphometric data from the sources shown in Table 1. Analyses on water column masses were done because Coveney (2016) found differences between the trends in concentrations and masses for Lake Apopka.

Because sampling frequency has varied over the period of record, trend analyses could be biased by over-weighting of years with higher sample frequencies (Helsel and Hirsch 2002). Therefore, the trend analyses were performed using subsets of the available data, selected to approximately equalize sample frequency among years. In a few cases, data for entire years were deleted if only a few samples were collected. Most commonly, the subsetting resulted in sample sizes of approximately 6 per year for the period of record trend analyses, and 12 samples per year for the 2001-2019 trend analyses.

For most of the period included in this report, SJRWMD has reported machine-readings with K, T, or W Data Qualifier codes for data below laboratory method detection limits (MDL). Below detection limit results occur most commonly for NH_4, NO_x, and PO_4. In calculation of TN, I used machine readings when reported for NO_x measurements that were below MDL. The rationale for that approach is that I am aware of no way to get a better estimate for an individual non-detect measurement, and in such cases the machine reading for NO_x is so small relative to TKN that it has negligible effect on the calculated TN. For trend analyses of data sets with few (<5%) measurements below MDL, I used the machine readings for the below-detection limit data. Helsel and Hirsch (2002) note the presence of only a few nondetected values in a record (less than about 5%) is not likely to affect the accuracy of the trend slope magnitude significantly. For data sets with larger numbers of measurements below MDL, special methods developed for censored data (Helsel 2012) were used, as described further below.

A more complete trend analysis was done with data sets with no or few (<5%) measurements below MDL. The basin has undergone some significant droughts in the last 20 years, and there is evidence that changes in water elevations have affected water quality in the lakes. These effects of water elevation changes could affect assessments of temporal trends in water quality. To adjust for effects of water elevation changes, I used the R function loess to compute Lowess (Locally Weighted Scatterplot Smoothing) regressions between water elevation and water quality measures. The Lowess regressions used a smoothing factor of 0.5. The Engineering Statistics Handbook (NIST/SEMATECH 2013) e-Handbook of Statistical Methods) states “Useful values of the smoothing parameter typically lie in the range 0.25 to 0.5 for most LOESS applications” (https://www.itl.nist.gov/div898/handbook/pmd/section1/pmd144.htm). Subsequent trend analyses were done on both the original water quality measurements and the residuals from the Lowess regression. The next step in the trend analysis was a test for seasonality, using a monthly seasonality (i.e. 12 seasons per year), with a Kruskal-Wallis test in the R EnvStats package (Millard 2013, 2018).

If the Kruskal-Wallis test indicated no statistically significant ($p \geq 0.05$) seasonality, I used the R mkar1 script (method described in Matalas and Langbein 1962) to compute Kendall’s tau and the Theil-Sen trend line and to correct the significance level of the Kendall’s tau for serial correlation. The mkar1 script also outputs an Auto Correlation Factor (ACF) plot that can be used as an indicator of serial correlation. If there was significant seasonality, the R script rkt (Marchetto 2015) was performed, to compute the Seasonal Kendall’s tau and slope of the Seasonal Kendall’s trend line, and to correct the significance level of the Seasonal Kendall’s tau for serial correlation (method described in Helsel and Hirsch 2002). The rkt script does not produce the intercept for the trend line, so in cases in which I wanted to plot a trend line I calculated the intercept as given by Helsel and Hirsch (2002:

$$\text{Intercept } = \text{Y median} - \text{X median} \times \text{slope}$$

For data sets with a substantial number (>15%) of measurements below MDL, the MDLs were substituted for the machine readings. In some cases, the MDLs were not available, so in those cases I substituted MDLs from prior or subsequent dates. An accompanying variable indicated whether each measurement was censored (below MDL). To adjust for effects of water elevation changes, I used the R function loess to compute Lowess (Locally Weighted Scatterplot Smoothing) regressions between water elevation and water quality measures. The Lowess regressions used a smoothing factor of 0.5. Subsequent trend analyses were done on both the original water quality measurements and the residuals from the Lowess regression.

Options for trend analysis with censored data are more limited. I am aware of no programs that will do a Seasonal Kendall test or correct for serial correlation with censored data. The first step was an Akritas-Theil-Sen (ATS) nonparametric regression for left-censored data to determine the apparent nonseasonal trends in unadjusted parameters and residuals using the R cenken script (part of the NADA package, Lee 2017). I also ran a Kruskal-Wallis test for censored data using the Minitab CensKW macro (method described in Helsel 2012; script downloadable from http://practicalstats.com/nada/downloads.html) on the unadjusted parameters and residuals to determine if seasonality is an issue. Seasonality adds variability that is unaccounted for in the nonseasonal ATS regression, so when seasonality is present the variance is inflated such that the test might not find a significant trend where one in fact exists. A finding of a significant trend is reliable, but it is possible that a trend exists and is masked by seasonal variability; applying a Seasonal Kendall test would allow the test to find the significant trend (D. Helsel, July 22, 2013, written response to questions on July 15, 2013 AES6 "Trend Analysis" Webinar).

For data sets with a moderate number (5-15%) of measurements below MDL, I ran both the procedures described above for few and many non-detects and compared the results.

For the trend tests on water column masses, there was no adjustment for water elevations, because those are already incorporated into the calculated masses. Otherwise, the same methods were used for water column masses as done with water quality parameters with no or few (<5%) measurements below MDL.

Over the period of record, for most of the lakes either total or dissolved fractions were measured for NH_4, NO_x, and PO_4, with total measured generally before 2005 and dissolved measured generally from the late 1990s to the present. For the 2001-2019 period, I conducted trend analyses for only the dissolved fractions of these nutrients. For lakes Apopka, Beauclair, and Denham, dissolved fractions were measured for the entire period of record, so I conducted trend analyses for the entire period of record using the dissolved fractions for those lakes. SJRWMD water quality sampling did not begin until 1999 in Lake Carlton and 2004 in Trout Lake, so for those lakes I did trend analyses for only the post-2001 period, using only the dissolved fractions. For the other lakes, I examined the data to determine whether the total and dissolved fractions could be combined for trend analysis for the entire period of record. Data plots indicated that PO_4-T was consistently higher than PO_4-D, but there was substantial overlap for total and dissolved fractions of NH_4 and NO_x (for example, plots for Lake Griffin are shown in Figure 3 – Figure 5). Since these data sets included many measurements below MDL, I used the R NADA censtats script to calculate summary statistics for the total and dissolved fractions. In Table 3, I report summary statistics produced by "Robust regression on order statistics" (ROS) (Helsel 2012). I used the Paired Prentice-Wilcoxon test (Minitab PPW macro) for equality of paired left-censored data to test whether the dissolved and total fractions differ. If they were significantly different, I used ATS nonparametric regression for left-censored data (R cenken script) to develop a regression line predicting the total fraction from the dissolved fraction. I used this "Synthetic" total fraction for period of record trend analysis. The data set used for the period of record trend analysis included the total fraction when it was measured, and the Synthetic total fraction when only the dissolved fraction was measured.

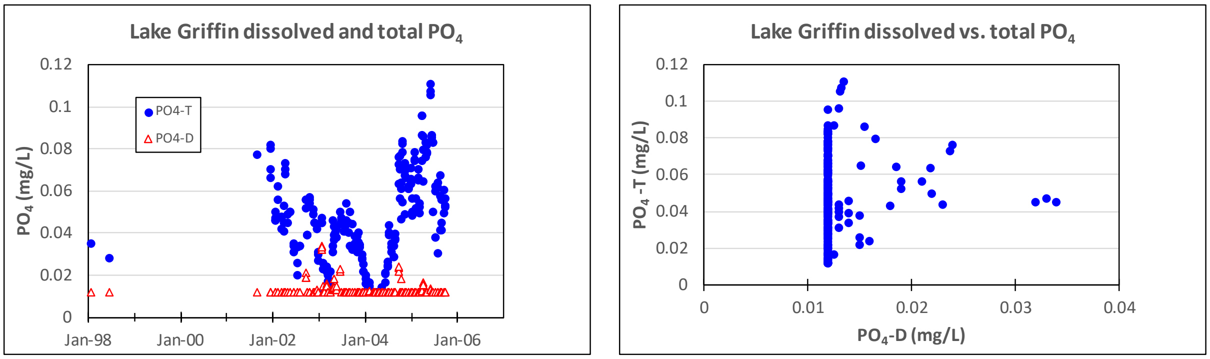


Figure 3. Lake Griffin dissolved and total PO_4.

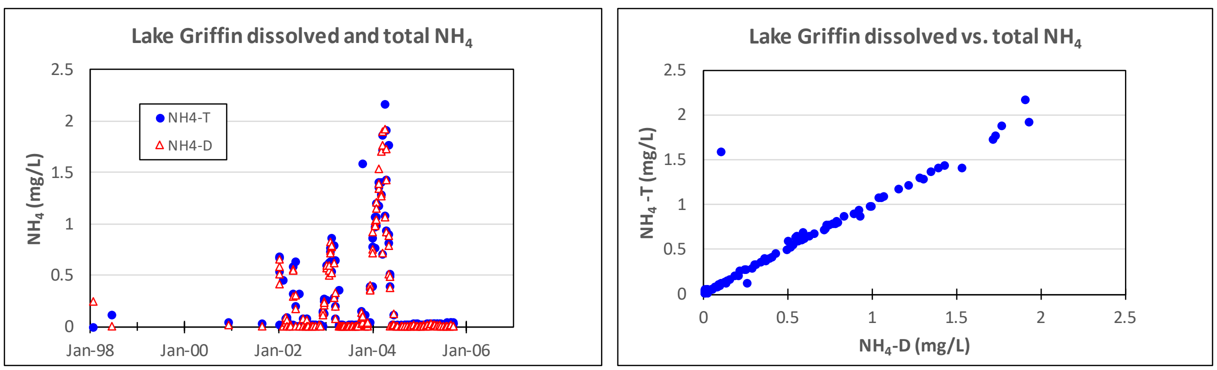


Figure 4. Lake Griffin dissolved and total NH_4.

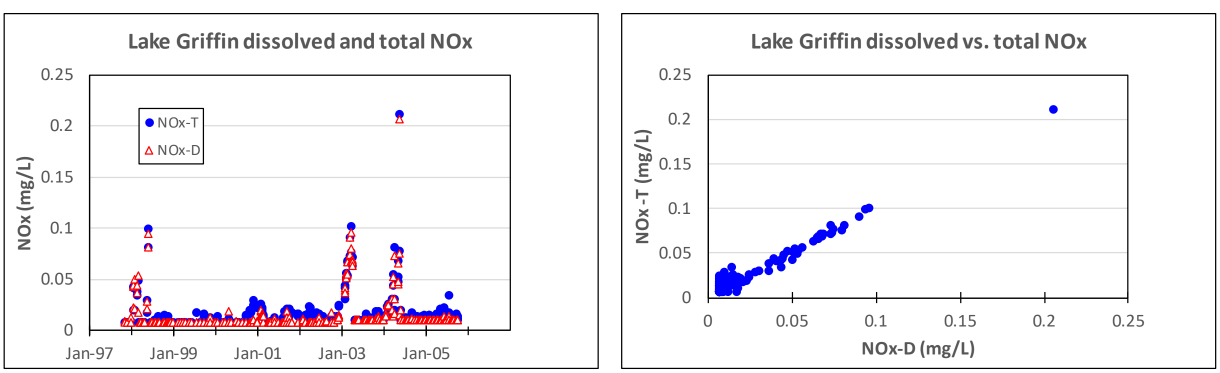


Figure 5. Lake Griffin dissolved and total NO_x.

Other than for NO_x in Lake Yale and NH_4 in Lake Weir, concentrations of the total fractions were significantly greater than the dissolved fractions (Table 3). Regression relationships between the total and dissolved fractions were nearly always significant, but showed stronger linear relationships for NH_4 and NO_x than for PO_4 (for example, plots for Lake Griffin are shown in Figure 3 – Figure 5). I decided that the differences between PO_4-T and PO_4-D were too large and the regression relationships between them were too weak to combine those data for trend analysis. However, for NO_x, and NH_4, I constructed synthetic total fractions for trend analyses for the entire period of record using the regression relationships (Table 3) to convert the dissolved fraction measurements to total fraction estimates. Those synthetic total fraction estimates were combined with the total fraction measurements for trend assessment. For NO_x in Lake Yale and for NH_4 in Lake Weir there were no significant differences between the two fractions, so I directly combined the total and dissolved measurements with no adjustment.

Ammonia-nitrogen (NH_4) measurements were compared with the Florida Total Ammonia Nitrogen (TAN) criterion. The TAN criterion is variable, depending on water temperature and pH. The criterion states that the 30-day average TAN value shall not exceed the average of the values calculated from an equation that includes water temperature and pH, with no single value exceeding 2.5 times the value from the equation (62-302.530 F.A.C.). In plots of NH_4, values which exceed the 30-day average criterion or 2.5 times the criterion are highlighted.

Table 3. Statistical analysis of paired measurements of total vs. dissolved PO_4, NO_x, and NH_4.

There were insufficient numbers of paired measurements for statistical analyses for lakes Apopka, Beauclair, Denham, Carlton, or Trout, or for PO_4 and NO_x in Lake Weir.

PO_4-T vs. PO_4-D					
Lake	**Number of pairs**	**ROS Median total (mg/L)**	**ROS Median dissolved (mg/L)**	**Paired PPW test p value**	**PO_4-T vs. PO_4-D ATS regression p value**
Dora	65	0.056	0.004	0.000	0.003
Harris-Little Harris	69	0.024	0.011	0.000	0.000
Eustis	30	0.029	0.004	0.000	0.000
Yale	49	0.022	0.005	0.000	0.010
Griffin	235	0.045	0.006	0.000	0.013
NO_x -T vs. NO_x -D					
Lake	**Number of pairs**	**ROS Median total (mg/L)**	**ROS Median dissolved (mg/L)**	**Paired PPW test p value**	**NO_x -T vs. NO_x -D ATS regression p value**
Dora	124	0.012	0.004	0.000	0.000
Harris-Little Harris	84	0.017	0.013	0.000	0.000
Eustis	57	0.008	0.006	0.017	0.000
Yale	73	0.008	0.005	0.054	0.000
Griffin	414	0.011	0.003	0.000	0.000
NH_4-T vs. NH_4-D					
Lake	**Number of pairs**	**ROS Median total (mg/L)**	**ROS Median dissolved (mg/L)**	**Paired PPW test p value**	**NH_4-T vs. NH_4-D ATS regression p value**
Dora	99	0.026	0.015	0.000	0.000
Harris-Little Harris	123	0.009	0.006	0.000	0.000
Eustis	41	0.021	0.020	0.000	0.000
Yale	69	0.016	0.014	0.009	0.000
Griffin	269	0.025	0.014	0.000	0.000
Weir	23	0.004	0.007	0.383	0.089

PHYTOPLANKTON ANALYSES

Phytoplankton biovolume and composition data are also summarized for the UORB lakes. Phytoplankton samples were collected from the basin lakes in conjunction with water quality sampling. Until 2015, samples were composites of subsamples collected with a horizontal Van Dorn sampler at the surface, 0.5, and 1 m depths. Beginning in 2015, vertically integrated samples over the top 1 m were collected with a tube sampler. Phytoplankton samples were preserved with Lugol's solution and shipped to contracted analysts.

Formal trend analysis was not performed for the phytoplankton data because of apparent significant differences in identification and biovolume estimates by different analysts. For Lake Apopka, phytoplankton analyses were performed by Joanne Burkholder for 1989-1994, by Phycotech Inc. from 1994-2011, and by BSA Inc. from 2011-2020. For the other lakes, quantitative phytoplankton monitoring stopped in 2018. For Lake Beauclair, phytoplankton analyses were performed by Joanne Burkholder for 1989-1993, by Phycotech Inc. from 1998-2011, and by BSA Inc. from 2011-2018. For Lake Denham, phytoplankton analyses were performed by Joanne Burkholder for 1989-1994, by Phycotech Inc. from 1994-2007, and by BSA Inc. from 2007-2018. For the other lakes, phytoplankton analyses were performed by Phycotech Inc. from the mid-late 1990s-2007, and by BSA Inc. from 2007-2018. Two phytoplankton time-series figures are shown for each lake. One shows temporal trends in biovolumes of the major phytoplankton Divisions and mean annual chlorophyll-*a* concentrations. The other figure shows time-series of percent composition of the major cyanobacteria genera, other cyanobacteria, and other taxa (all other phytoplankton Divisions combined). After the cessation of quantitative phytoplankton monitoring, occasional samples were collected from the lakes when algal blooms were observed. These samples were analyzed for cyanotoxins and a qualitative assessment of the dominant phytoplankton species.

RESULTS AND DISCUSSION

EXTERNAL NUTRIENT LOADING OVERVIEW

Average external TP loads to the lakes are shown in Figure 6. The TMDL baseline period TP loads were highest for Lake Apopka, and lowest for lakes Yale and Weir (Figure 6A). However, the ability of a lake to effectively assimilate nutrients is related to the size of the lake. When external TP loading is expressed relative to lake surface area, Lake Beauclair had baseline loads almost five times higher than any other lake (Figure 6B).

In the 2015-2019 period, external TP loads were reduced in all the lakes except Yale and Weir. The largest load reductions were almost 80% for lakes Beauclair and Apopka, 72% for Lake Dora, and 62% for Lake Griffin. Average external loads during the 2015-2019 period were below the adopted TMDLs for lakes Apopka and Dora, and nearly equal to the adopted TMDL for Lake Eustis.

Average external TN loads to the lakes are shown in Figure 7. External TN loads were not estimated for Lake Apopka during the TMDL baseline period. For the other lakes, the TMDL baseline (1991-2000) external TN loads were highest for lakes Eustis and Griffin, and lowest for lakes Yale and Weir (Figure 7A). However, when external TN loading is expressed relative to lake surface area, Lake Beauclair had baseline loads more than four times higher than any other lake (Figure 7B). In the 2015-2019 period, TN loads were reduced in all the lakes except Harris, Yale, and Weir. The largest load reductions were nearly 60% for lakes Beauclair and Dora, and nearly 45% for Lake Eustis.

In Figure 8, above zero on the vertical axis are cumulative external TP loads for Lake Apopka over the period 1995-2019, divided into loads from the Lake Apopka North Shore and all other sources, and below zero on the vertical scale are the estimated TP reductions. About 64% of the total external load came from the LANS during this period. The TMDL was not met for cumulative loading to Lake Apopka over this entire period (although a later figure will show it has been met in many individual years). However, the cumulative TP load from the LANS was reduced by nearly 1,000 metric tons compared to the average for the TMDL baseline period, a reduction of more than 70%. This estimated load reduction from the LANS was almost three times the actual TP load from the LANS during this period, so without those reductions the TP load from the LANS would have been nearly four times higher. The external load reductions from the LANS were about ten times larger than TP removals from the lake by the shad harvest, which were about three times larger than those by the marsh flow-way.

In Figure 9, above zero on the vertical axis are cumulative external TP loads for the lakes downstream of Lake Apopka for the period 2000-2019, divided into loads from Lake Apopka discharges and all other sources. Below zero on the vertical scale are the estimated TP reductions. The adopted TMDLs were not met over this period for lakes Beauclair and Harris but were for the other three lakes. None of the lakes would have met their TMDLs without the load reductions by the restoration projects.

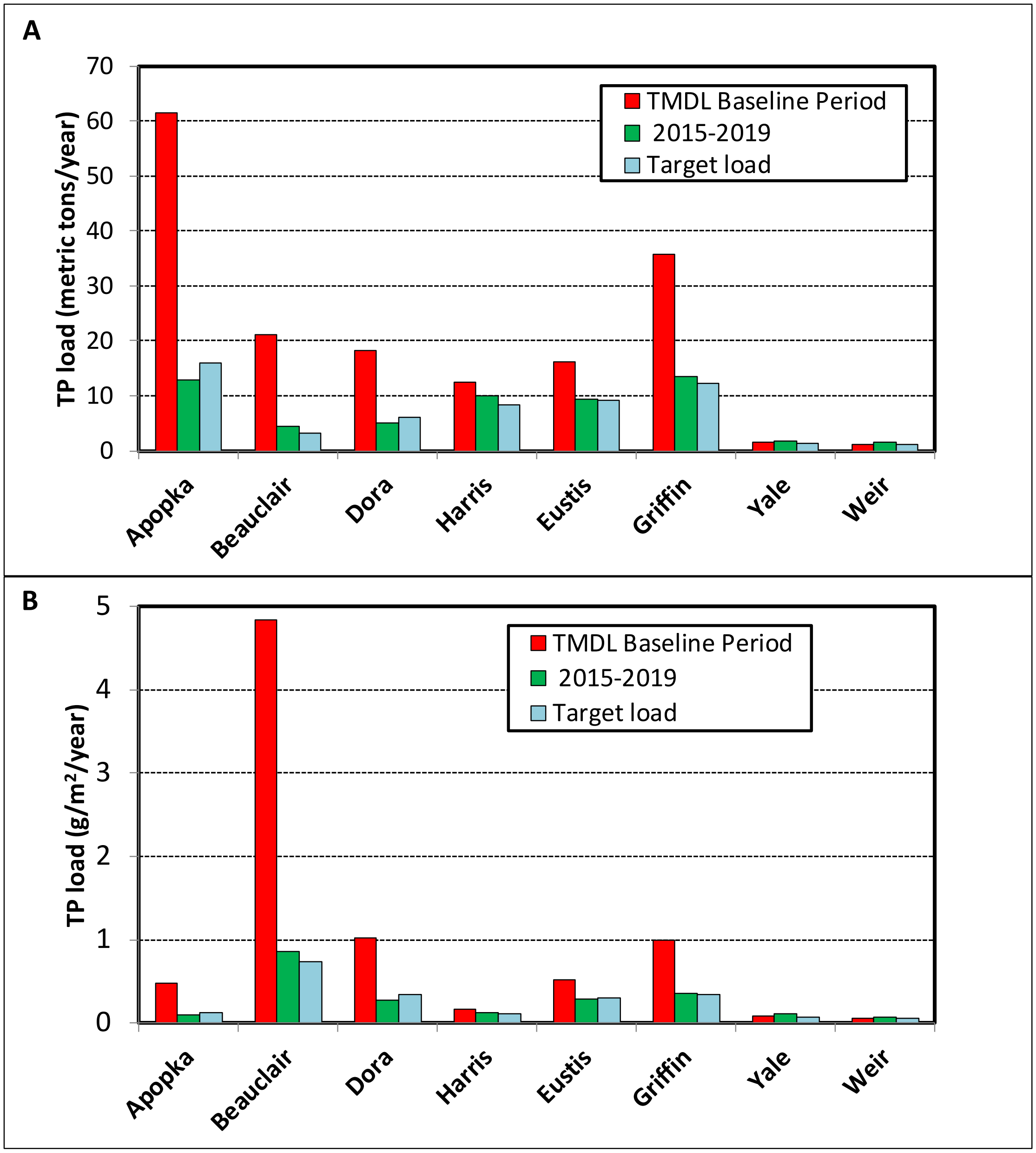


Figure 6. External total phosphorus loads to the Upper Ocklawaha Basin lakes. A. Mass loads. B. Loads per unit lake surface area.

Target loads are the PLRG target for Lake Weir and TMDL targets for the other lakes.

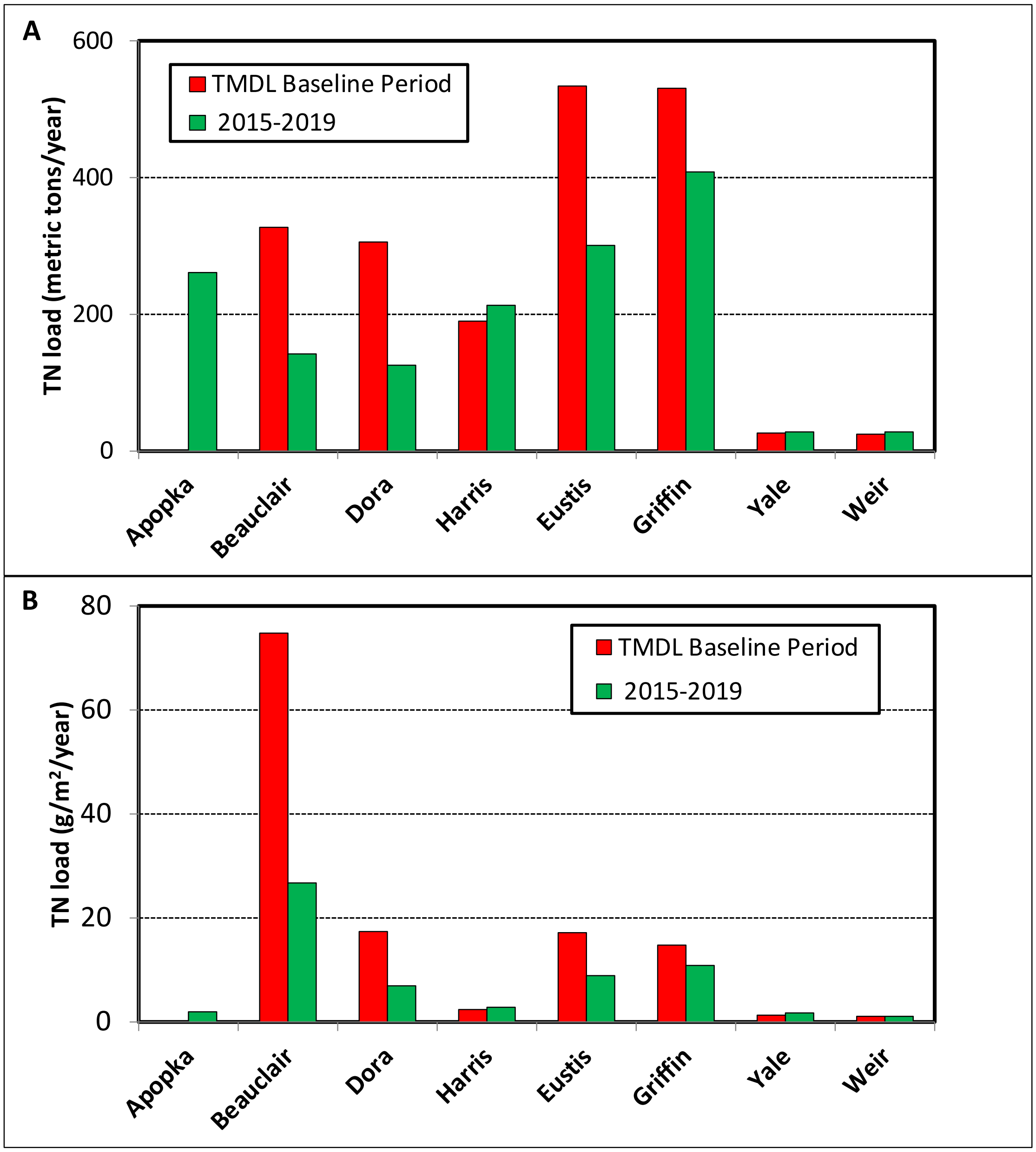


Figure 7. External total nitrogen loads to the Upper Ocklawaha Basin lakes. A. Mass loads. B. Loads per unit lake surface area.

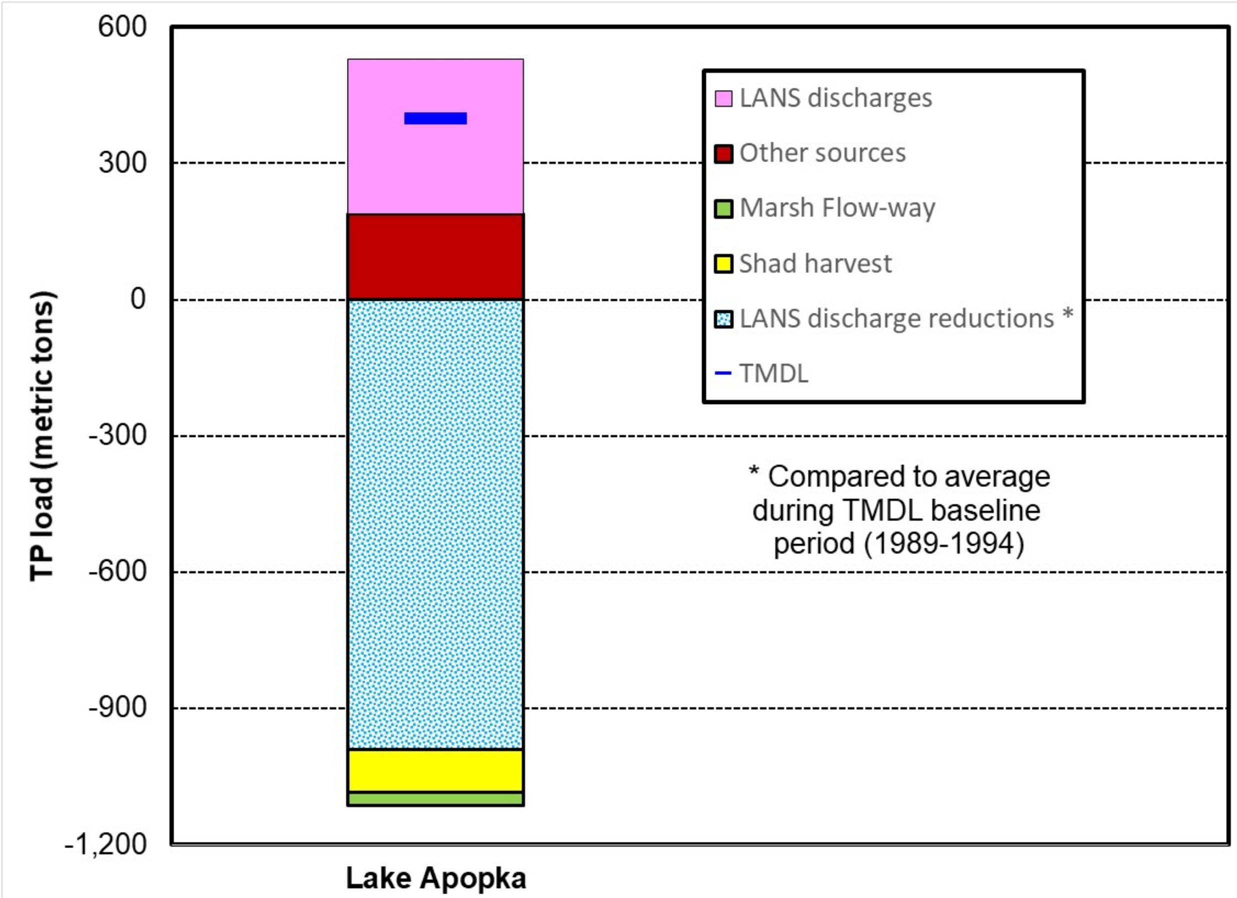


Figure 8. Estimated cumulative external total phosphorus loads and reductions due to restoration projects for Lake Apopka 1995-2019.

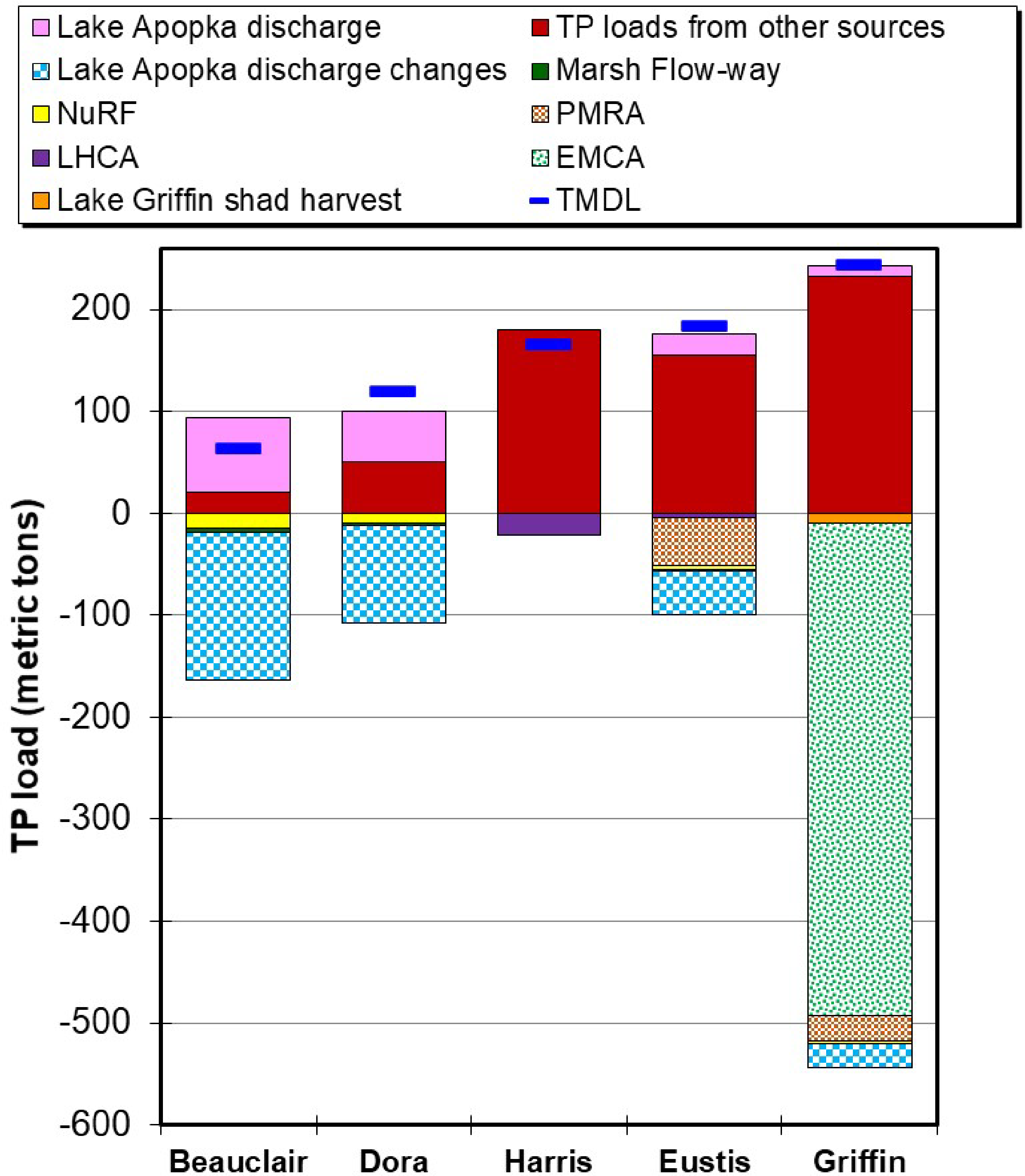


Figure 9. Estimated cumulative external total phosphorus loads and reductions due to restoration projects for the UORB lakes 2000-2019.

For Lake Beauclair, about 79% of the external TP load for 2000-2019 came from Lake Apopka. The estimated TP loads to Lake Beauclair were reduced by about 64%, so without these reductions the load would have been more than twice as high during this period. About 89% of the reductions over this 20-year period were due to the changes in Lake Apopka discharge concentrations, and about 9% due to the NuRF project and 2% due to the marsh flow-way. The impacts of both the flow-way and the NuRF were limited because there were low or no downstream discharges from Lake Apopka for several years, due to persistent drought conditions. During those years, nearly all the flow-way discharges went back into Lake Apopka, and the NuRF project could operate only infrequently.

For Lake Dora, the loading derived from Lake Apopka discharges is reduced because of nutrient retention in Lake Beauclair and addition of other nutrient sources; an estimated 49% of the external TP load to Lake Dora came from Lake Apopka. The estimated TP loads to Lake Dora were reduced by about 52% over this period, and as with Beauclair, 89% of that reduction was due to the reduced Apopka discharge concentrations, and about 9% due to the NuRF project and 2% due to the marsh flow-way.

For Lake Harris, I assumed that there were no significant nutrient flows from Lake Eustis to Harris, so Lake Apopka discharges have no impact on external loading to Lake Harris. The only project affecting loading to Lake Harris is the Lake Harris Conservation Area, which reduced estimated loading by about 10%.

For Lake Eustis, discharges from Lake Apopka account for only 11% of the external TP load. The estimated TP loads to Lake Eustis were reduced by about 36%. About 46% of that reduction was due to the Pine Meadows Restoration Area, and about 43% due to reduced Lake Apopka discharge concentrations. The other projects had small contributions to the load reductions to Lake Eustis; 5% for Lake Harris Conservation Area, 4% for the NuRF, and 1% for the marsh flow-way.

Finally, for Lake Griffin, discharges from Lake Apopka account for only 4% of the external TP load. The estimated TP loads to Lake Griffin were reduced by about 70%. About 91% of that reduction was due to reduced discharges from the Emeralda Marsh Conservation Area, about 5% was due to the Pine Meadows Restoration Area, and about 4% was due to reduced Lake Apopka discharge concentrations. TP removals from the lake in the shad harvest were about 4% of the external load during this period.

WATER QUALITY OVERVIEW

Mean total phosphorus concentrations for the basin lakes are shown in Figure 10, and in Appendix C, Table C- 1. Baseline TP concentrations were highest in lakes Apopka, Beauclair, Trout, Griffin, and Dora. Average concentrations were reduced in the 2016–2020 period for eight of the lakes. The largest percent reductions were for lakes Beauclair (77%), Griffin (67%), Apopka (65%), and Dora (62%). TP concentrations increased in the 2016–2020 period for lakes Weir (41%), Yale (35%), and Trout (9%). The average TP concentrations for the 2016-2020 period nearly met the TMDL targets for lakes Harris (27 vs. a target of 26 μg/L) and Eustis (26 vs. a target of 25 μg/L). The 2016-2020 averages were also close to the TMDL targets for lakes Griffin (34 vs. a target of 32 μg/L) and Dora (36 vs. a target of 31 μg/L).

Corrected chlorophyll-*a* concentrations for the basin lakes are shown in Figure 11, and in Appendix C, Table C- 2. Baseline chlorophyll-*a* concentrations were highest in lakes Carlton, Beauclair, Griffin, and Dora. Average concentrations were reduced in the 2016–2020 period for ten of the lakes. The largest percent reductions were for lakes Griffin (80%), Beauclair (77%), Dora (76%), Harris (69%), and Carlton (69%). Chlorophyll-*a* concentrations increased in the 2016–2020 period for Lake Weir (78%). Average chlorophyll concentrations for the 2016-2020 period met those expected if the TMDL TP concentration targets are met in lakes Harris, Eustis, and Griffin. The 2016–2020 averages were close to the TMDL expected concentrations for Yale (16 vs. an expectation of 14 μg/L), Dora (32 vs. an expectation of 29 μg/L), and Beauclair (35 vs. an expectation of 30 μg/L). Expected concentrations were not established for Lake Apopka.

Secchi depth transparency means for the basin lakes are shown in Figure 12, and in Appendix C, Table C- 3. Baseline Secchi depths were lowest in lakes Apopka, Griffin, Beauclair, and Carlton. Average Secchi depths increased in the 2016–2020 period for nine of the lakes. The largest percent increases were for lakes Griffin (148%), Beauclair (121%), and Dora (102%). Average Secchi depths decreased in the 2016–2020 period for lakes Yale (27%) and Weir (21%). Average Secchi depths for the 2016–2020 period reached the depths expected if the TMDL TP concentration targets are met for Lake Harris. The 2016–2020 averages were close to the TMDL expected depths for lakes Griffin (0.706 vs. an expectation of 0.71 m), Dora (0.72 vs. an expectation of 0.73 m), Beauclair (0.68 vs. an expectation of 0.71 m), and (Eustis (0.82 vs. an expectation of 0.87 m). Expected Secchi depths were not established for lakes Apopka, Weir, Trout, and Denham.

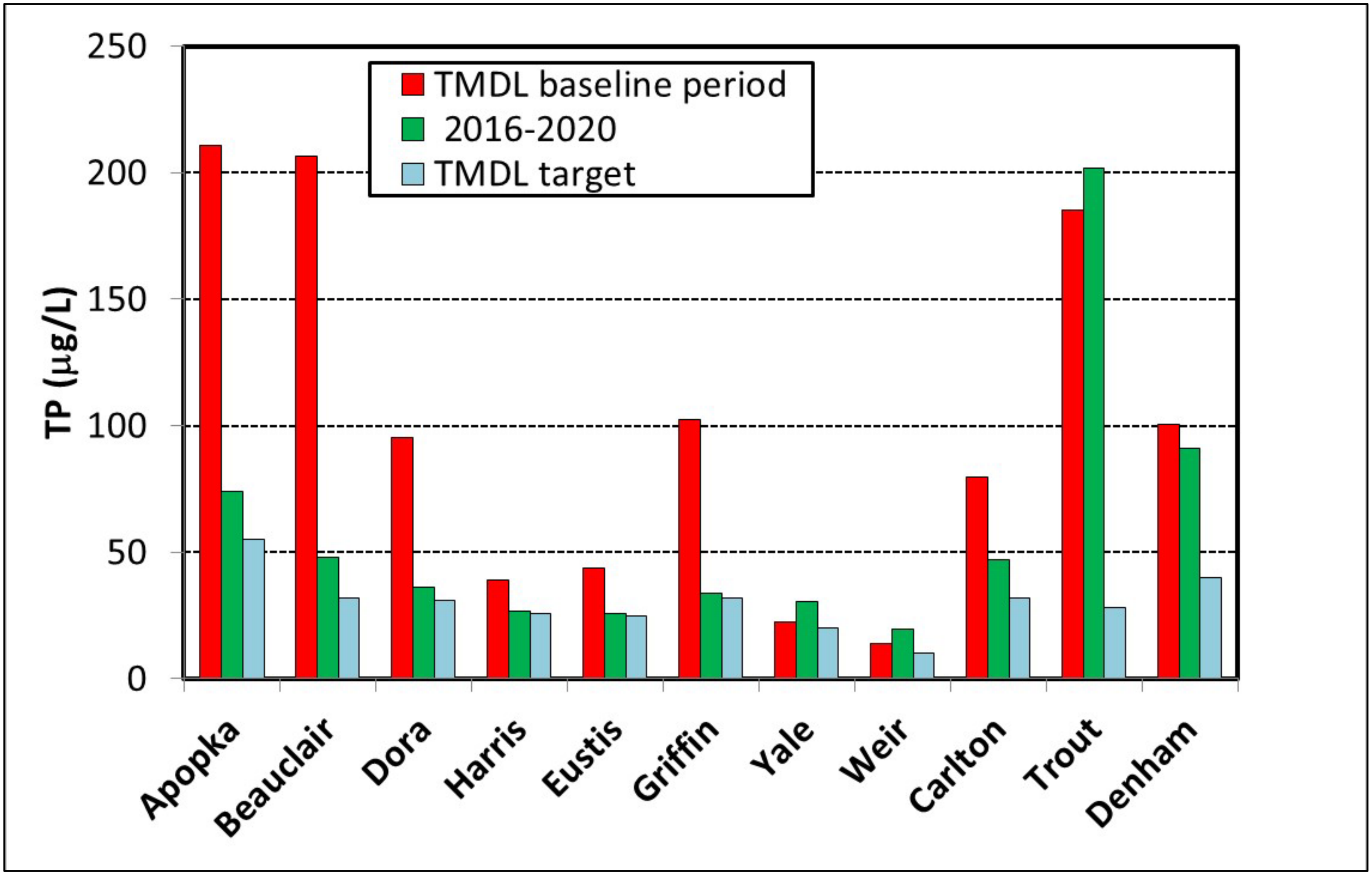


Figure 10. Mean total phosphorus concentrations in Upper Ocklawaha Basin lakes.

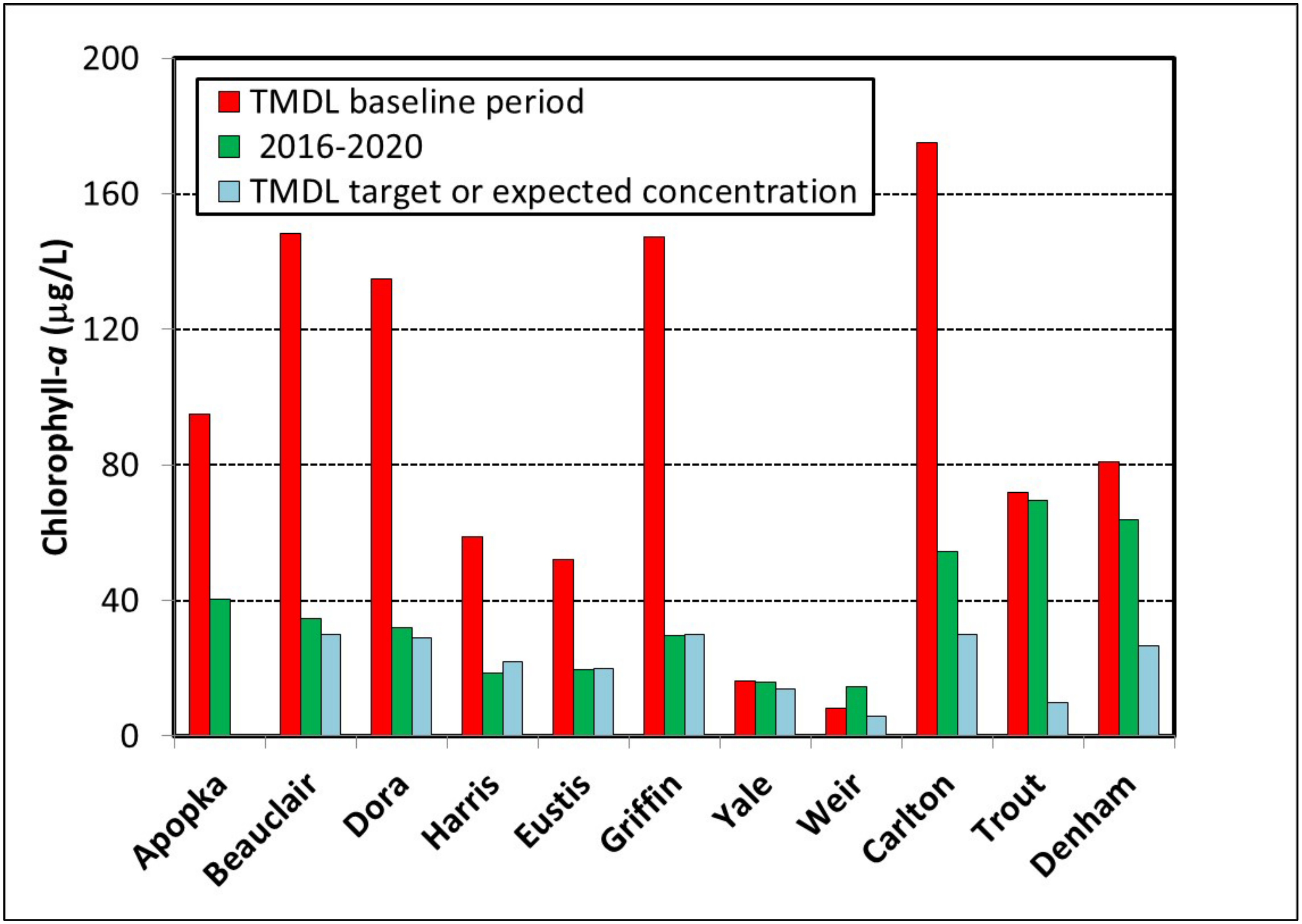


Figure 11. Mean corrected chlorophyll-*a* concentrations in Upper Ocklawaha Basin lakes.

Expected concentrations are those at the TMDL target TP concentration; not determined for Lake Apopka.

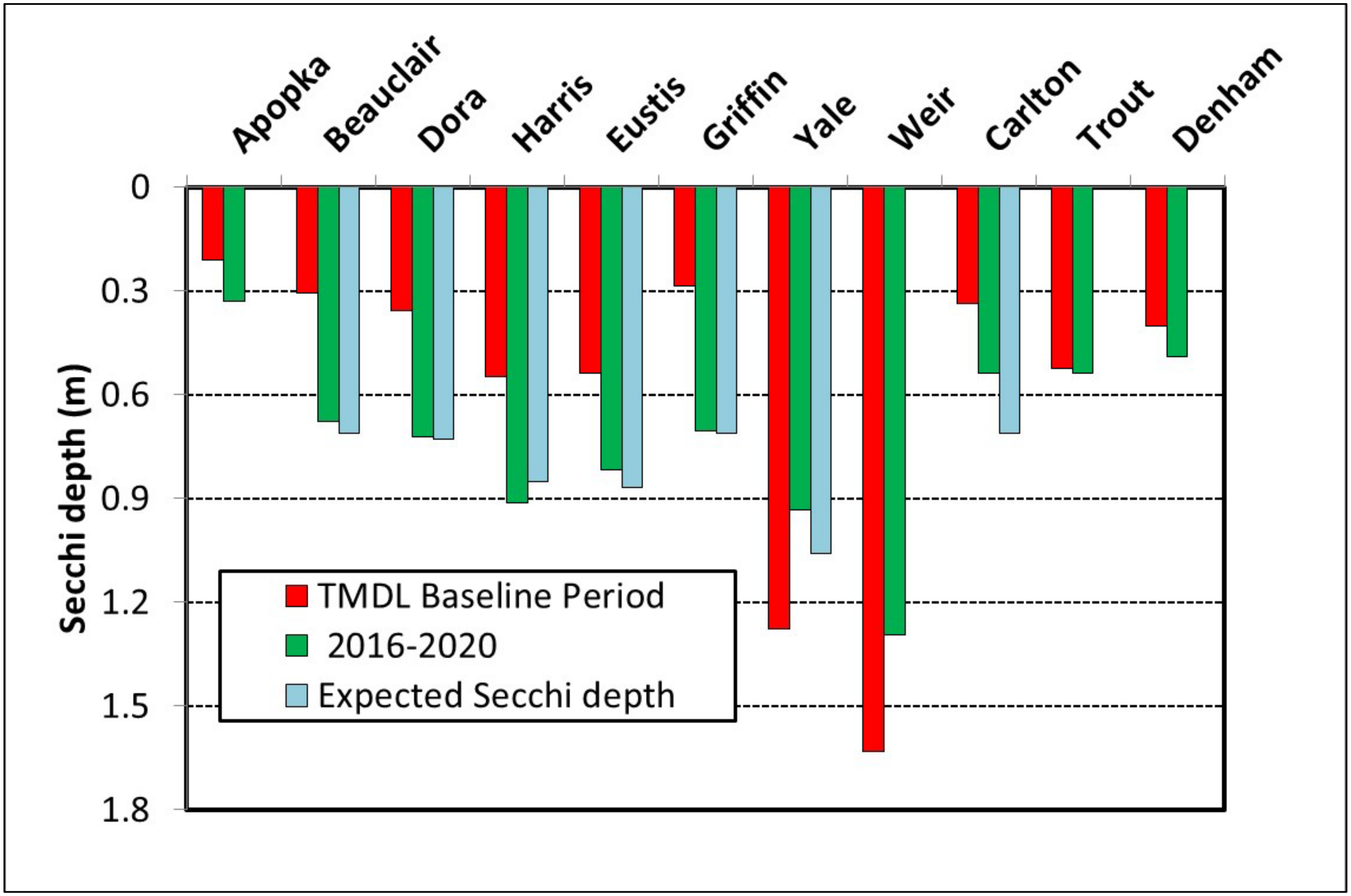


Figure 12. Mean Secchi transparency in Upper Ocklawaha Basin lakes.

Expected Secchi depths are those at the TMDL TP targets; not determined for lakes Apopka, Weir, Trout, or Denham.

Total nitrogen concentrations for the basin lakes are shown in Figure 13, and in Appendix C, Table C- 4. Baseline mean TN concentrations were highest in lakes Apopka, Griffin, Beauclair, and Dora. Average concentrations were reduced in the 2016–2020 period for eight of the lakes. The largest percent reductions were for lakes Griffin (63%), Dora (61%), and Beauclair (59%). TN concentrations increased in the 2016–2020 period for lakes Yale (13%), Weir (11%) and Trout (3%). Expected concentrations for TN if the TMDL TP concentration targets are met have not been established for the basin lakes.

Total suspended concentrations for the basin lakes are shown in Figure 14, and in Appendix C, Table C- 5. Baseline mean TSS concentrations were highest in lakes Apopka, Griffin, Beauclair, and Dora (no baseline period data were available for Trout Lake). Average concentrations were reduced in the 2016–2020 period for nine of the lakes. The largest percent reductions were for lakes Griffin (69%), Beauclair (66%), and Dora (63%). TSS concentrations increased in the 2016–2020 period for Lake Weir (20%).

All five water quality parameters improved in 2016–2020 in eight of the basin lakes, with the largest improvements generally for lakes Griffin, Beauclair, and Dora. These three lakes were affected by significant SJRWMD restoration projects. Lake Griffin was affected primarily by reductions in nutrient loads from the Emeralda Marsh Conservation Area and to a smaller extent by gizzard shad harvesting and projects in upstream basins (Figure 9). Lakes Beauclair and Dora were affected by upstream restoration actions including the Lake Apopka North Shore Restoration Area, the Lake Apopka marsh flow-way, and Lake County Water Authority’s NuRF project. There have also been smaller-scale stormwater projects constructed by local governments as part of implementation of the Basin Management Action Plan (BMAP) for these lakes.

In contrast, all five water quality parameters deteriorated in 2016–2020 in Lake Weir, 3 of the 5 parameters deteriorated in Lakes Yale, and 2 deteriorated in Trout Lake. There have been no significant restoration projects to date in the watersheds for lakes Weir and Yale, although Trout Lake is affected by the Pine Meadows Restoration Area project.

More information on trends in external loading and water quality for the basin lakes is presented in subsequent sections.

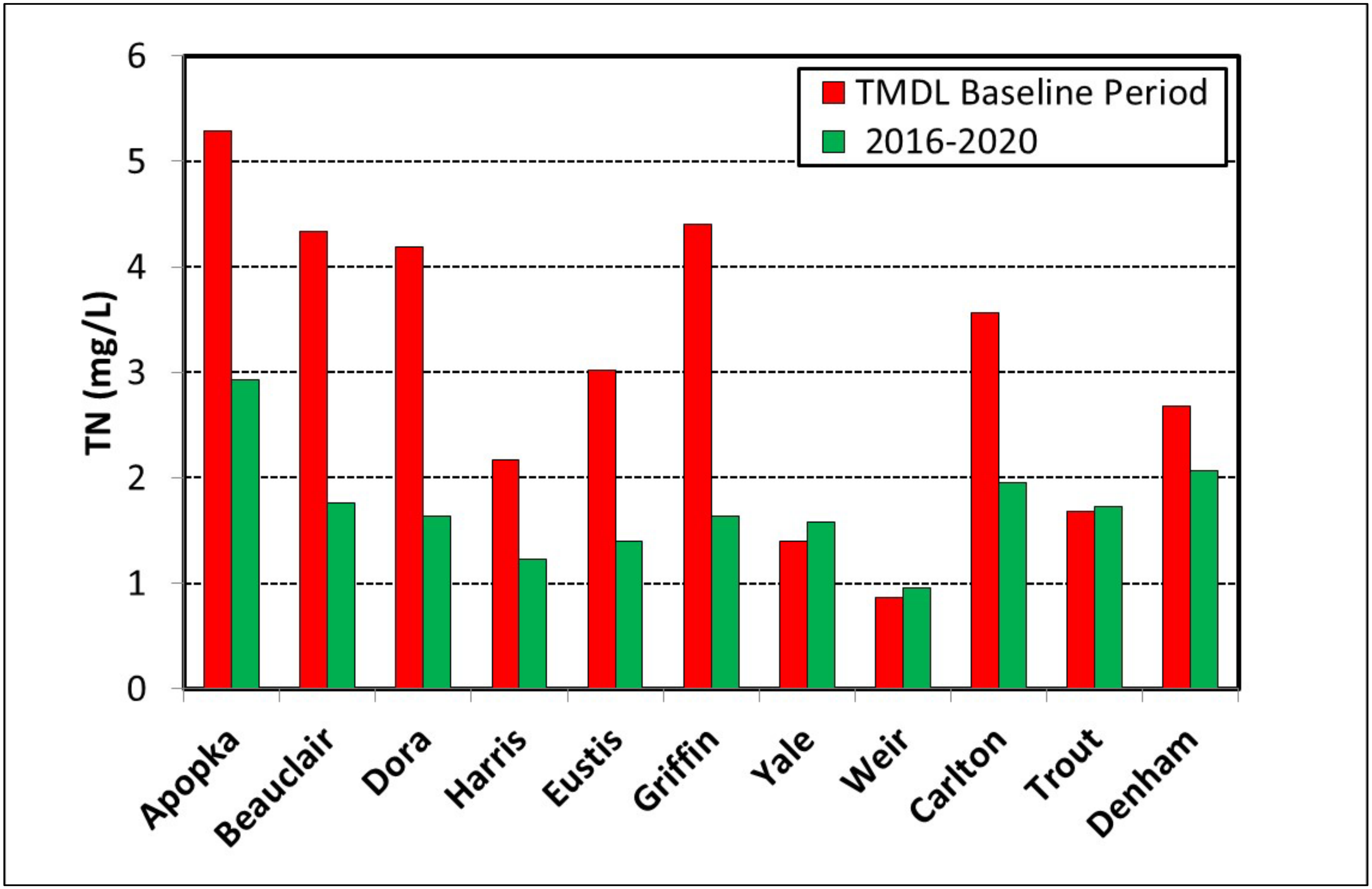


Figure 13. Mean total nitrogen concentrations in Upper Ocklawaha Basin lakes.

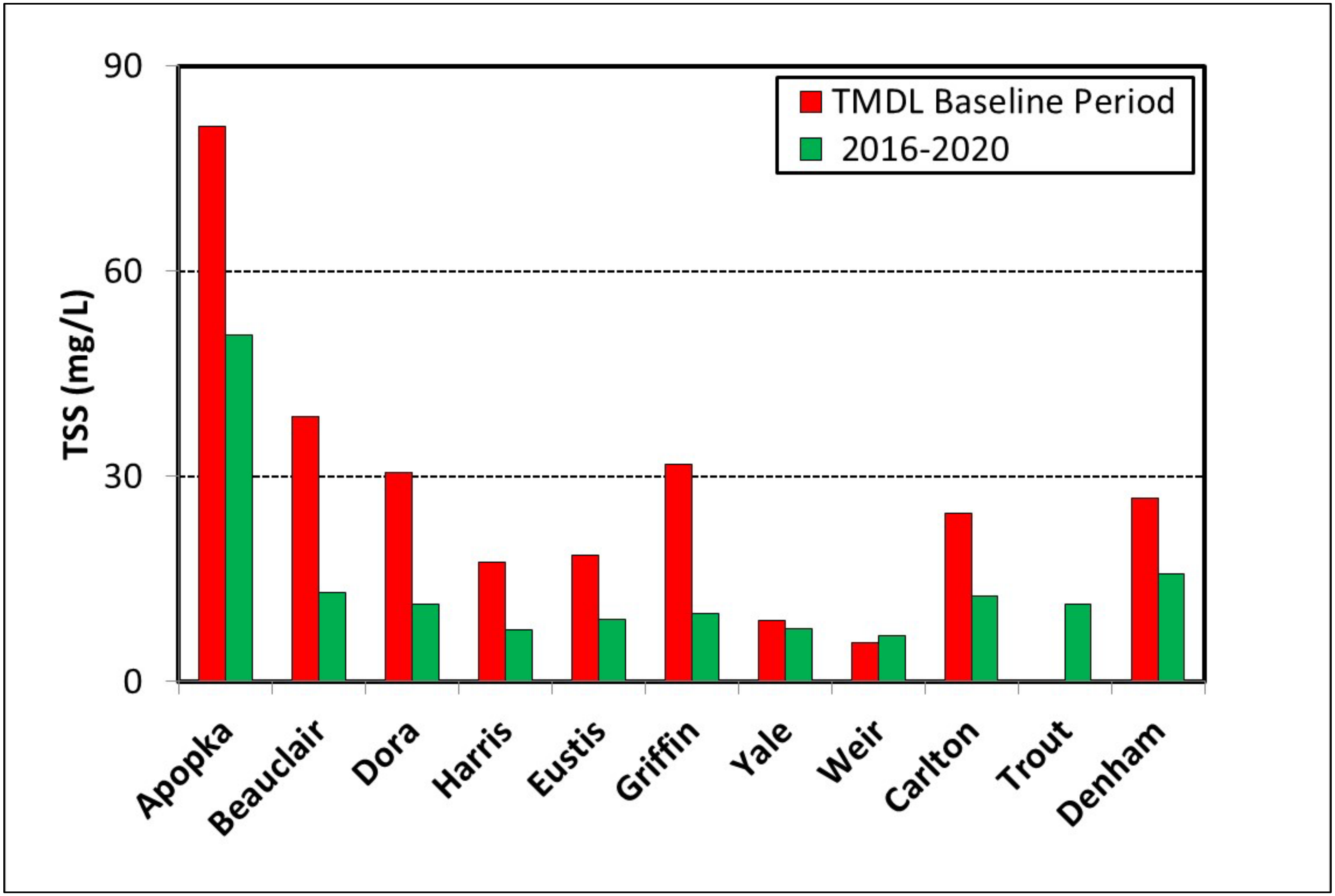


Figure 14. Mean total suspended solids concentrations in Upper Ocklawaha Basin lakes.

LAKE APOPKA

For much of the period of record, estimated external TP loading to Lake Apopka has been heavily dominated by discharges from the Lake Apopka North Shore, although there has been a clear decreasing trend (Figure 15, Appendix Table D- 1). Loading from the LANS was below its TMDL load allocation (5.53 MT/year) in the dry years 2000-2001 and 2006-2007, as well as the last nine years. TP loading from the LANS has been somewhat higher the last three years, due primarily to Hurricane Irma in 2017, and continued high rainfall the next two years. For 8 of the last 9 years, the external TP load to the lake has been below the adopted TMDL for the lake (15.9 MT/year), slightly exceeding the TMDL in 2017.

The major external TP sources to Lake Apopka for the last 5 years have been atmospheric deposition (40.0%), the LANS (22.8%), basin runoff (19.4%), and Apopka Springs discharges (5.8%). The basin runoff in the Lake Apopka nutrient budget is estimated for only part of the watershed. Loading to Lake Apopka from the part of the watershed that drains to Johns Lake is included in tributary discharges from that lake. Unlike the other lakes, basin runoff to Lake Apopka from the other parts of the watershed was not divided by source until 2011. Over the last 5 years, TP loading in basin runoff from the parts of the watershed not draining to Johns Lake was divided as 12.0% of the total load from urban-residential, 6.0% from natural areas, and 1.4% from agricultural lands.

External TN loading to Lake Apopka has been estimated only since 2012 (Figure 16, Appendix Table D- 2). Over the most recent 5-year period the major TN sources for Lake Apopka have been atmospheric deposition (32.2%), Apopka Spring discharges (26.5%), LANS discharges (16.3%), runoff from natural areas (7.0%), and urban-residential runoff (5.1%).

Concentrations of TP, TN, and chlorophyll-*a* generally decreased in Lake Apopka during the 1990s. However, concentrations increased at low lake water levels during three severe multi-year droughts since 2000, then recovered when lake stages subsequently recovered to more normal elevations (Figure 17, Figure 18). The lake lost 72% of its mean volume in the 2002 drought and about 50% in 2008 and 2012 (Coveney 2016). However, there were smaller increases in masses of TP, TN, and chlorophyll-*a* during these drought periods, suggesting that the increases in concentrations were due to relatively stable masses becoming concentrated in a smaller lake volume (Coveney 2016). In contrast, for TSS there were large increases in both concentrations and masses during the 2008 and 2012 droughts (Figure 18).

Water quality generally improved over the last five years, after lake stages recovered from the most recent severe drought. Average TP concentrations in 2016 and 2020 were the lowest in the period of record, and only slightly higher in 2018-19, close to the TMDL target of 55 µg/L (Figure 17). Average concentrations of TN, chlorophyll-*a*, and TSS have also been close to or at their period of record minima in the last 3 to 5 years.

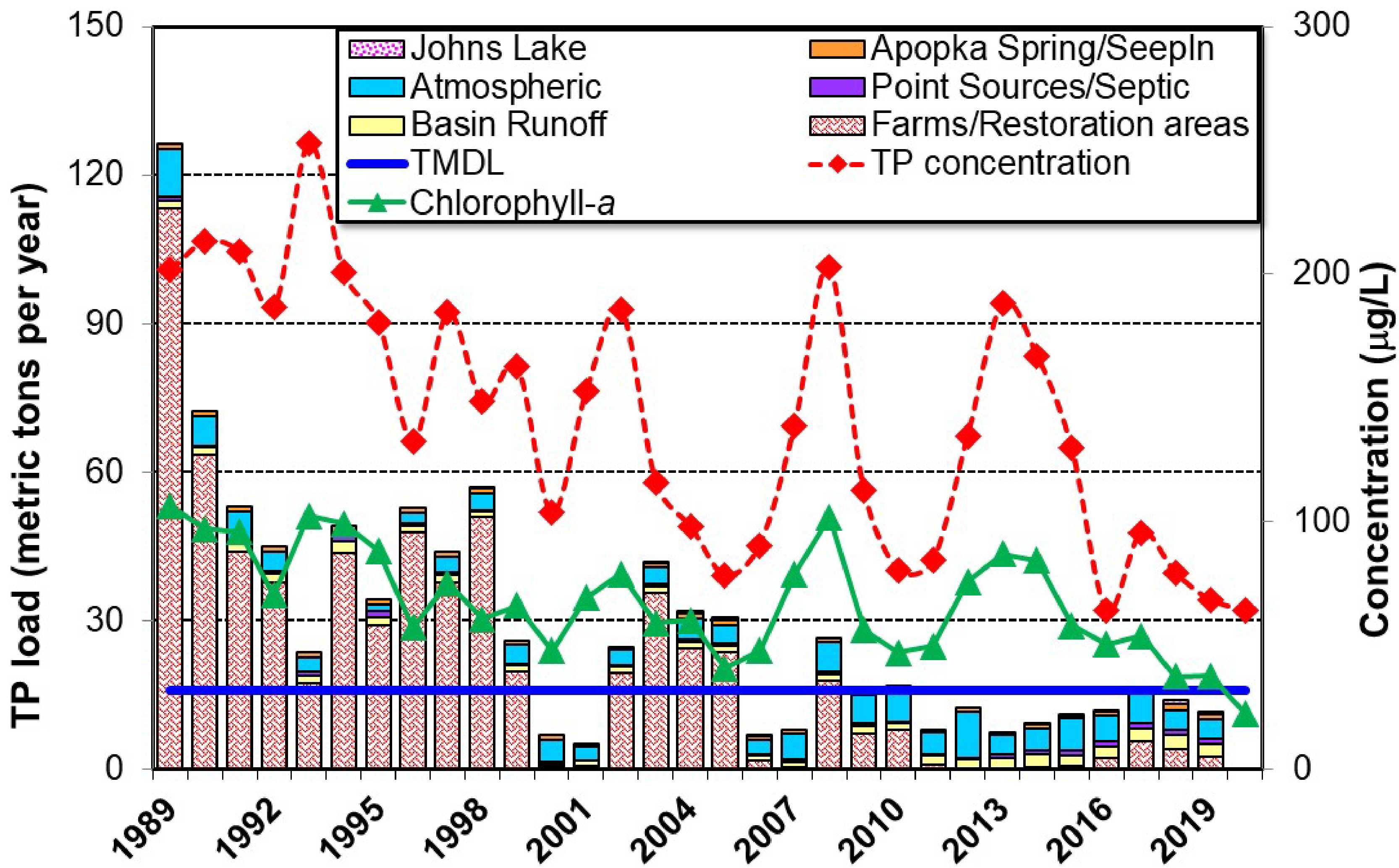


Figure 15. Lake Apopka estimated annual external total phosphorus loads (bars), TMDL loading target, and mean annual concentrations of TP and chlorophyll-*a* (lines).

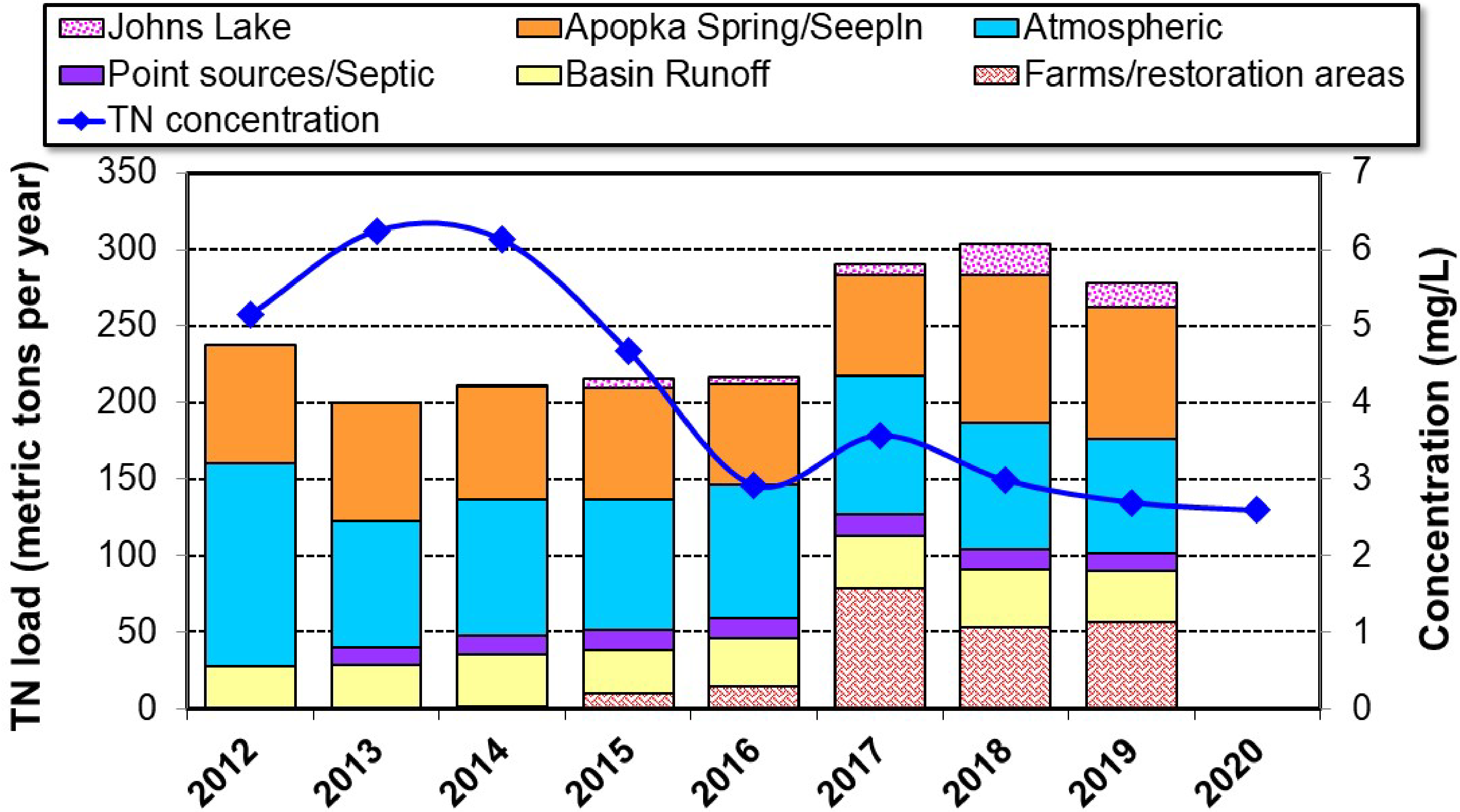


Figure 16. Lake Apopka estimated annual external total nitrogen loads (bars) and mean annual concentrations of TN.

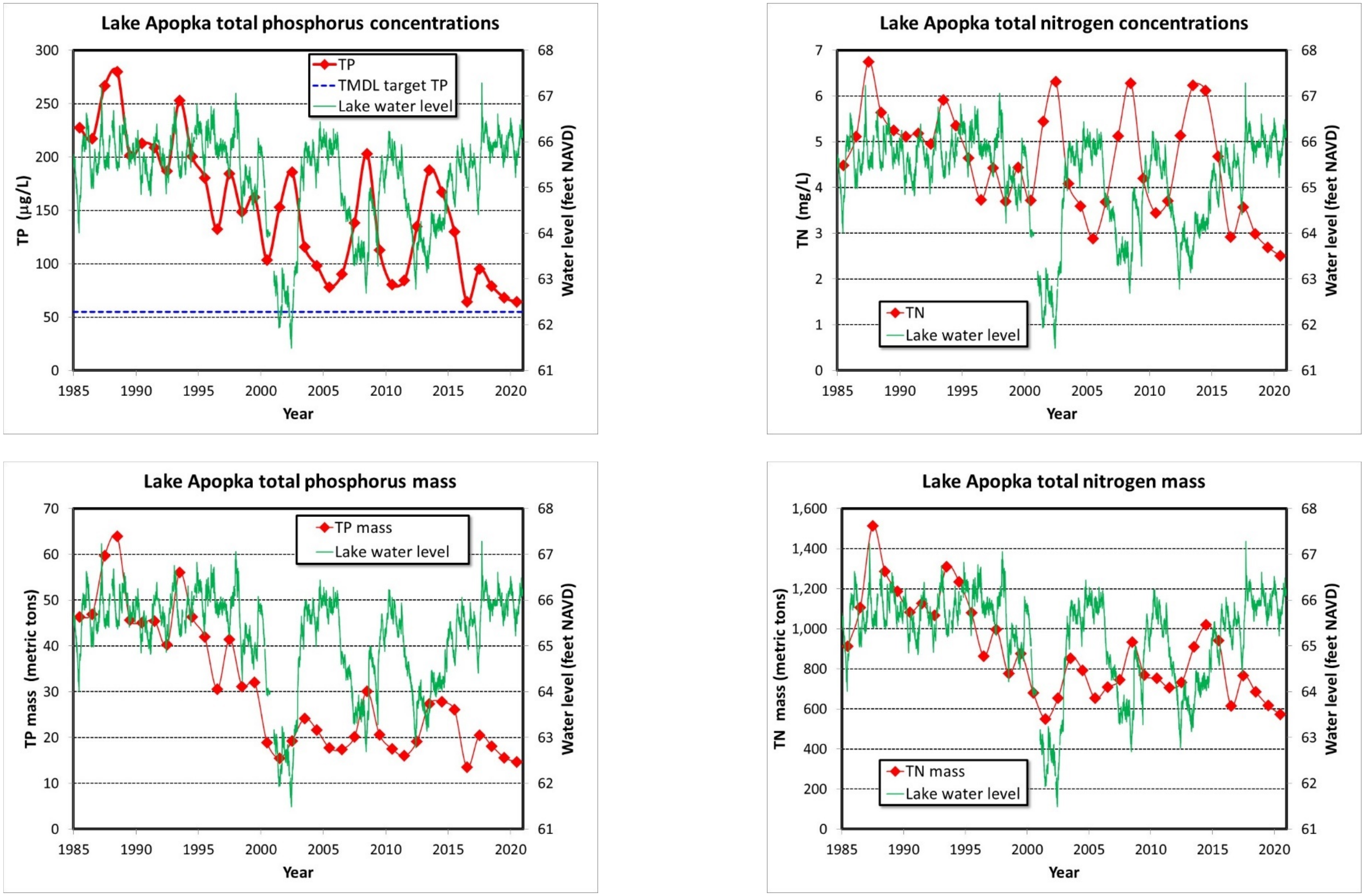


Figure 17. Lake Apopka mean annual total phosphorus and total nitrogen concentrations and masses, and water elevations.

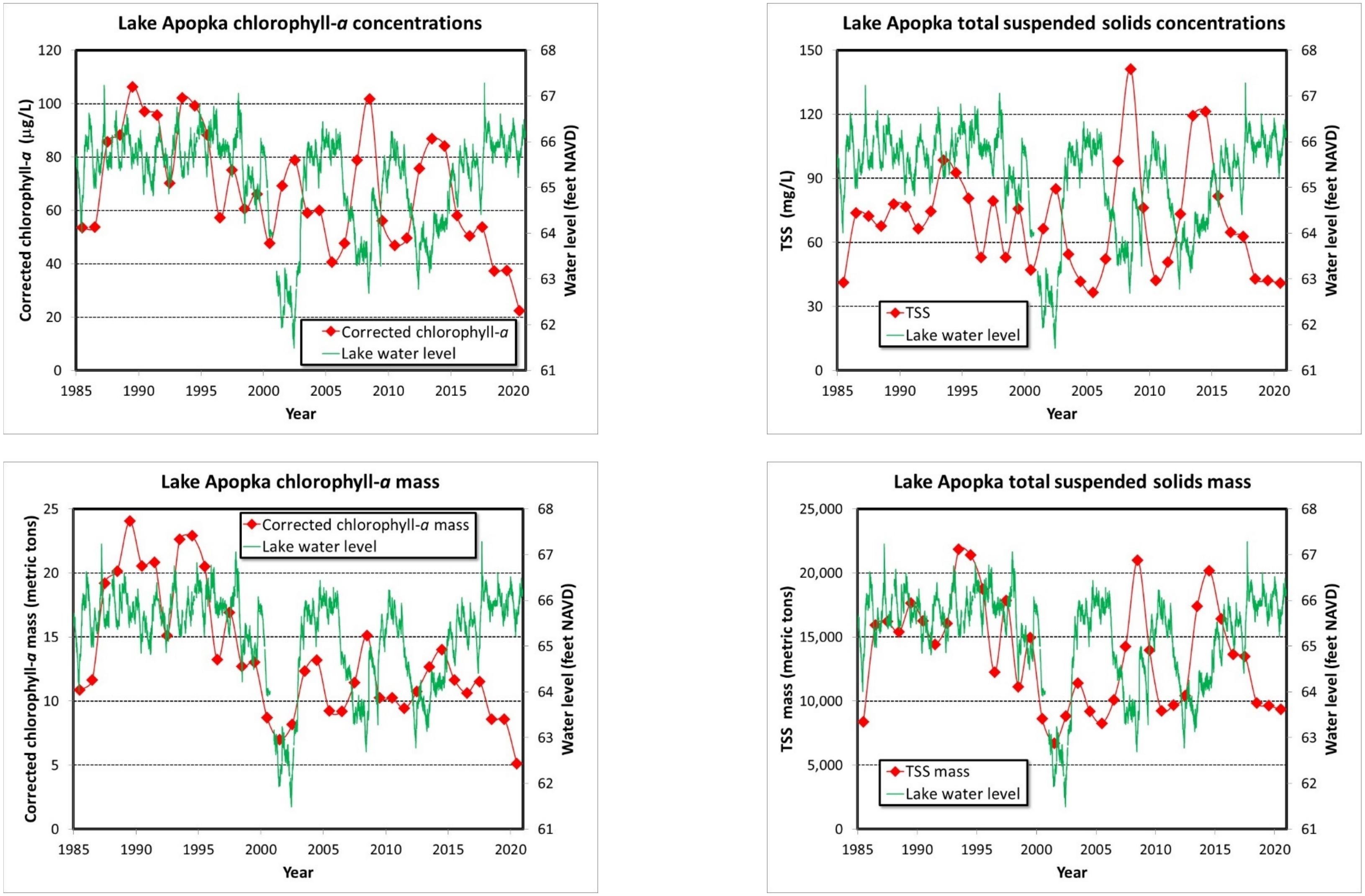


Figure 18. Lake Apopka mean annual chlorophyll-*a* and total suspended solids concentrations and masses, and water elevations.

Results of water quality trend analyses for Lake Apopka are summarized in Table 4, with more detailed results including Kendall's tau nonparametric correlation coefficient and statistical significance level of the nonparametric Theil-Sen trend lines in Appendix Table B-1. Time series plots are shown for several parameters in Figure 19 and Figure 20.

Table 4. Water quality trends for Lake Apopka.

Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality, $p<0.05$

(0) No trend, $p \geq 0.05$

(–) Significant decreasing trend, $p<0.05$

(+) Significant increasing trend, $p<0.05$

	1987 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, –	S, –	0	0
TN	S, 0	S, –	0	0
Chlorophyll-*a*	S, –	S, –	0	0
TSS	S, 0	S, 0	0	+
Secchi	S, +	S, +	S, 0	S, –
NH_4 [a]	0	0	–	0
NH_4 [a, b, c]	0	0		
NO_x [a, b]	–	–	–	0
PO_4 [a, b]	–	–	0	0
TP mass	–		0	
TN mass	–		0	
Chlorophyll-*a* mass	–		0	
TSS mass	S, 0		0	

[a] Dissolved fraction for both time periods

[b] Non-detect methods

[c] NH4-D not analyzed using non-detect methods for 2001-2019 because <5% below MDL

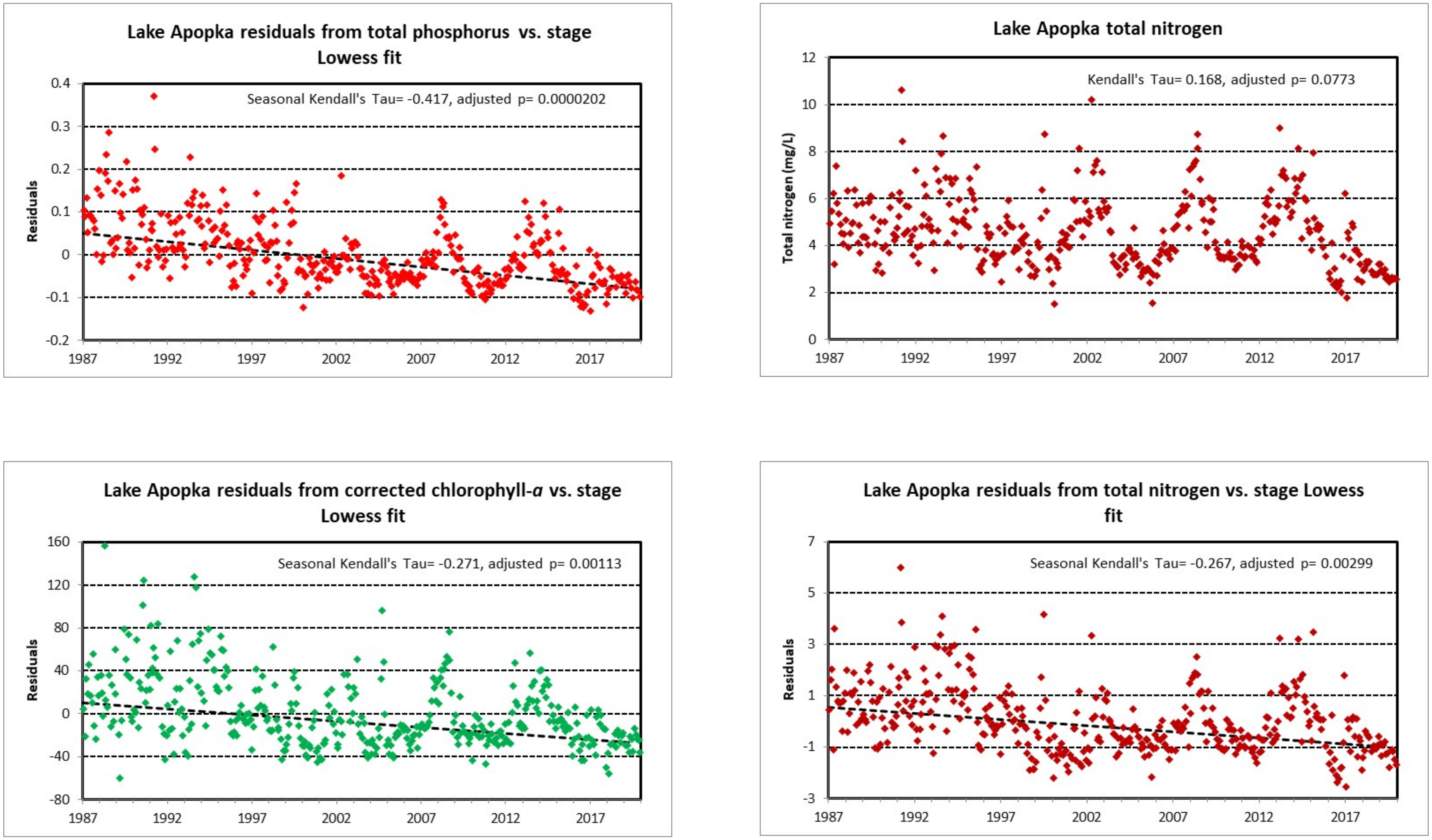


Figure 19. Water quality trends in Lake Apopka, 1987-2019.

Daily means and Theil-Sen trend lines.

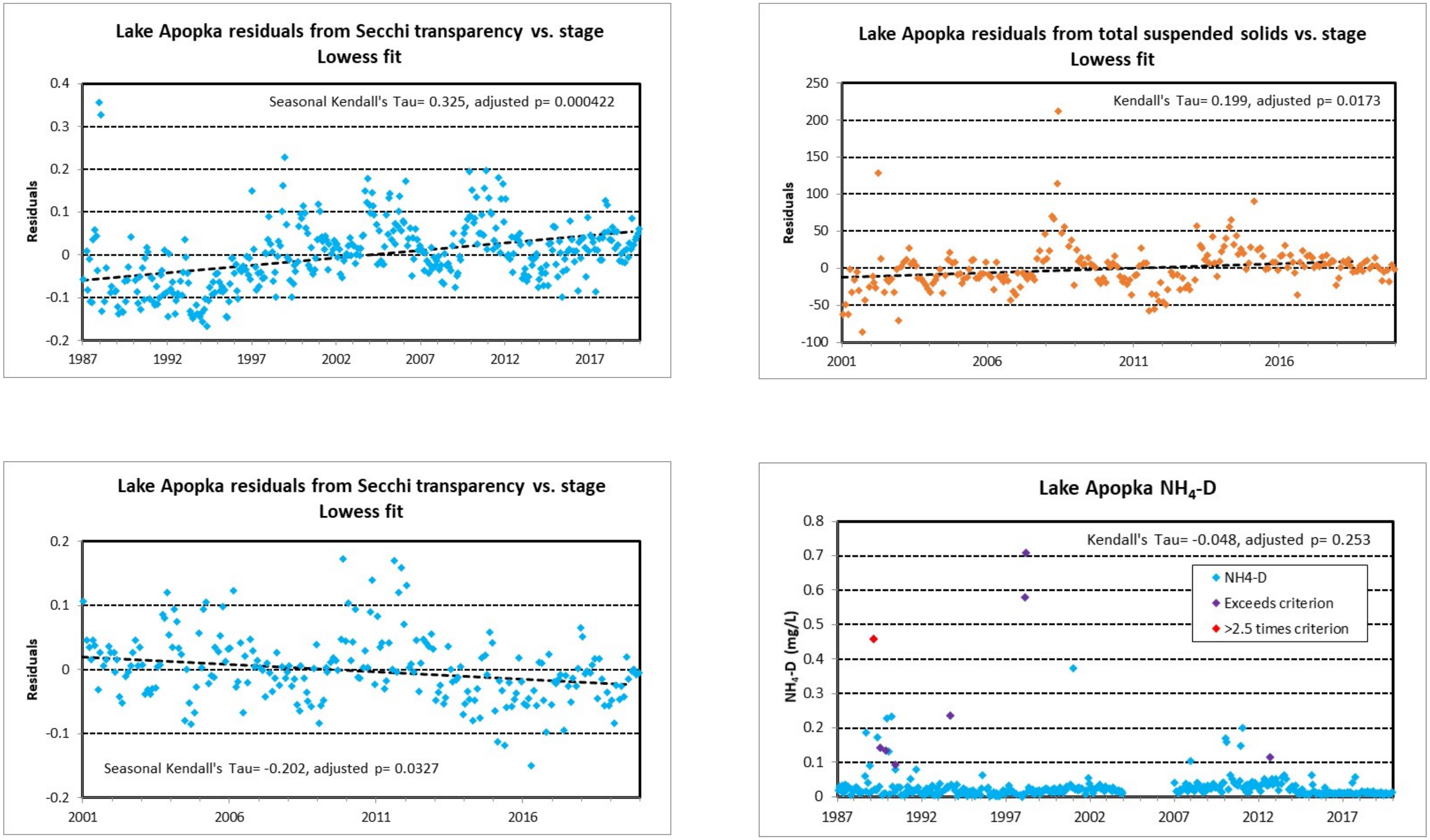


Figure 20. Water quality trends in Lake Apopka, 1987-2019 and 2001-2019.

Daily means and Theil-Sen trend lines.

Lake Apopka showed improving trends for several water quality variables over the entire period of record, including decreases in TP, chlorophyll-*a*, NO_x-D, and PO_4-D concentrations, decreases in TP, TN, and chlorophyll-*a* masses, and an increase in Secchi transparency. Over the period of record, there was no trend in TN concentrations, but a decreasing trend in residuals from the Lowess smoothed relationship, suggesting that water level fluctuations masked a decreasing trend, which is also supported by the decreasing trend in TN mass. However, for the 2001-2019 time period there were fewer significant trends, and some were indicators of deterioration, including an increase in TSS residuals and a decrease in Secchi residuals. Over the 2001-2019 time period there were significant decreasing trends for the unadjusted NH_4-D and NO_x-D measurements but no significant trend in residuals from the Lowess smoothed relationship between these parameters and lake water elevation (Table 4). This result may indicate that these apparent decreasing trends were associated with a change in water elevations over this period rather than representing an independent trend through time. There were several exceedances of the Florida ammonia standard, all but one prior to 2000 (Figure 20).

Some major differences from the previous trend assessment (Fulton 2018) were that analysis found significant increasing trends in unadjusted TSS concentration and in masses of chlorophyll-*a*, TN, and TSS for the 2001-2016 period, whereas this study found no significant trends in those parameters over the 2001-2019 period. This difference is probably because the 2001-2016 assessment period ended about two years after the last severe drought, whereas this study included three more years of relatively high water levels (Figure 18, Figure 18). A continued period of relatively high water levels may eventually result in decreasing trends in chlorophyll-*a*, TN, and TP even for the post-2001 period. However, the deteriorating trends in the residuals from the Lowess smoothed relationships of lake water elevations with TSS and Secchi depth suggest that improvements in these parameters may require further intervention. Several projects have been in progress or proposed to try to reduce resuspended sediment concentrations, including the Lake Apopka marsh flow-way, rough fish harvesting, aquatic vegetation plantings, and targeted dredging of easily resuspended sediments (more details in the Discussion section).

Lake Apopka phytoplankton have been dominated by cyanobacteria throughout the period of record, other than a few samples that showed high biovolume of diatoms (Figure 21). These diatom peaks are probably due to resuspension of meroplanktonic populations (Carrick et al. 1993). Highest reported biovolumes occurred in the 1993-1994 (analyzed by Burkholder), in 2003 (diatom peak), 2007 (both drought periods, analyzed by Phycotech), and in 2019 (analyzed by BSA). The high biovolumes in summer 2019 are not consistent with the chlorophyll-*a* concentrations.

Phytoplankton composition was most commonly dominated by the cyanobacterial genus *Planktolyngbya* (Figure 22). Unlike other lakes in the basin, the cyanobacterial genera *Cylindrospermopsis* and *Pseudanabaena* were not prominent in Lake Apopka. There are some apparent taxonomic differences among analysts, particularly the prominence of the cyanobacterial genus *Raphidiopsis* in BSA analyses and its near absence in analyses by Burkholder and Phycotech. Recent genetic studies indicate *Raphidiopsis* and *Cylindrospermopsis* should be considered the same genus (Aguilera et al. 2018). The cyanobacterial genus *Aphanocapsa* was more prominent in analyses by Phycotech, while the cyanobacterial genus *Microcystis* tended to be more prominent in analyses by Burkholder and BSA. Both *Aphanocapsa* and *Microcystis* are small coccoid cells, sometimes mistaken for each other (http://www.cfb.unh.edu/phycokey/Choices/Cyanobacteria/cyano_colonies/APHANOCAPSA/Aphanocapsa_key.html).

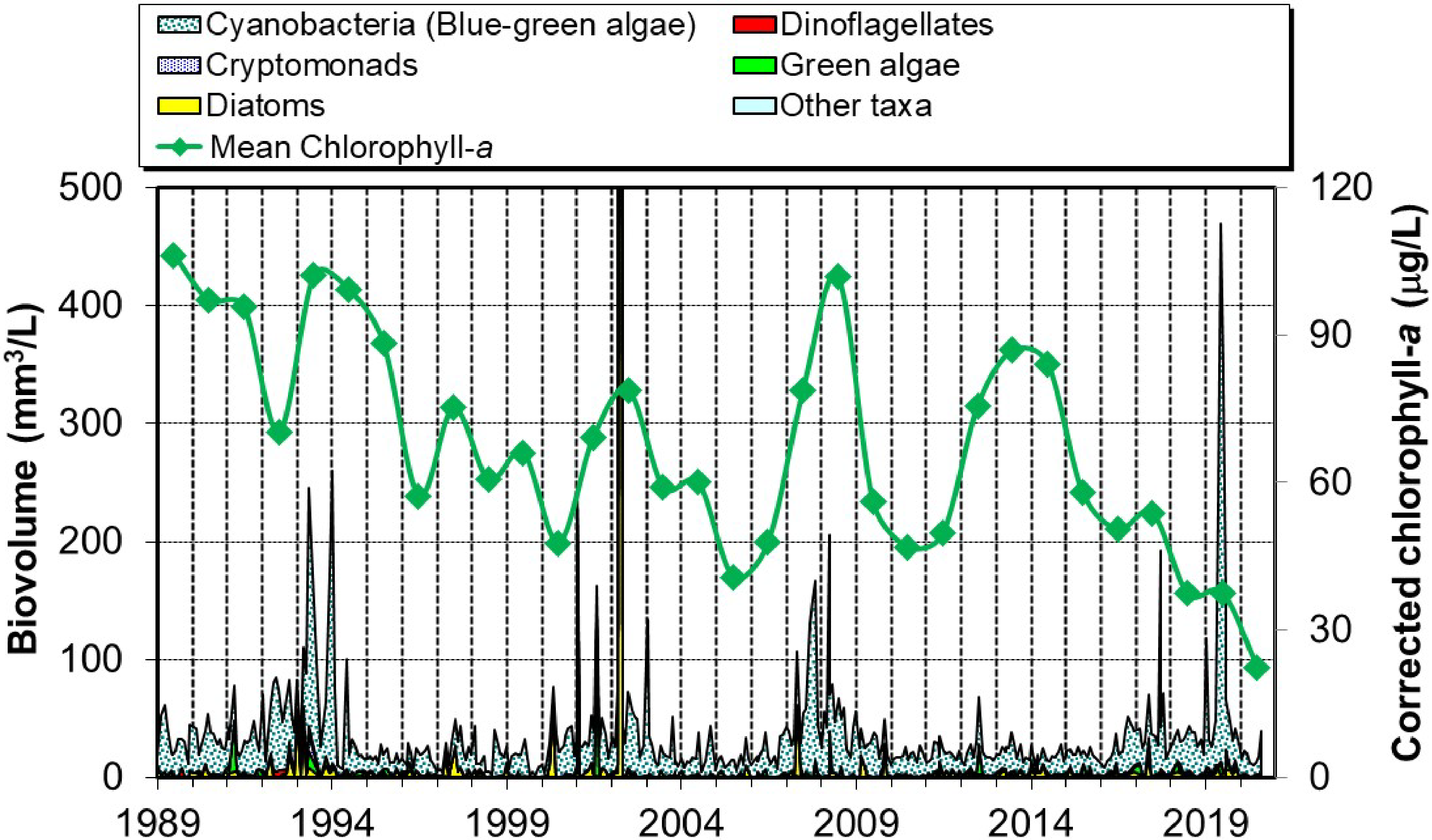


Figure 21. Lake Apopka phytoplankton biovolumes and chlorophyll-*a* concentrations.

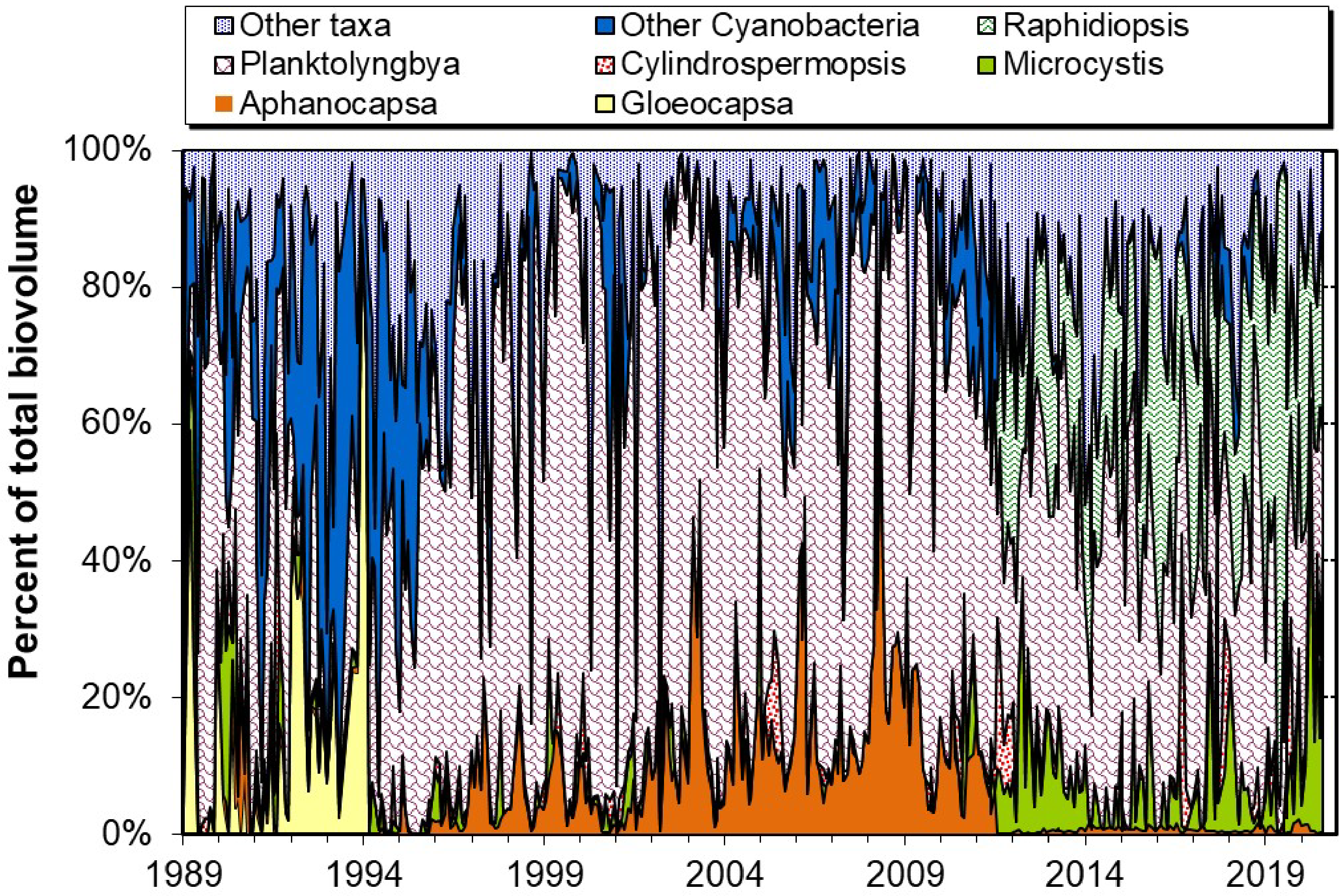


Figure 22. Lake Apopka phytoplankton percent composition.

Lake Beauclair

For most of the period of record, estimated external TP loading to Lake Beauclair has been heavily dominated by discharges through the Apopka-Beauclair Canal, reaching 40 metric tons (MT) in 1998 (Figure 23, Appendix Table D- 3). For the 11 years from 2006-2016, the external TP load to the lake was below the adopted TMDL for the lake (3.2 MT/year), averaging about 1.6 MT/year, due a combination of low discharge volumes from Lake Apopka, low TP concentrations in those discharges (Figure 2), and TP removals by the Lake Apopka marsh flow-way and the NuRF project. However, for the last 3 years external loading has exceeded the TMDL, due to higher tributary flows related to Hurricane Irma in 2017 and continued high rainfall in subsequent years. Nevertheless, TP discharges have been substantially lower than in past high-flow years, as the upstream water quality improvement projects have continued. I think the TMDL should be thought of as a long-term goal, not a target that should be met every year. It likely is not possible to meet the Lake Beauclair TMDL in years with high tributary flows.

The major external TP sources to Lake Beauclair for the last 5 years have been tributary discharges (79%), discharges from the Hurley muck farm (6%), atmospheric deposition (5%), runoff from natural areas (3%), septic tank effluents (2%), and urban-residential runoff (2%).

External TN loading to Lake Beauclair has also generally been heavily dominated by discharges through the Apopka-Beauclair Canal (Figure 24, Appendix Table D- 4). The external TN load to the lake has also increased in the last 3 years after having been low for the previous 11 years. The major TN sources to Lake Beauclair for the last 5 years have been tributary discharges (83%), discharges from the Hurley muck farm (8%), runoff from natural areas (3%), atmospheric deposition (2%), and septic tank effluents (1%).

A recent report (Amec 2017) has documented high nutrient concentrations in ditches draining land east of the Hurley farm and developed a model predicting substantially higher nutrient discharges from that part of the watershed than estimated by our modeling. Their runoff estimates are not high enough to significantly affect the total load estimates in the years with high tributary loading but could more substantially affect the load estimates in low-flow years.

TP and chlorophyll-*a* concentrations have substantially decreased in Lake Beauclair. TP and chlorophyll-*a* concentrations in the lake decreased beginning in 2000, appear to have reached a plateau in 2004-2008 (chlorophyll somewhat increasing), and then decreased further after the NuRF started (Figure 23). A minimum in concentrations occurred in 2010, which was the first year in which the NuRF operated for almost all the year. After 2010 there were only intermittent discharges from Lake Apopka or operation of the NuRF through mid-2017. After 2010 concentrations increased through 2013 before generally decreasing over the last several years to the lowest concentrations in the period of record. In the last 5 years, average concentrations have been 48 μg TP/L and 35 μg chlorophyll-*a*/L. These concentrations still exceed the TMDL target TP concentration of 32 μg/L, and the expected chlorophyll-*a* concentration of 30 μg/L at the TP target. TN concentrations have also decreased since the early 2000s, also reaching a minimum in 2010, increasing for the next 4 years, then generally decreasing over the last several years (Figure 24). In the last 5 years, average concentrations have been 1.8 mg TN/L.

Unlike in Lake Apopka, concentrations and masses of TP show rather similar trends (Figure 25), indicating little evidence for concentration in a smaller water volume during drought periods. There is some indication of drought-related concentration of TN, chlorophyll-*a*, and TSS in Lake Beauclair (Figure 25, Figure 26). Water quality tended to deteriorate during these drought periods, although not to the extent seen in Lake Apopka.

Results of water quality trend analyses for Lake Beauclair are summarized in Table 5, with more detailed results including Kendall's tau nonparametric correlation coefficient and statistical significance level of the nonparametric Theil-Sen trend lines in Appendix Table B-2. Time series plots are shown for several parameters in Figure 27 and Figure 28.

Table 5. Water quality trends for Lake Beauclair.

Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality, $p<0.05$

(0) No trend, $p \geq 0.05$

(–) Significant decreasing trend, $p<0.05$

(+) Significant increasing trend, $p<0.05$

	1989 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	–	S, –	S, –	S, –
TN	–	–	–	–
Chlorophyll-*a*	–	S, –	–	–
TSS	–	–	–	–
Secchi	+	+	+	+
NH_4 [a]	0	S, 0	–	S, 0
NH_4 [a, b]	0	S, –	–	S, –
NO_x [a, b]	0	0	S, 0	–
PO_4 [a, b]	–	0	–	–
TP mass	S, –		S, –	
TN mass	–		–	
Chlorophyll-*a* mass	–		–	
TSS mass	–		–	

[a] Dissolved fraction for both time periods

[b] Non-detect methods

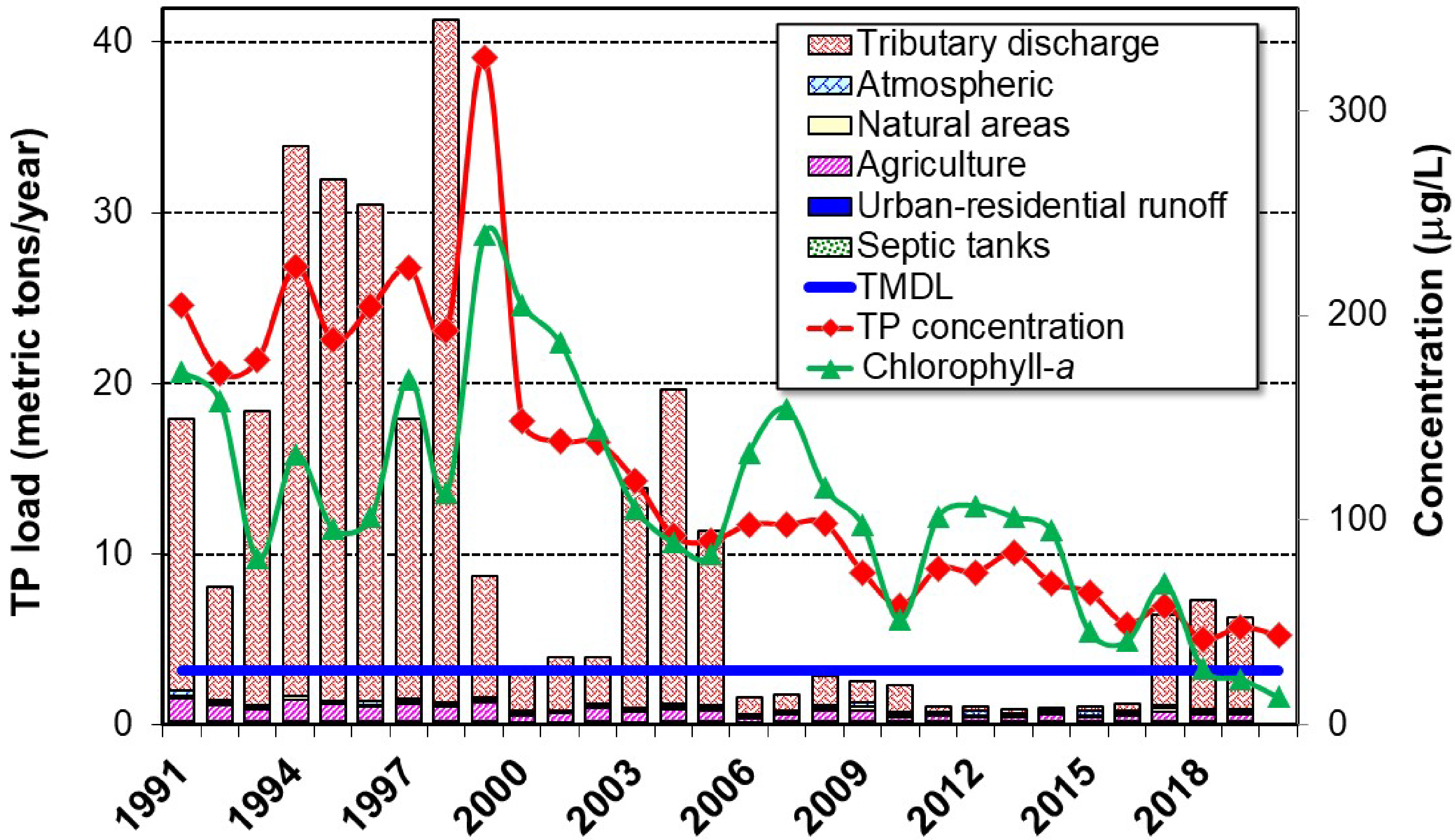


Figure 23. Lake Beauclair estimated annual external total phosphorus loads (bars), TMDL loading target, and mean annual concentrations of TP and chlorophyll-*a* (lines).

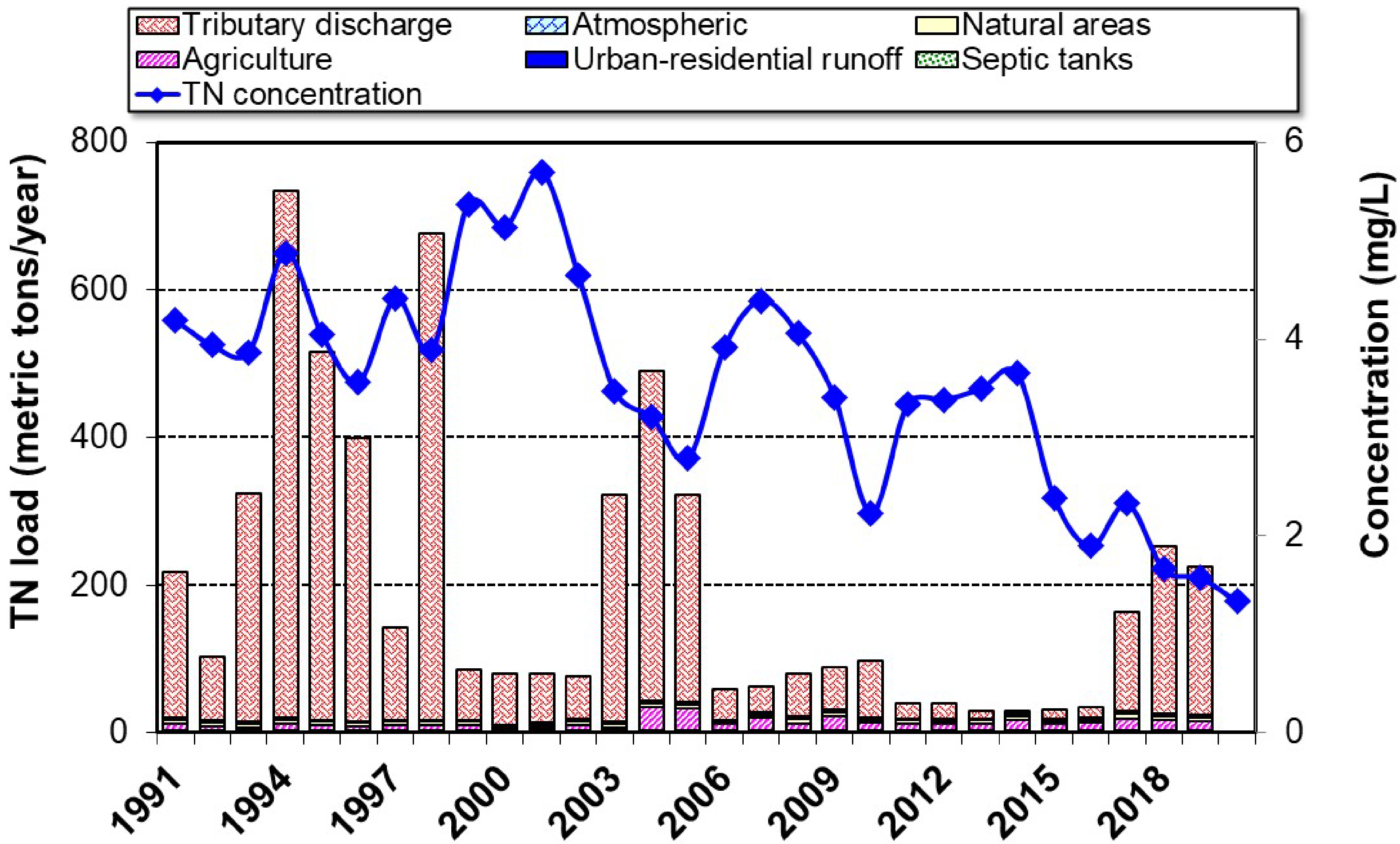


Figure 24. Lake Beauclair estimated annual external total nitrogen loads (bars), and mean annual concentrations of TN.

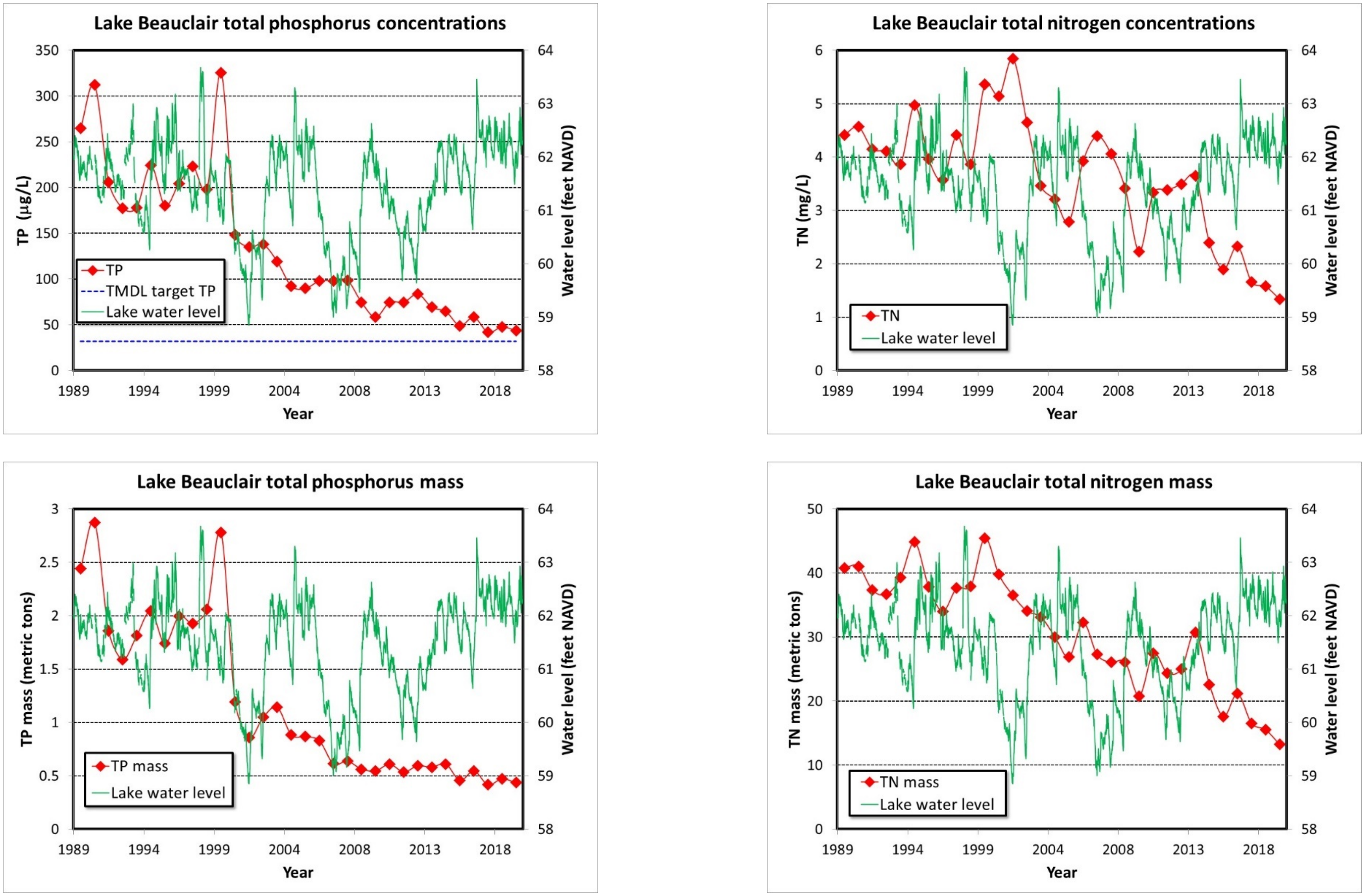


Figure 25. Lake Beauclair mean annual total phosphorus and total nitrogen concentrations and masses, and water elevations.

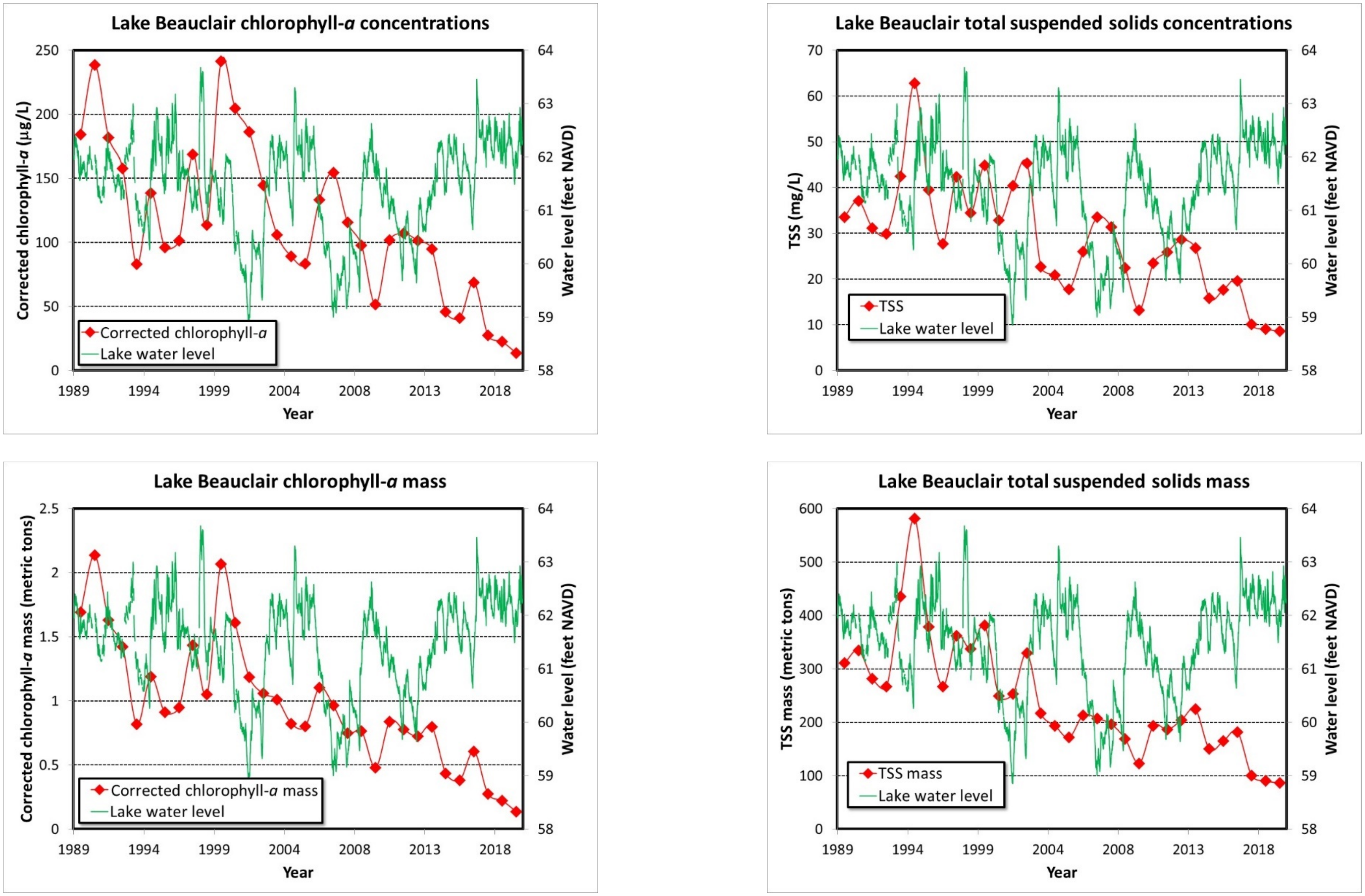


Figure 26. Lake Beauclair, mean annual chlorophyll-*a* and total suspended solids concentrations and masses, and water elevations.

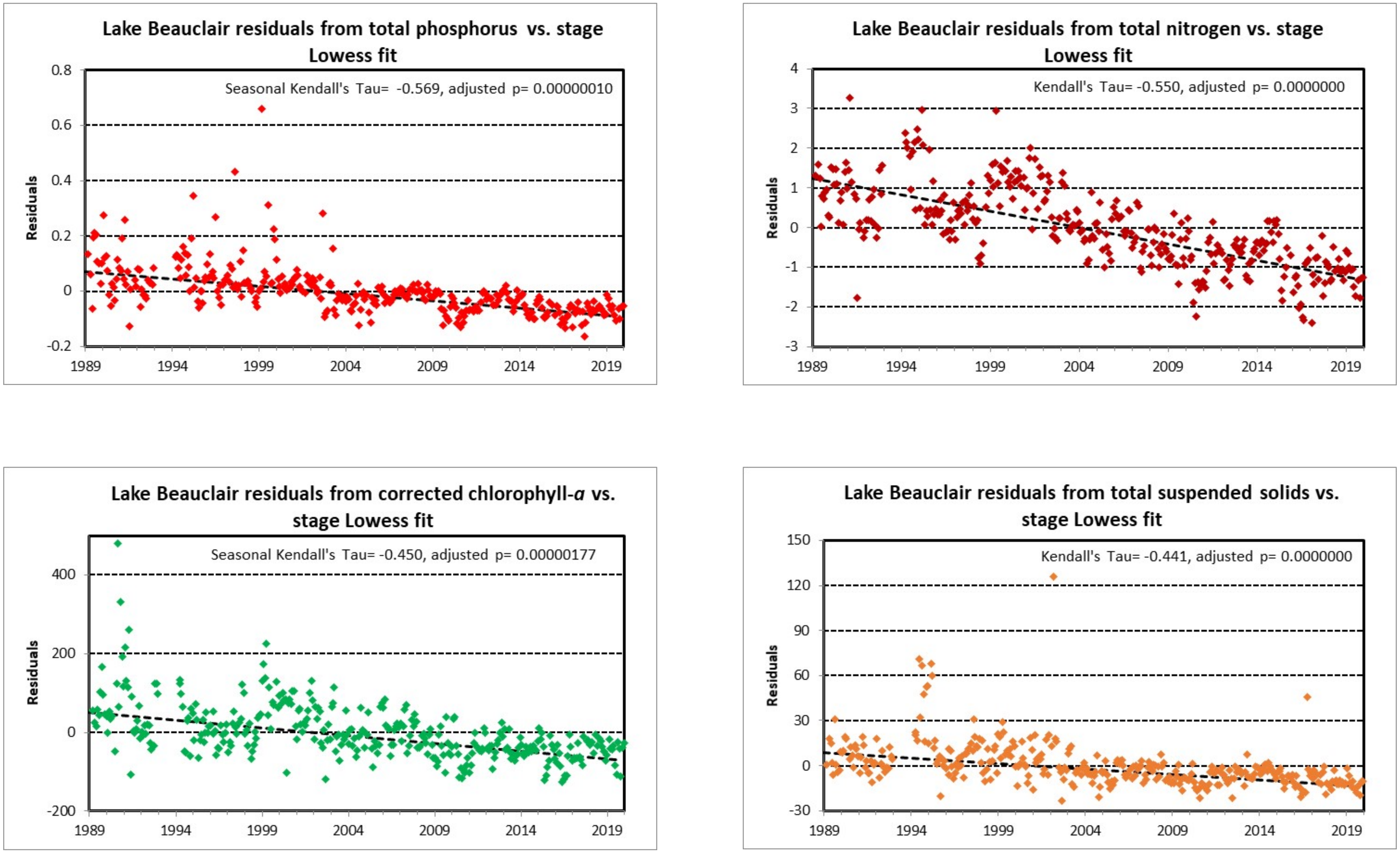
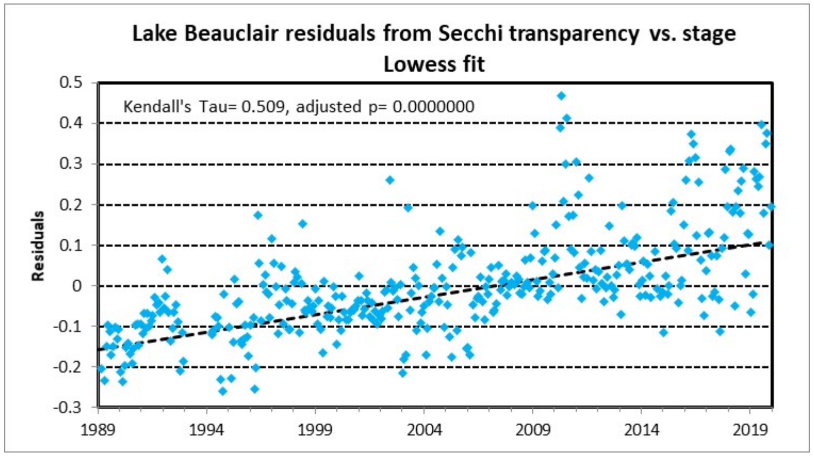
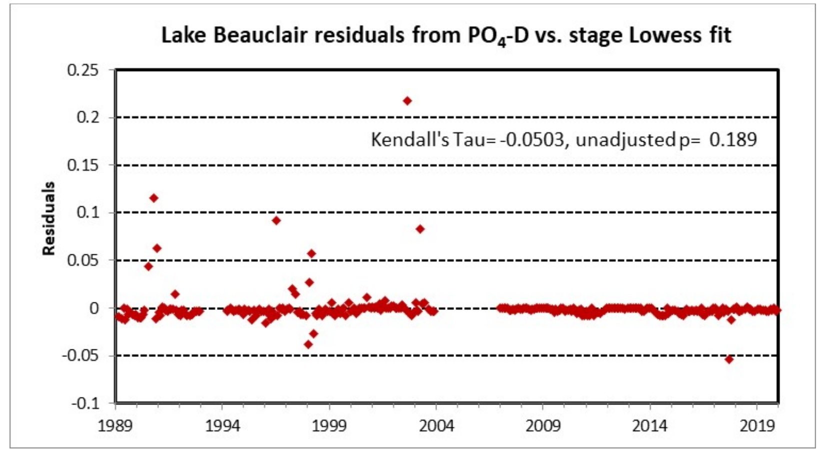
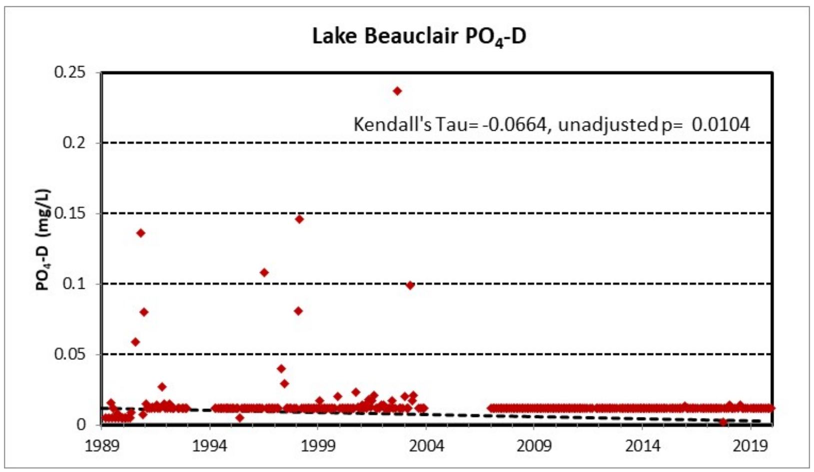


Figure 27. Water quality trends in Lake Beauclair, 1989-2019.

Daily means and Theil-Sen trend lines.







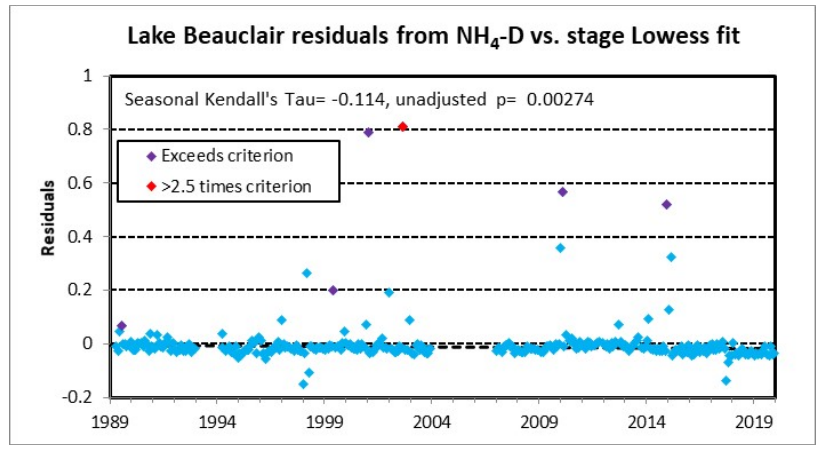


Figure 28. Water quality trends in Lake Beauclair, 1989-2019.

Daily means and Theil-Sen or ATS trend lines.

Lake Beauclair showed improving trends for several water quality variables. For both time periods, there were decreasing trends for both concentrations and masses of TP, TN, chlorophyll-*a*, and TSS. Secchi transparency increased in both time periods. PO_4-D also showed a decreasing trend over the both time periods. Over the entire period, there was a significant decreasing trend for the unadjusted PO_4-D measurements but no significant trend in residuals from the Lowess smoothed relationship between PO_4-D and lake water elevation (Table 5). This result may indicate that the apparent decreasing trend in PO_4-D was associated with changes in water elevations over this period rather than representing an independent trend through time. When analyzed using non-detect methods, NH_4-D showed decreasing trends in the water level-adjusted residuals in both time periods, but not when analyzed with standard methods. However, the Kendall's tau values were actually lower when analyzed using non-detect methods (Table 5), so the significance of the non-detect trends may be due to the absence of adjustment for serial correlation in those analyses. Significant trends for PO_4-D and NO_x-D despite relatively low Kendall's tau values may also reflect the lack of adjustment for serial correlation. There were several exceedances of the Florida ammonia standard (Figure 28).

Overall, the results of the trend analyses for Lake Beauclair were very similar to the previous trend assessment (Fulton 2018), reflecting a continuation of the improving water quality apparent at that time. This is despite relatively high nutrient loading for the last three years. However, that high nutrient loading was primarily due to a large volume of discharge from Lake Apopka, and the concentration of TP in those discharges has been relatively low (Figure 2). Treatment of some of the flows by the Lake Apopka Marsh Flow-way and the NuRF further reduced the TP concentrations entering Lake Beauclair.

The phytoplankton monitoring in Lake Beauclair ended in August 2018. Lake Beauclair phytoplankton have been dominated by cyanobacteria throughout the period of record, other than a few samples that showed high biovolume of diatoms (Figure 29). Highest reported biovolumes occurred in the 1989-1993 period (analyzed by Burkholder) and in 2000-2001 (analyzed by Phycotech). The decrease in phytoplankton biovolumes was more subtle than that of chlorophyll-*a*, but for both there were low levels in 2010 (Phycotech) and 2016 (BSA).

Phytoplankton composition was dominated by *Cylindrospermopsis* in the periods analyzed by Burkholder and BSA (Figure 30). Burkholder reported *Anabaenopsis* as the dominant, but reanalysis of two samples from that period by BSA and GreenWater Laboratories identified the dominant as *Cylindrospermopsis*, and I assumed that is the correct identification. However, Phycotech identified *Pseudanabaena limnetica* (alternate name *Oscillatoria limnetica*) as the dominant phytoplankton taxon in Lake Beauclair. A possible explanation for this discrepancy comes from a study of Lake Griffin by Phlips and Schelske (2004). They reported both *Cylindrospermopsis* and *Oscillatoria* in Lake Griffin, noting they were very similar in appearance, with the main difference being the presence of heterocysts in *Cylindrospermopsis* and their absence in *Oscillatoria*. Phlips noted, "In the absence of more specific genetic or biochemical markers it is not possible to completely exclude the possibility that the latter form of *Oscillatoria* is an ecomorphotype of *Cylindrospermopsis*". Therefore, it is possible that Phycotech identified filaments lacking heterocysts as *Pseudanabaena/Oscillatoria*, while BSA identified them as *Cylindrospermopsis* (only a fraction of the *Cylindrospermopsis* identified by BSA had heterocysts). This is an important difference in taxonomy, since *Cylindrospermopsis* can produce cyanotoxins, while *Pseudanabaena/Oscillatoria* is not known to be toxic. In a limited number of samples from the UORB lakes, GreenWater Laboratories usually identified *Cylindrospermopsis* and *Pseudanabaena* as co-dominant (unpublished data). *Cylindrospermopsis* has been genetically identified from the UORB lakes (Dyble et al. 2002), but that does not exclude the possibility that *Pseudanabaena* is also present. Another possible taxonomic difference between analysts has been consistently higher biovolumes of *Microcystis* and *Planktolyngbya* since BSA began analyses (Figure 30).

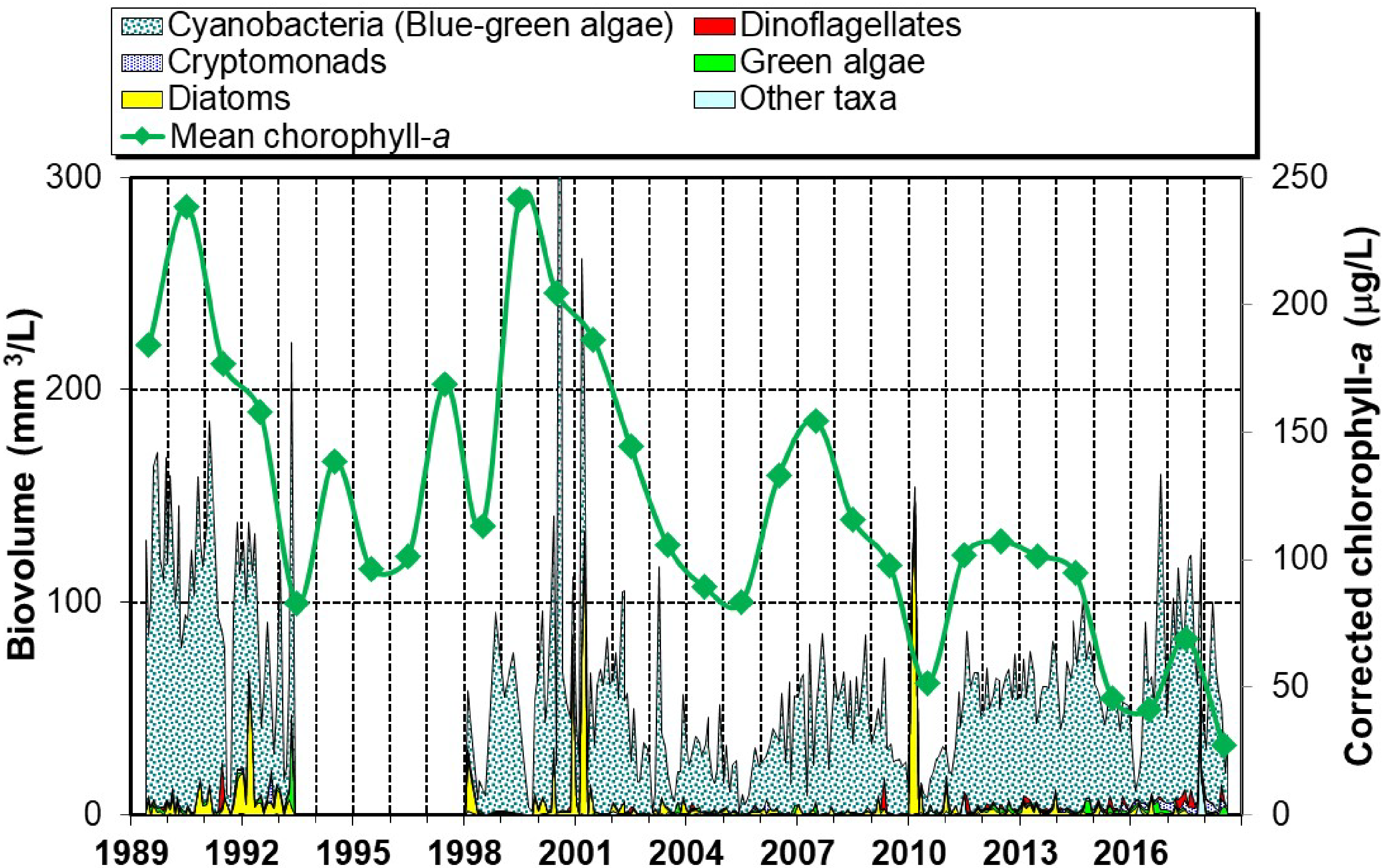


Figure 29. Lake Beauclair phytoplankton biovolumes and chlorophyll-*a* concentrations.

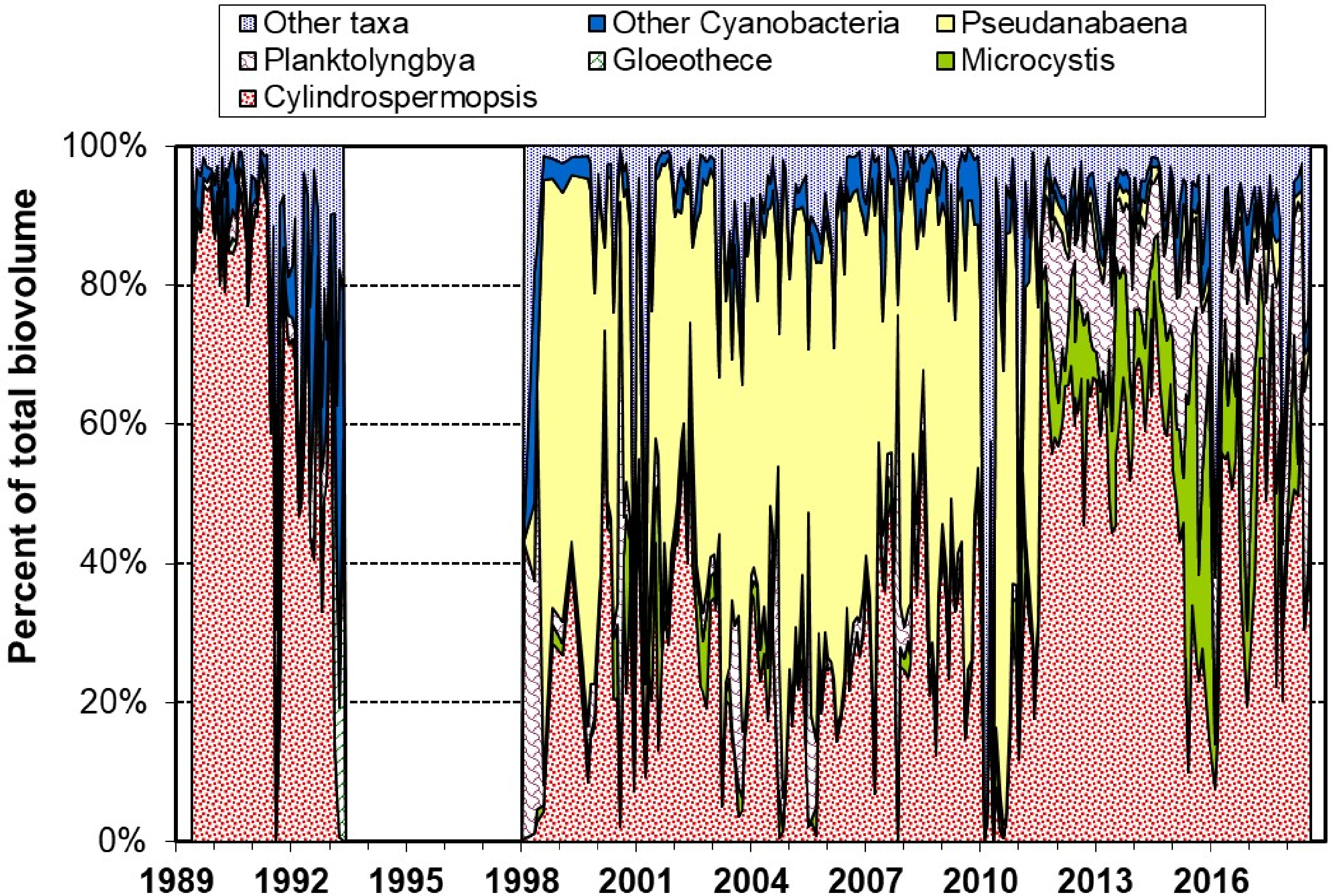


Figure 30. Lake Beauclair phytoplankton percent composition.

LAKE DORA

For most of the period of record, estimated external TP loading to Lake Dora has been heavily dominated by tributary discharges, primarily from Lake Beauclair, reaching 46 metric tons (MT) in 1998 (Figure 31, Appendix Table D- 5). Estimated external TP loads to Lake Dora have declined substantially due to a combination of decreases in discharge volume and concentrations from upstream Lake Beauclair. As with Lake Beauclair, for the 11 years from 2006-2016, the external TP load to Lake Dora was below the adopted TMDL for the lake (6 MT/year), averaging about 3.0 MT/year, but has slightly exceeded the TMDL the last 3 years, due primarily to higher tributary discharges from Lake Beauclair.

The major external TP sources to Lake Dora for the last 5 years have been tributary discharges (57%), urban-residential runoff (19%), atmospheric deposition (14%), and septic tank effluents (6%). Tributary discharges were divided into 56% from Lake Beauclair and 0.4% backflows from Lake Eustis.

External TN loading to Lake Dora has also generally been heavily dominated by discharges from Lake Beauclair (Figure 32, Appendix Table D- 6). The external TN load to the lake was also low for the 11 years from 2006-2016, then increased for the last 3 years. The major TN sources to Lake Dora for the last 5 years have been tributary discharges (76%; 75% from Lake Beauclair, 1% from Lake Eustis), atmospheric deposition (9%), urban-residential runoff (6%), septic tank effluents (4%), and runoff from natural areas (4%).

TP and chlorophyll-*a* concentrations have substantially decreased in Lake Dora, showing similar trends to Lake Beauclair (Figure 31). TP concentrations in the lake decreased substantially from 1999 to 2000, with smaller decreases in subsequent years. Chlorophyll-*a* decreased from 1999-2002, with another decrease in 2010. As with upstream Lake Beauclair, low concentrations of both TP and chlorophyll-*a* occurred in 2010, followed by increased concentrations through 2014 and a general decrease in subsequent years (other than an increase in chlorophyll-*a* in 2017) to the lowest in the period of record. In the last 5 years, average concentrations have been 36 μg TP/L and 32 μg chlorophyll-*a*/L. These concentrations slightly exceed the TMDL target TP concentration of 31 μg/L, and the expected chlorophyll-*a* concentration of 29 μg/L at the TP target. TN concentrations have also decreased since the early 2000s, also reaching the lowest concentrations in the last few years (Figure 32). In the last 5 years, average concentrations have been 1.6 mg TN/L. Concentrations and masses of TP show rather similar trends, although there is some evidence for drought-induced increased concentrations of TN, chlorophyll-*a*, and TSS (Figure 33, Figure 34). Water quality tended to deteriorate during the 2000-2002 and 2007-2008 droughts.

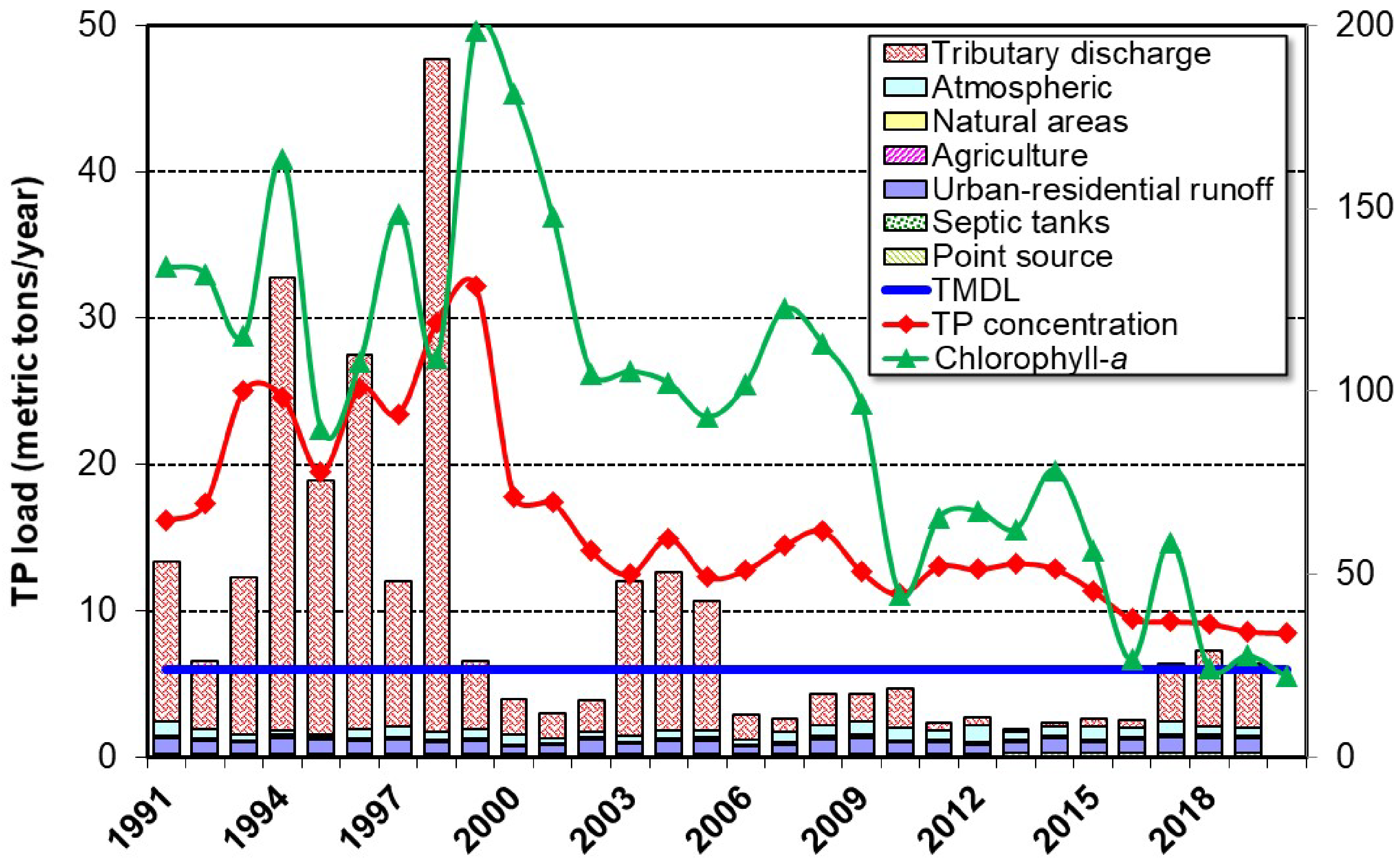


Figure 31. Lake Dora estimated annual external total phosphorus loads (bars), TMDL loading target, and mean annual concentrations of TP and chlorophyll-*a* (lines).

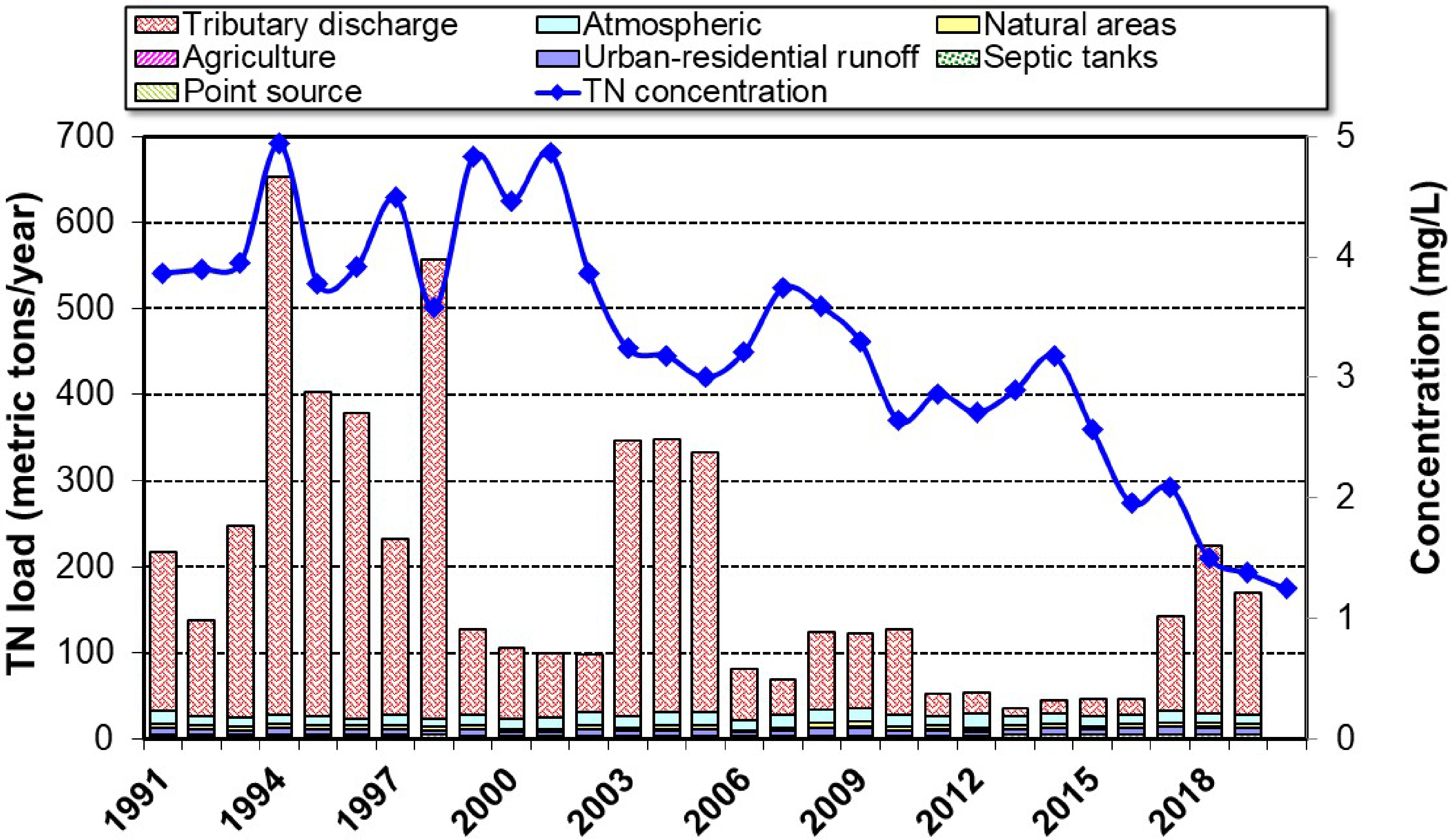


Figure 32. Lake Dora estimated annual external total nitrogen loads (bars), and mean annual concentrations of TN.

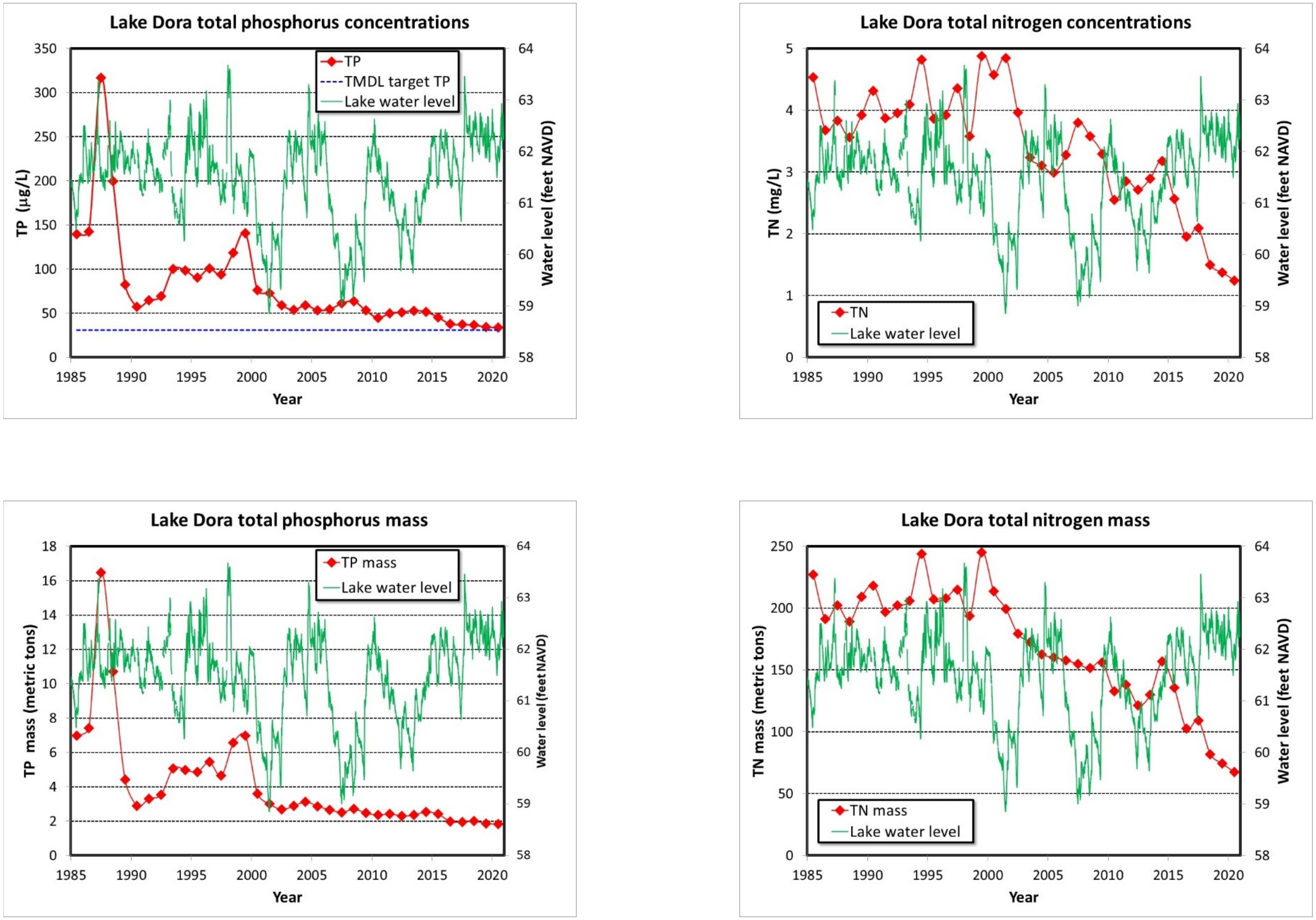


Figure 33. Lake Dora mean annual total phosphorus and total nitrogen concentrations and masses, and water elevations.

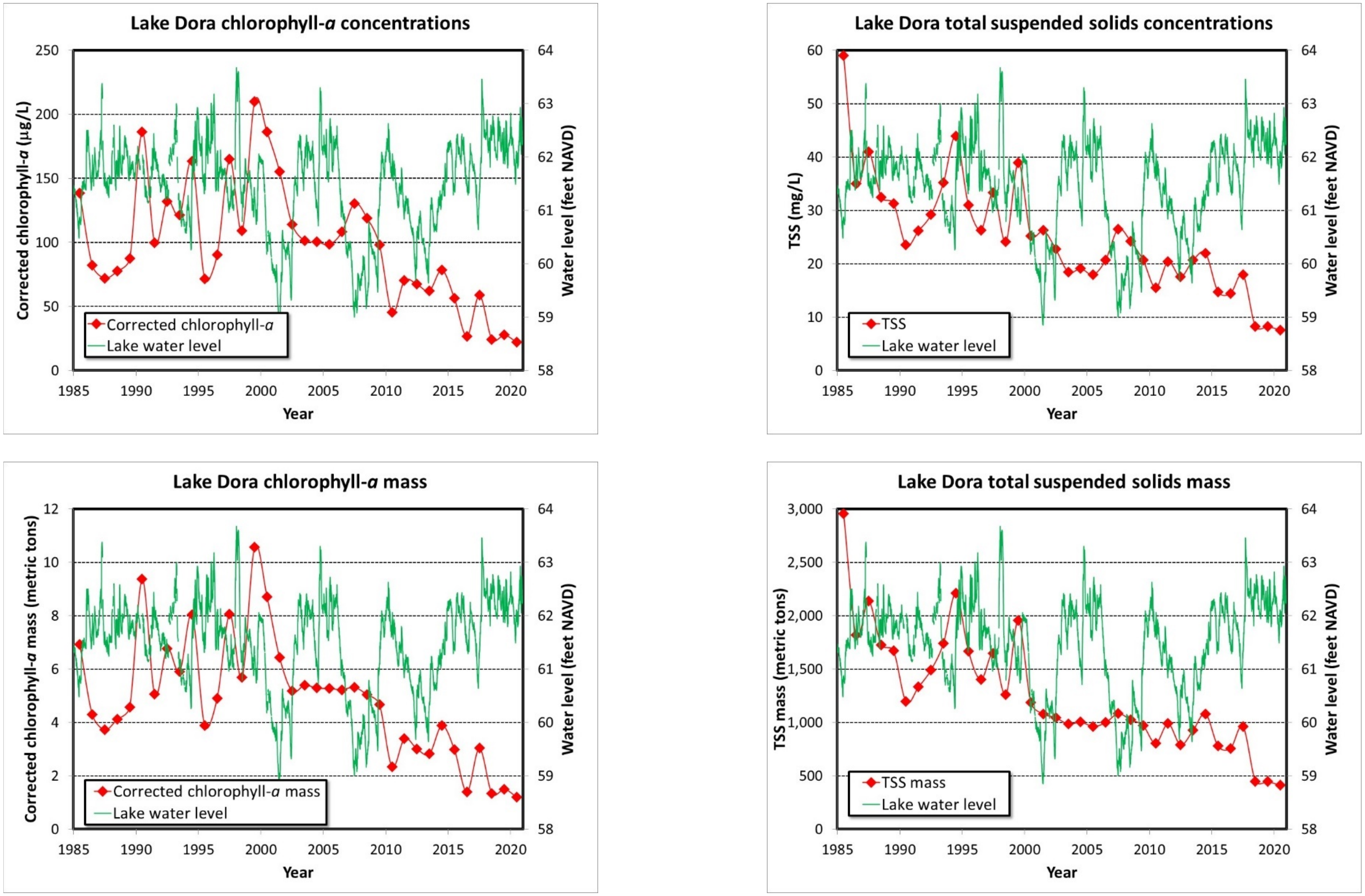


Figure 34. Lake Dora, mean annual chlorophyll-*a* and total suspended solids concentrations and masses, and water elevations.

Results of water quality trend analyses for Lake Dora are summarized in Table 6, with more detailed results including Kendall's tau nonparametric correlation coefficient and statistical significance level of the nonparametric Theil-Sen trend lines in Appendix Table B- 3. Time series plots are shown for several parameters in Figure 35 and Figure 36.

Lake Dora showed improving trends for several water quality variables. For both time periods, there were decreasing trends for both concentrations and masses of TP, TN, chlorophyll-*a*, and TSS (Table 6). Secchi transparency increased in both time periods. Over the 2001-2019 period,

Table 6. Water quality trends for Lake Dora.

Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality, $p<0.05$

(0) No trend, $p\geq0.05$

(–) Significant decreasing trend, $p<0.05$

(+) Significant increasing trend, $p<0.05$

	1986 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, –	S, –	S, –	S, –
TN	–	–	–	–
Chlorophyll-*a*	–	–	–	–
TSS	–	–	–	S, –
Secchi	+	+	+	+
NH_4 [a]	S, 0	0	0	0
NH_4 [a, b]	S, 0	–	0	0
NO_x [a, b]	0	0	0	0
PO_4 [b]	ND [c]	ND	–	0
TP mass	S, –		S, –	
TN mass	–		–	
Chlorophyll-*a* mass	–		–	
TSS mass	–		–	

[a] Synthetic total fraction for 1986-2019; dissolved fraction for 2001-2019

[b] Non-detect methods

[c] ND – No data

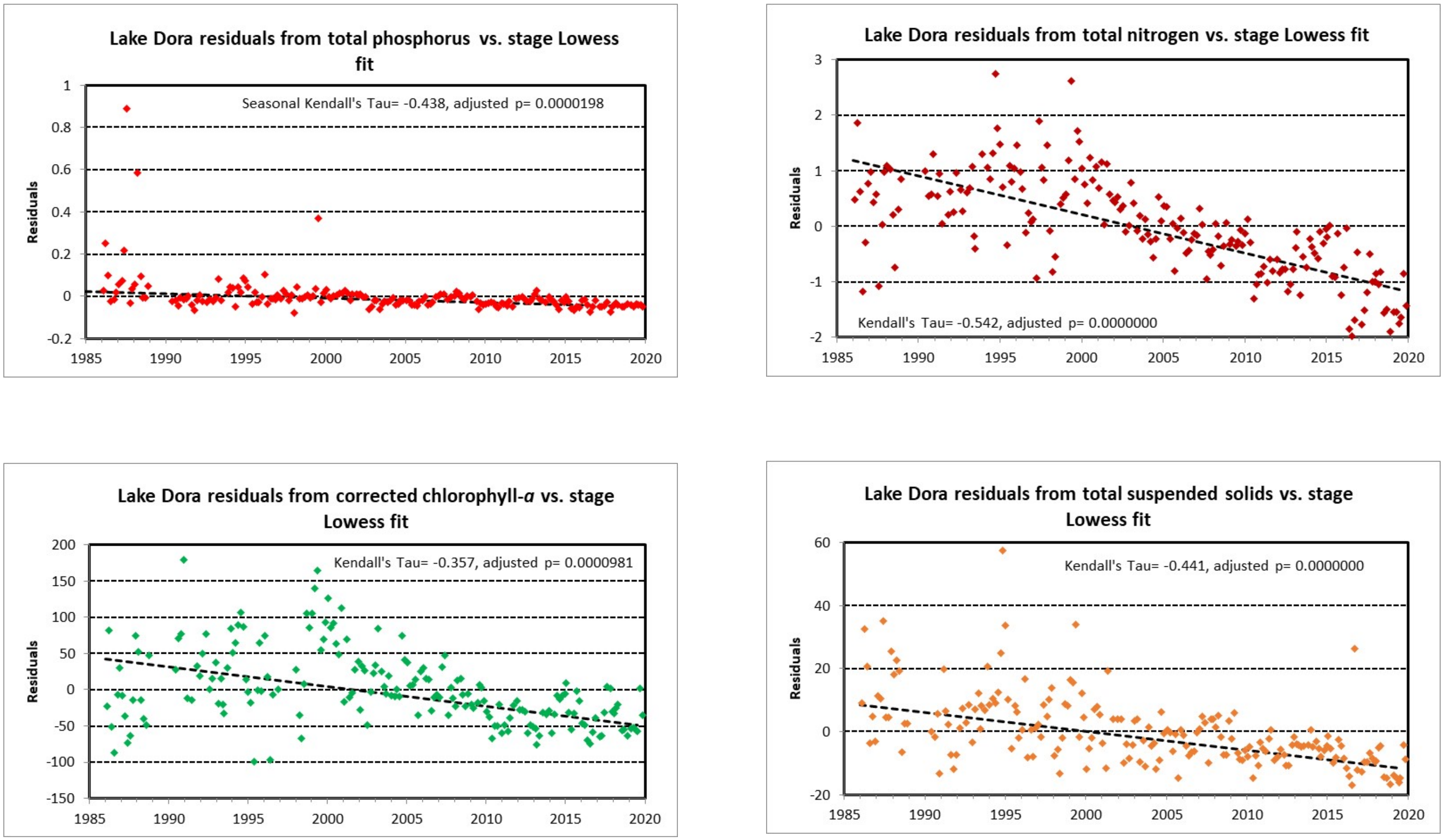
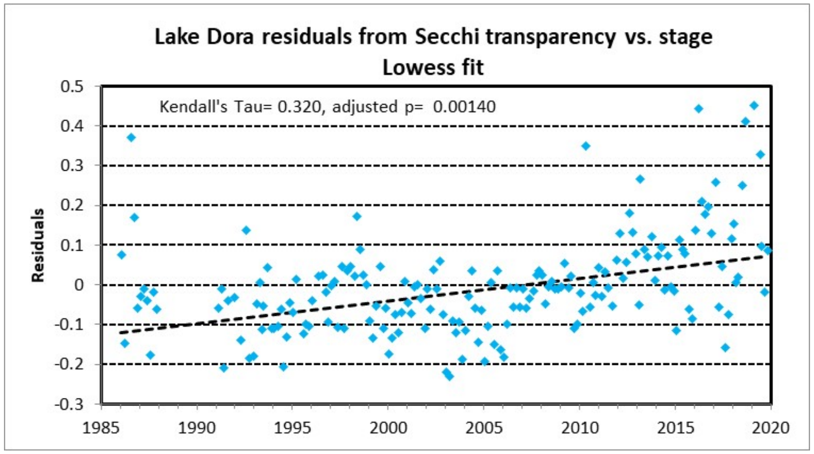
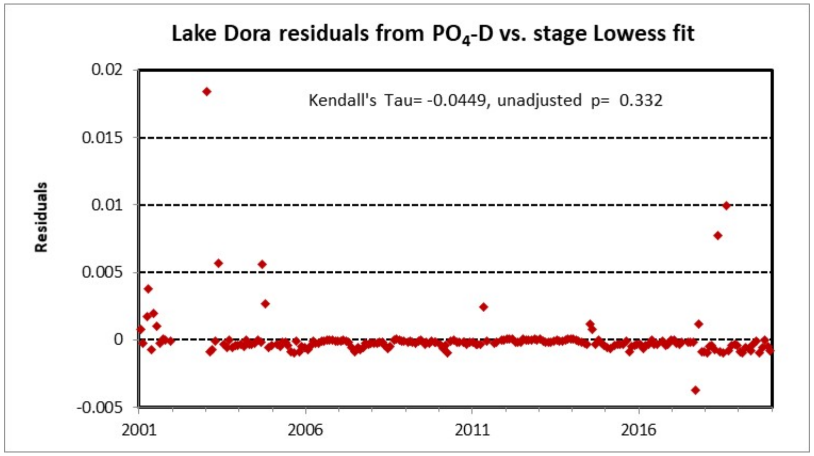
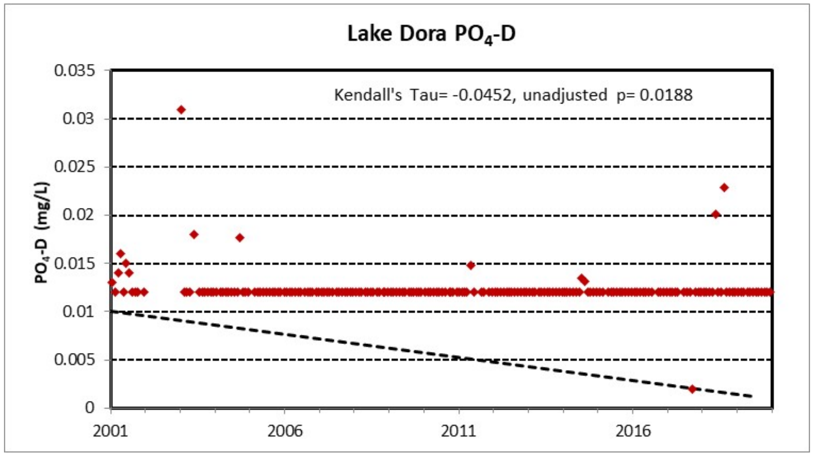


Figure 35. Water quality trends in Lake Dora, 1986-2019.

Daily means and Theil-Sen trend lines.







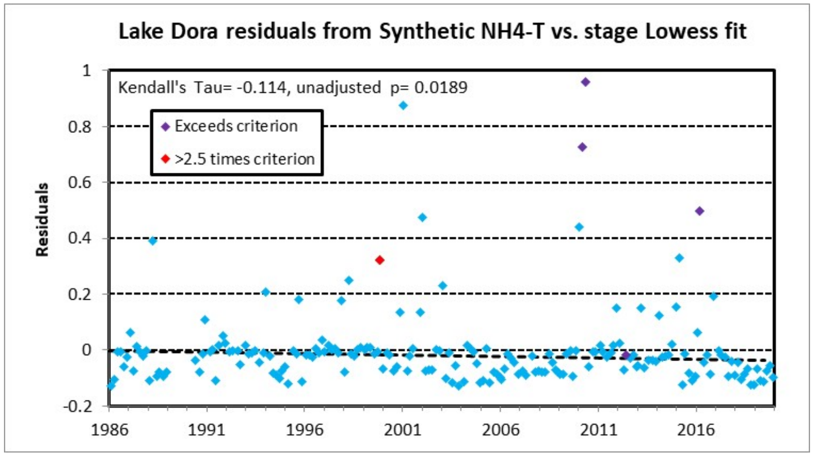


Figure 36. Water quality trends in Lake Dora, 1986-2019 and 2001-2019.

Daily means and Theil-Sen or ATS trend lines.

there was a significant decreasing trend for the unadjusted PO_4-D measurements but no significant trend in residuals from the Lowess smoothed relationship between PO_4-D and lake water elevation. This result may indicate that the apparent decreasing trend in PO_4-D was associated with changes in water elevations over this period rather than representing an independent trend through time (Figure 35, Figure 36). However, the ATS trend line does not provide a good fit to the PO_4-D data points, seemingly overly influenced by an outlier point. Re-doing the analysis deleting that point did not improve the trend line fit. The trend tests for PO_4-D may be questionable. Although the analysis is designed for data sets with non-detects, over 93% of the values in this data set were below MDL. Synthetic NH_4-T showed decreasing trends in the water level-adjusted residuals for the 1986-2019 time period when analyzed using non-detect methods, but not when analyzed with standard methods. The significance of the non-detect trend may be due to the absence of adjustment for serial correlation in those analyses. There were several exceedances of the Florida ammonia standard (Figure 36).

Overall, the results of the trend analyses for Lake Dora were very similar to the previous trend assessment (Fulton 2018), reflecting a continuation of the improving water quality apparent at that time. The Lake Dora trends were also very similar to those for upstream Lake Beauclair. As with Lake Beauclair, the relatively high nutrient loading for the last three years does not appear to have adversely affected water quality in Lake Dora. Again, the higher TP loads in the last three years (Figure 31) were primarily due to a large volume of discharge from upstream, and the concentration of TP in those discharges has been relatively low.

The phytoplankton monitoring in Lake Dora ended in August 2018. Lake Dora phytoplankton have been dominated by cyanobacteria throughout the period of record, other than a few samples that showed high biovolume of diatoms (Figure 37). Unlike chlorophyll-*a*, there has been no clear trend in phytoplankton biovolume in Lake Dora. Assessment of trends in phytoplankton biovolume in Lake Dora is complicated by the changes in analysts, and in station locations (from 2007-2011 phytoplankton samples were composites of east and west lake stations, but for other periods data in Figure 36 are from the west lake station. Like Lake Beauclair, 2010 and 2016 had both low chlorophyll-*a* and low phytoplankton biovolume in Lake Dora. As with Lake Beauclair, BSA identified *Cylindrospermopsis* as the dominant phytoplankton genus (2007-2018), but Phycotech identified *Cylindrospermopsis* and *Pseudanabaena* as co-dominant (1998-2007) (Figure 38). Again, as with Lake Beauclair, *Microcystis* and *Planktolyngbya* were more prominent in the BSA analyses than for Phycotech.

An algal bloom was observed in Lake Dora on March 12, 2020 (Figure 39). Samples were collected for qualitative assessment of the dominant taxa and measurement of cyanotoxins. The dominant taxon was *Microcystis aeruginosa*. Cyanotoxin concentrations were below detection limits. Despite the apparent bloom, chlorophyll-*a* concentration at the near-center lake station on March 12 was only 14.4 μg/L, below average for Lake Dora.

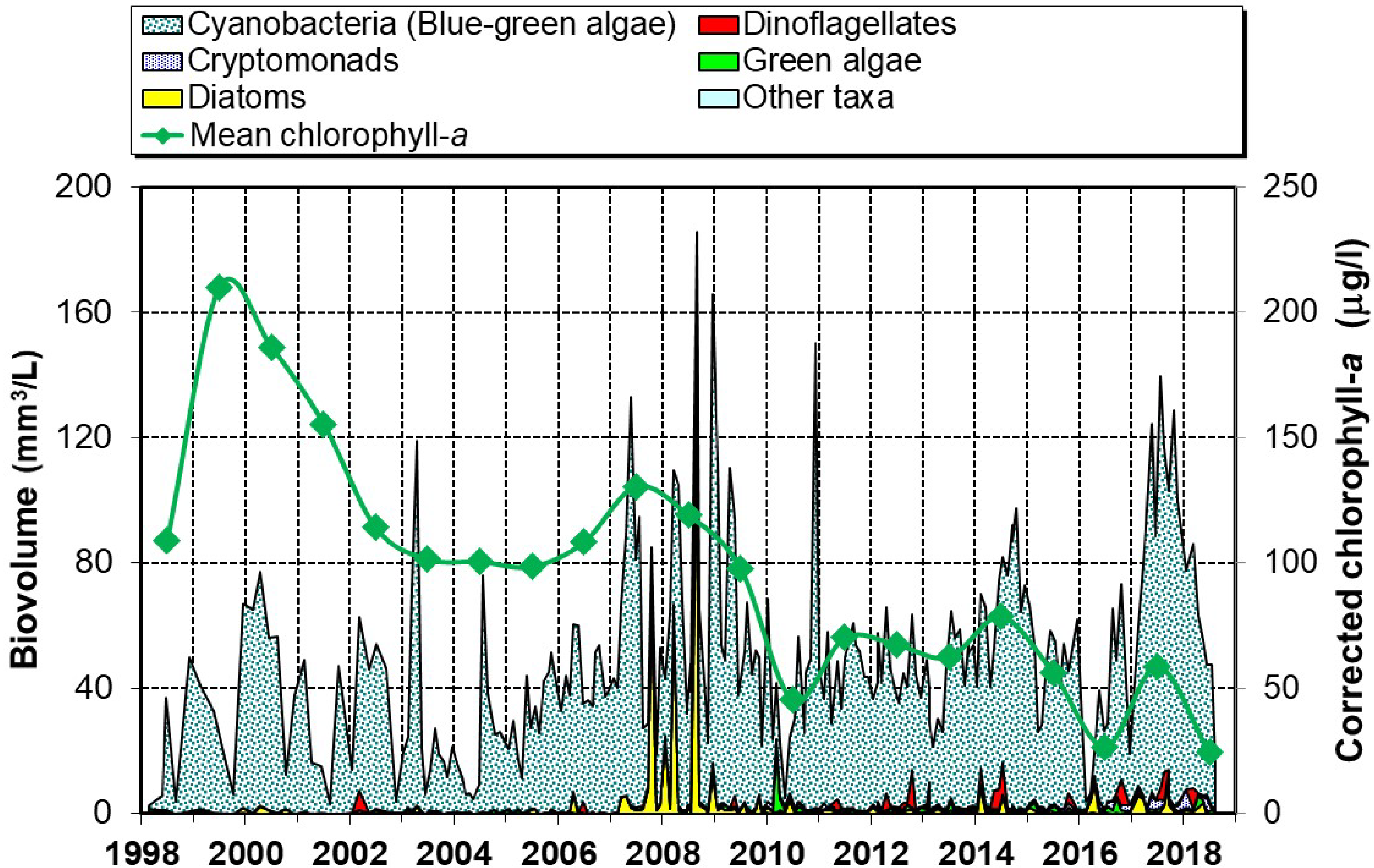


Figure 37. Lake Dora phytoplankton biovolumes and chlorophyll-*a* concentrations.

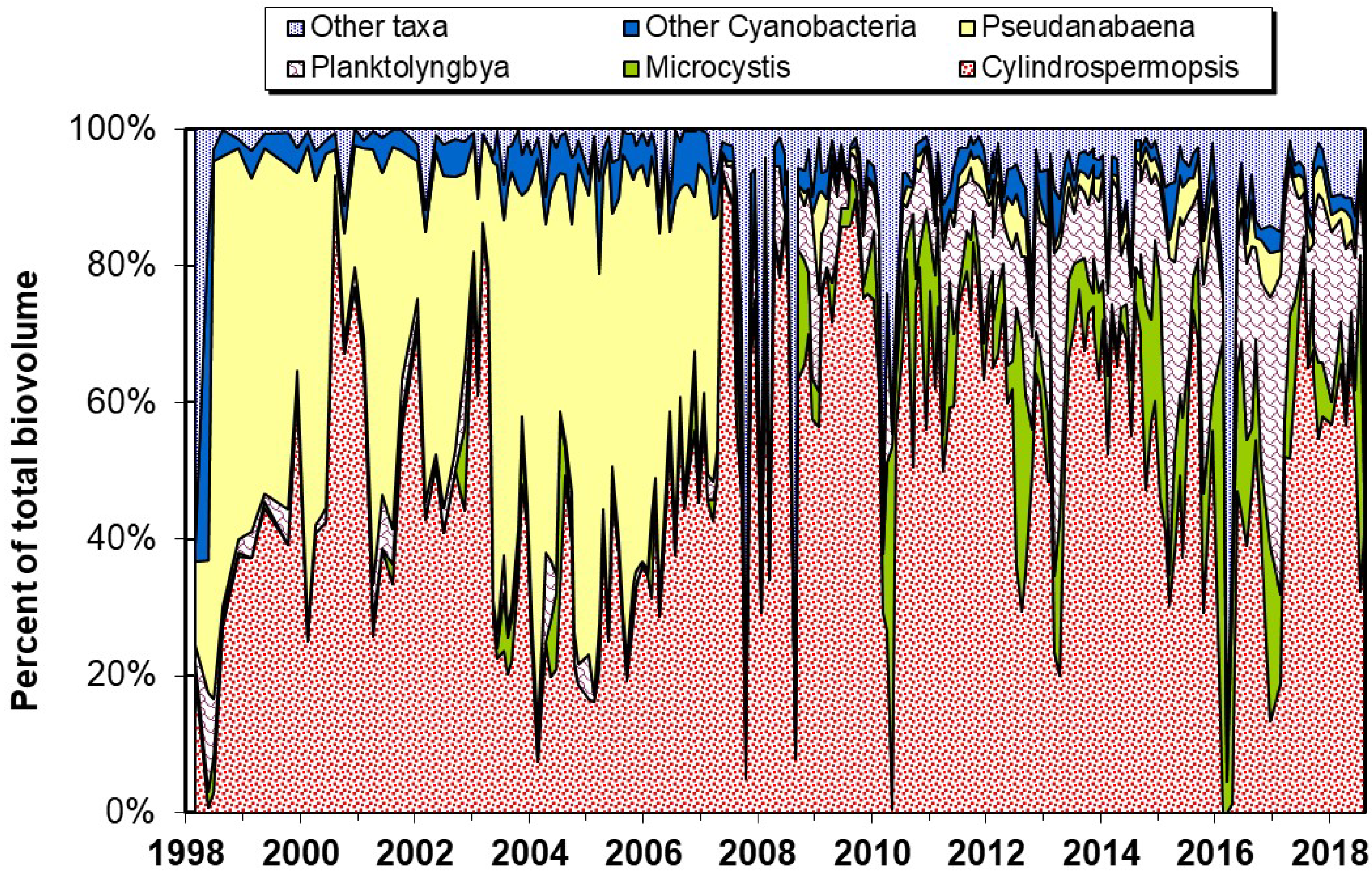


Figure 38. Lake Dora phytoplankton percent composition.

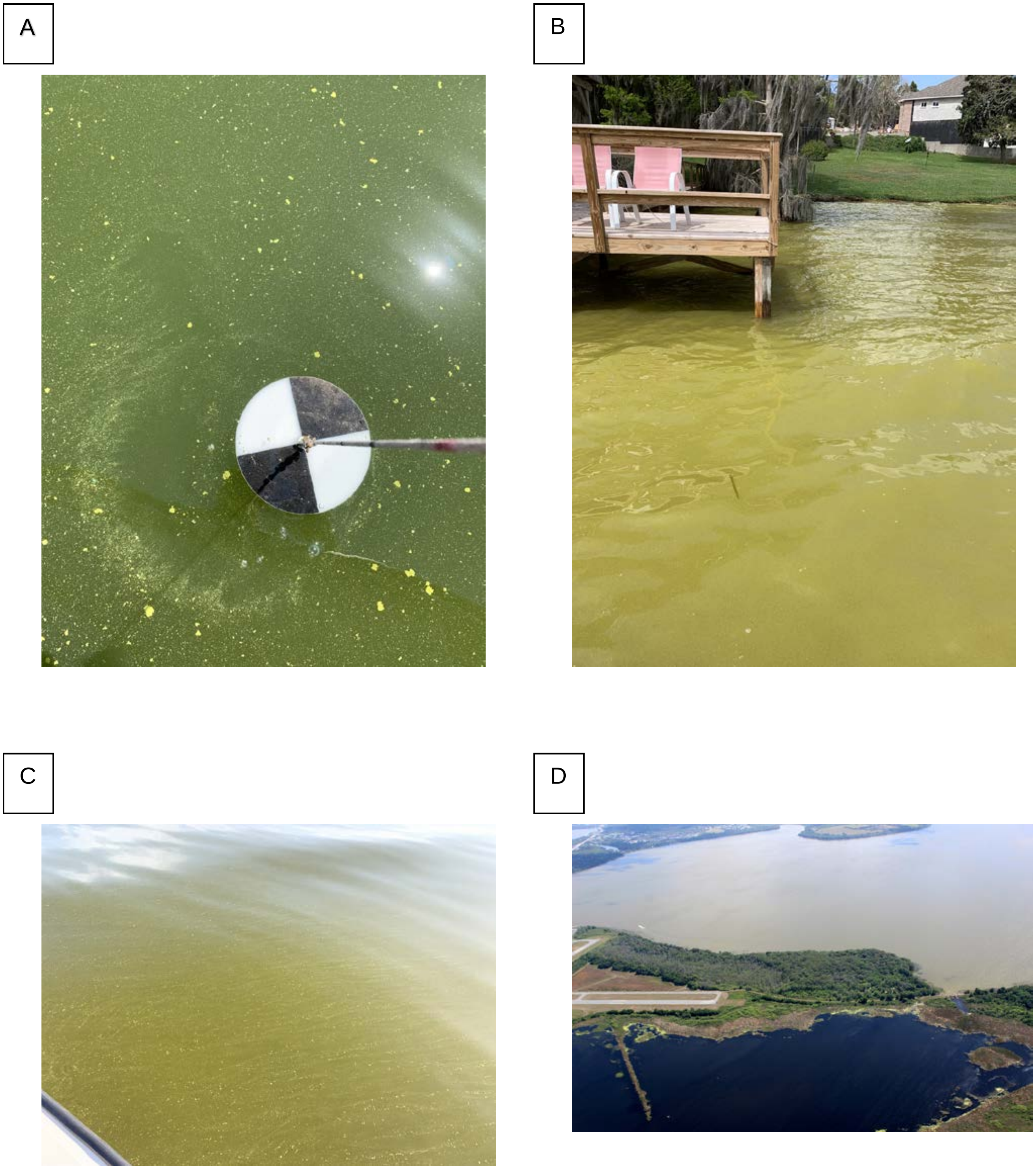


Figure 39. Pictures of algal blooms in lakes Dora and Harris, March 2020.

A. and B. Lake Dora March 12, 2020. C. Lake Harris March 11, 2020. D. Aerial photo of Lake Harris (top) and Harris Bayou Restoration Area (bottom) March 19, 2020.

LAKE HARRIS-LITTLE HARRIS

The external TP load to Lake Harris-Little Harris is divided among several sources, most of which have not shown much temporal trend (Figure 40, Appendix Table D- 7). One source that has decreased is the load from the Harris Bayou Restoration Area (Lake Harris Conservation Area). The last discharges from Harris Bayou to Lake Harris-Little Harris were in 2008 – since then the site has discharged to Lake Griffin. In some high-flow years there has been high TP loading to Lake Harris-Little Harris in tributary flows from the Palatlakaha River (particularly in 1998), but in many years loading from this source has been negligible. There was significant loading from the Palatlakaha River in 2015-2019, contributing to exceedances of the TMDL (8.3 MT/year) in those years. TP loading was below the TMDL in 5 of the 9 years prior to 2015.

The major external TP sources to Lake Harris-Little Harris for the last 5 years have been atmospheric deposition (30%), tributary discharges (19%; 18% from Palatlakaha River, <1% from Lake Eustis), urban-residential runoff (12%), septic tank effluents (11%), spring discharges (10%), natural area runoff (10%), and agriculture (8%). Lake County Water Authority has recently purchased and plans to restore the Lake Denham muck farm, located in the Lake Harris watershed west of the lake (Figure 1).

The external TN load to Lake Harris-Little Harris is also divided among several sources, most of which have not shown much temporal trend (Figure 41, Appendix Table D- 8). The Harris Bayou Restoration Area has always been a minor source of TN to the lake. The major external TN sources to Lake Harris-Little Harris for the last 5 years have been tributary discharges (32%; 32% from Palatlakaha River, <1% from Lake Eustis), atmospheric deposition (23%), natural area runoff (14%), spring discharges (12%), septic tank effluents (9%), urban-residential runoff (6%), and agriculture (4%).

TP, chlorophyll-*a*, and TN concentrations have been variable in Lake Harris-Little Harris, generally decreasing from 1999-2006, increasing during the 2007-2008 drought period, then generally decreasing, although there was some increase during recovery from another dry period in 2013-2014 (Figure 40 to Figure 43). Chlorophyll-*a* has shown some increase from its lowest level 2017, and TP sharply increased in the last half of 2020. The 2020 increase in TP concentrations may have resulted from relatively high external loading for the previous 5 years, or from herbicide treatments, primarily for *Hydrilla*. The Florida Fish and Wildlife Conservation Commission treated over 6,000 acres in fiscal year 2018-19 and over 400 acres in fiscal year 2019-20. However, other water quality parameters did not substantially increase in 2020. In the last 5 years, average concentrations have been 26 μg TP/L and 18 μg chlorophyll-*a*/L. These concentrations are equal to the TMDL target TP concentration of 26 μg/L, and below the expected chlorophyll-*a* concentration of 22 μg/L at the TP target. Average TN concentrations in the last 5 years have been 1.2 mg/L. Trends in concentrations and masses were rather similar in Lake Harris, other than an elevation of TN concentrations during the 2008-2009 drought (Figure 43, Figure 43).

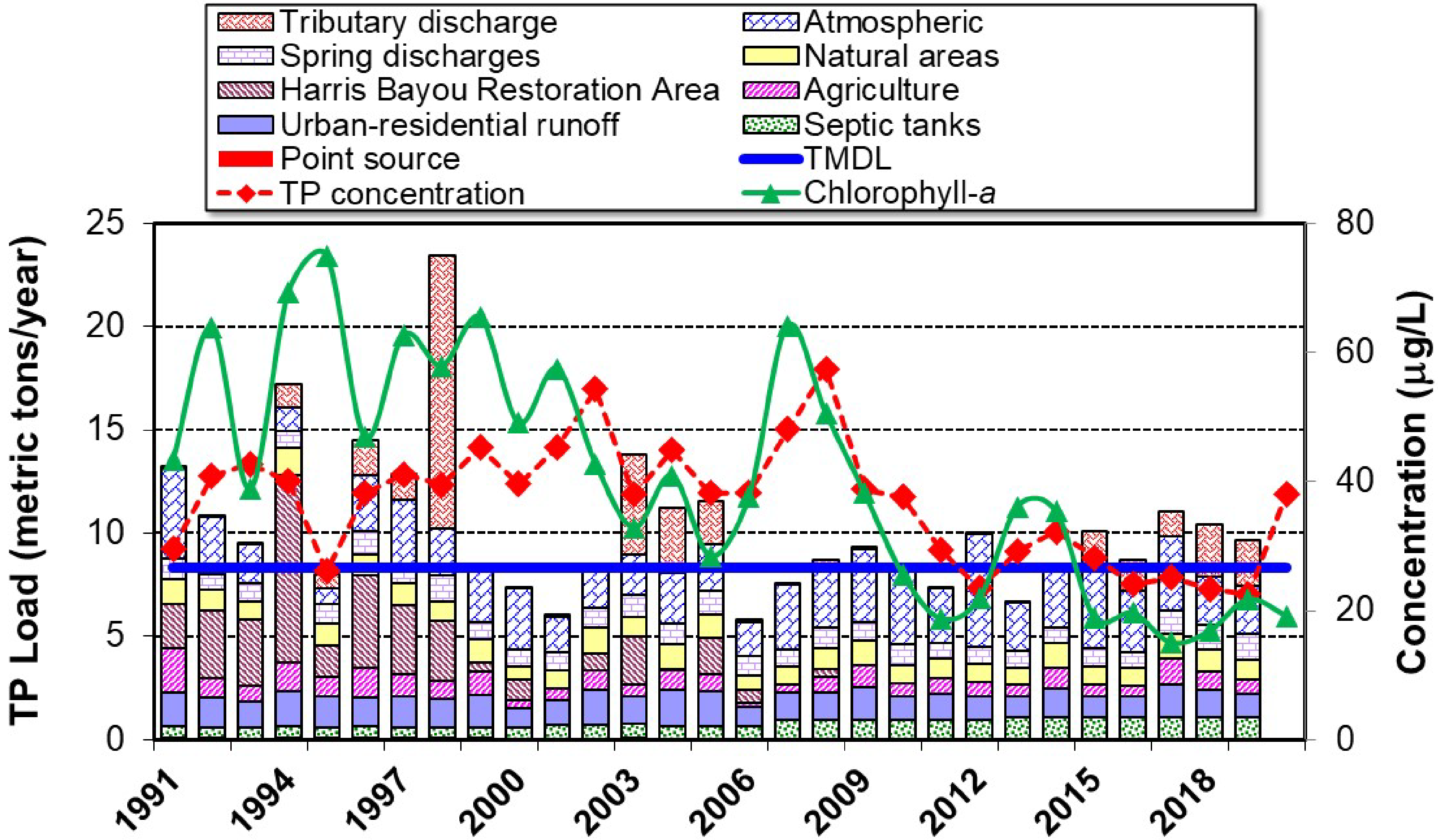


Figure 40. Lake Harris-Little Harris estimated annual external total phosphorus loads (bars), TMDL loading target, and mean annual concentrations of TP and chlorophyll-*a* (lines).

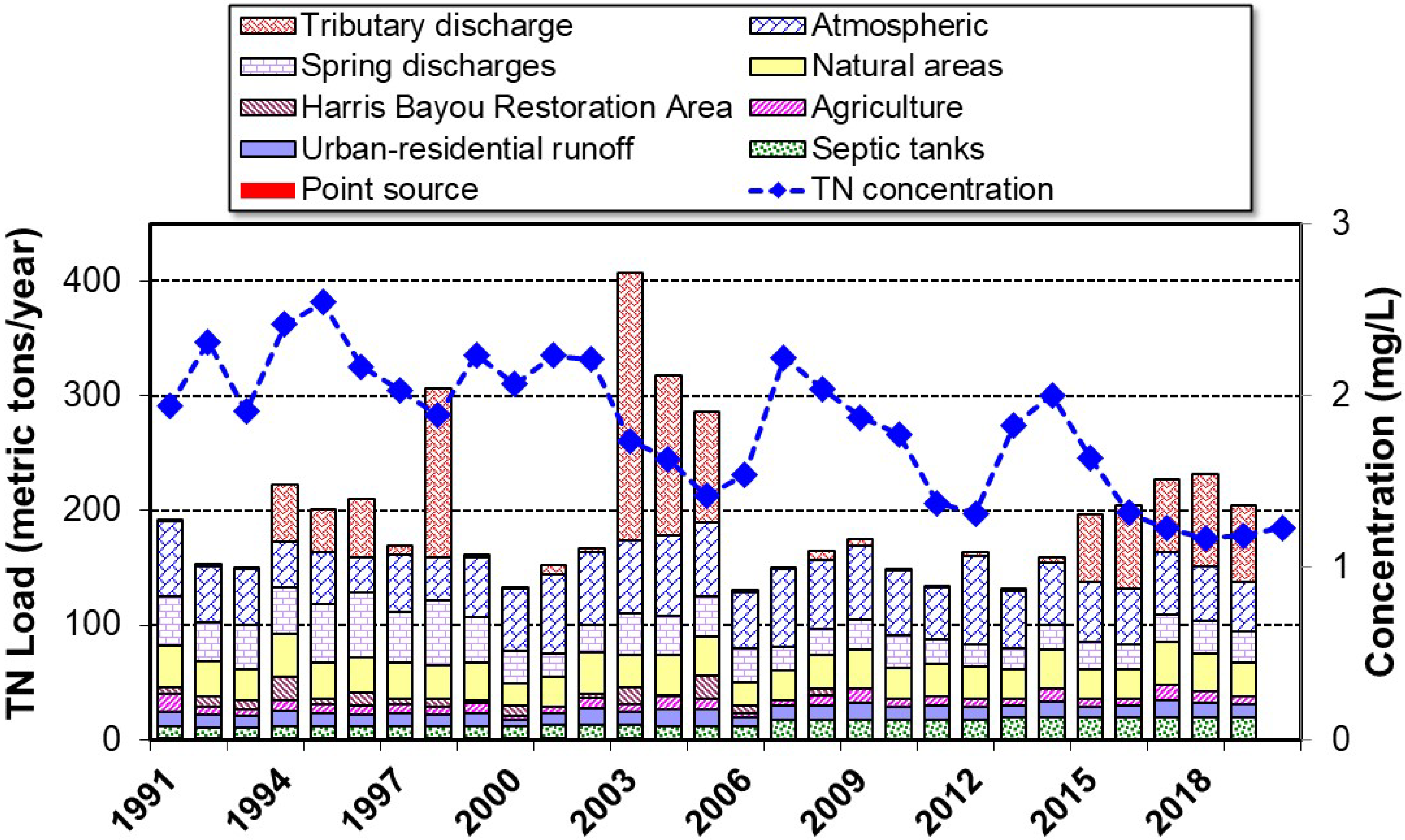


Figure 41. Lake Harris-Little Harris estimated annual external total nitrogen loads (bars), and mean annual concentrations of TN.

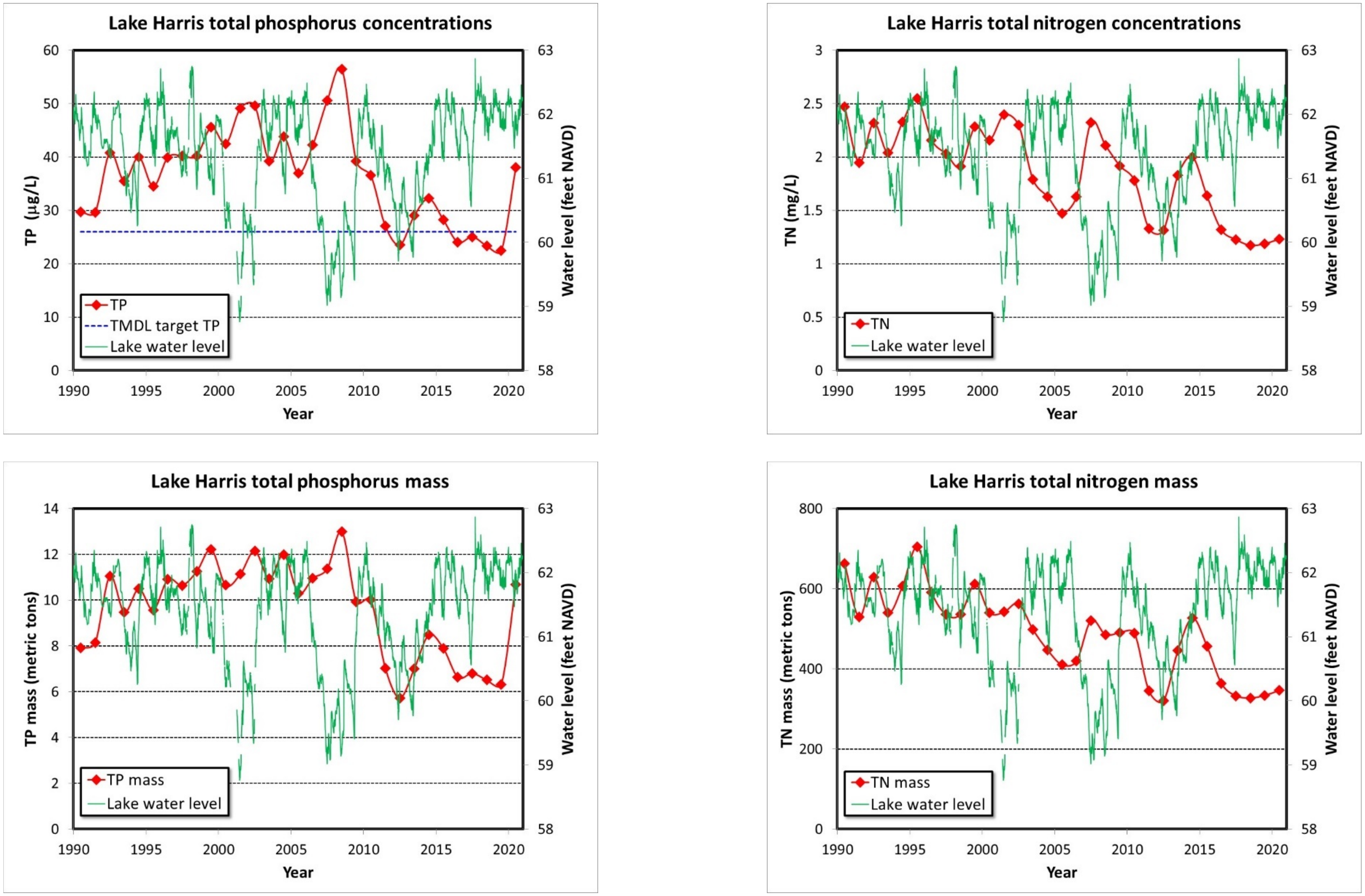


Figure 42. Lake Harris-Little Harris mean annual total phosphorus and total nitrogen concentrations and masses, and water elevations.

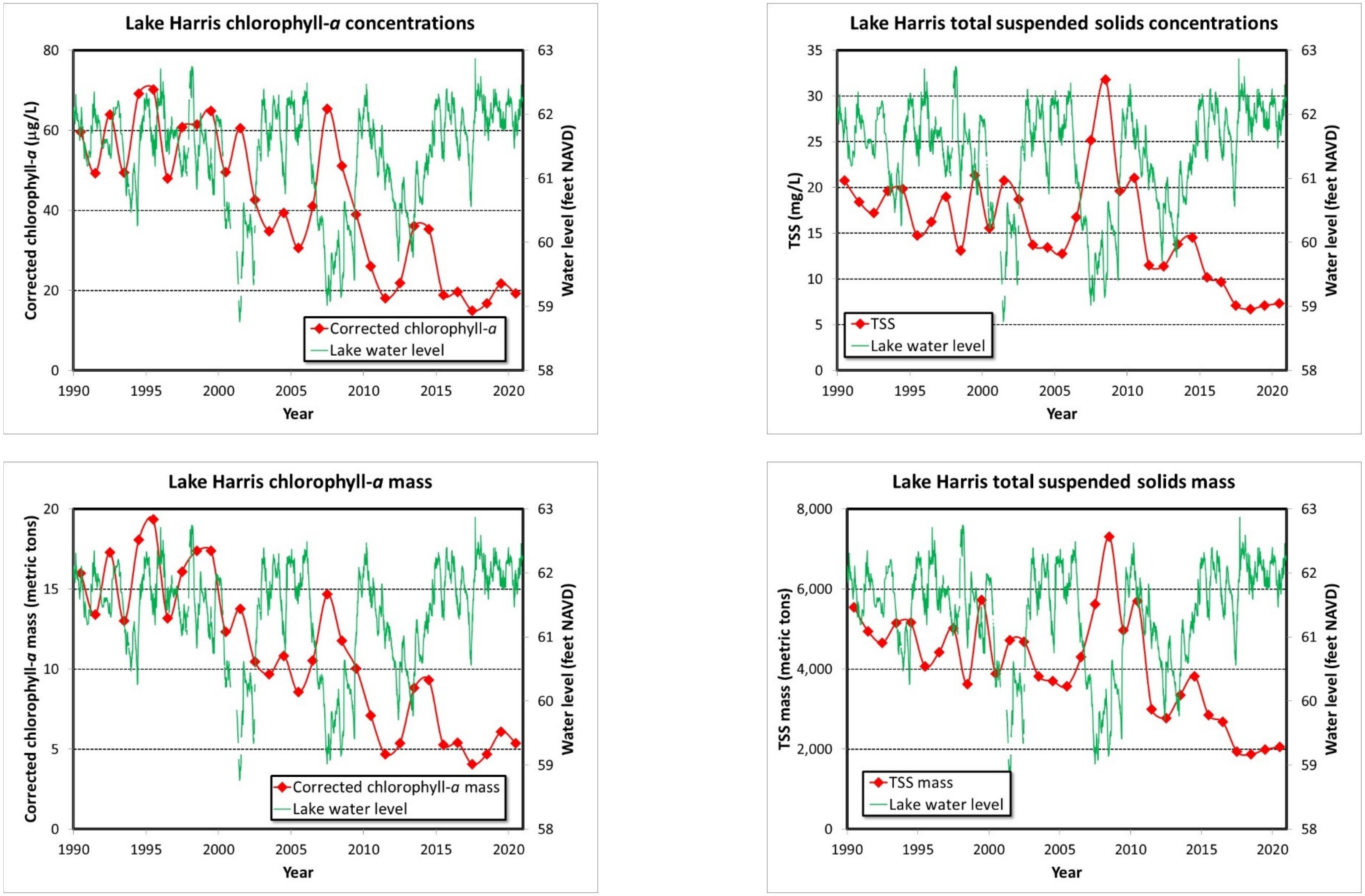


Figure 43. Lake Harris-Little Harris mean annual total chlorophyll-*a* and total suspended solids concentrations and masses, and water elevations.

Results of water quality trend analyses for Lake Harris-Little Harris are summarized in Table 7, with more detailed results including Kendall's tau nonparametric correlation coefficient and statistical significance level of the nonparametric Theil-Sen trend lines in Appendix Table B- 4. Time series plots are shown for several parameters in Figure 44 and Figure 45.

Table 7. Water quality trends for Lake Harris-Little Harris.

Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality, $p<0.05$

(0) No trend, $p\geq0.05$

(–) Significant decreasing trend, $p<0.05$

(+) Significant increasing trend, $p<0.05$

	1990 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, –	S, –	–	–
TN	S, –	S, –	–	–
Chlorophyll-*a*	S, –	S, –	–	S, –
TSS	S, –	S, –	–	–
Secchi	S, +	S, +	+	S, +
NH_4 [a, b]	–	0	S, 0	S, 0
NO_x [a, b]	S, 0	S, 0	S, 0	S, –
PO_4 [b]	ND [c]	ND	–	0
TP mass	S, –		–	
TN mass	S, –		–	
Chlorophyll-*a* mass	S, –		–	
TSS mass	S, –		–	

[a] Synthetic total fraction for 1990-2019; dissolved fraction for 2001-2019

[b] Non-detect methods

[c] ND – No data

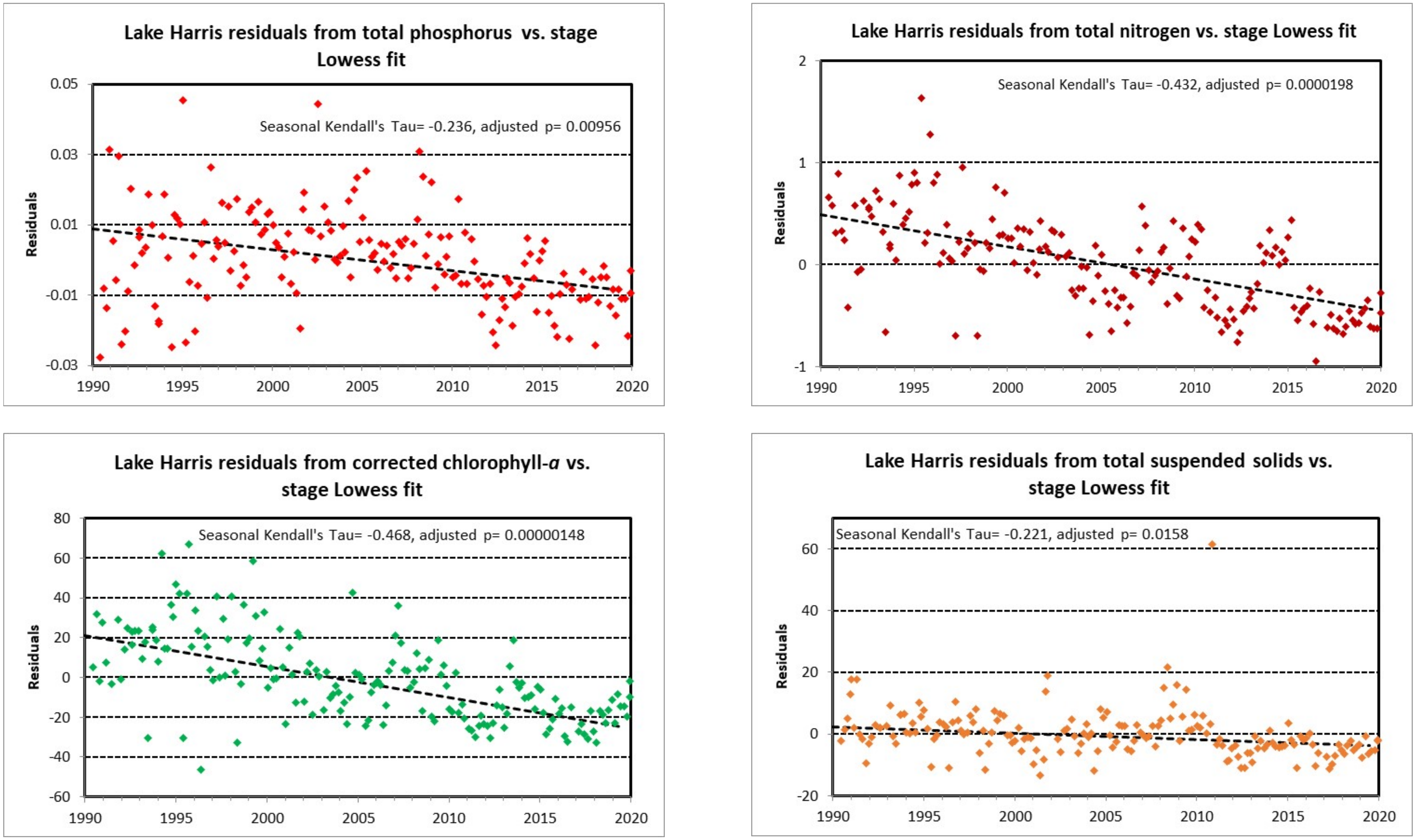
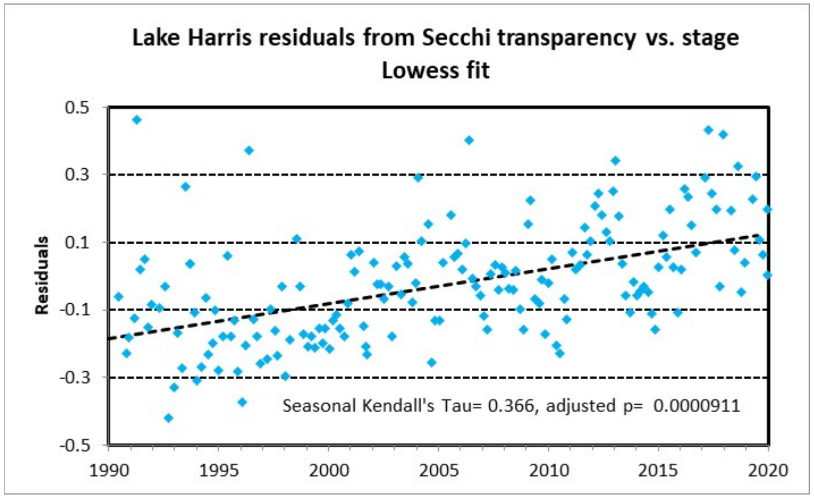
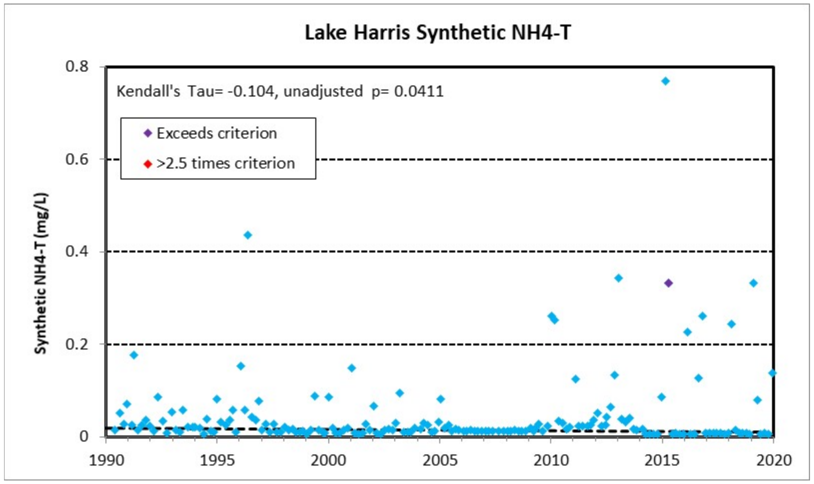
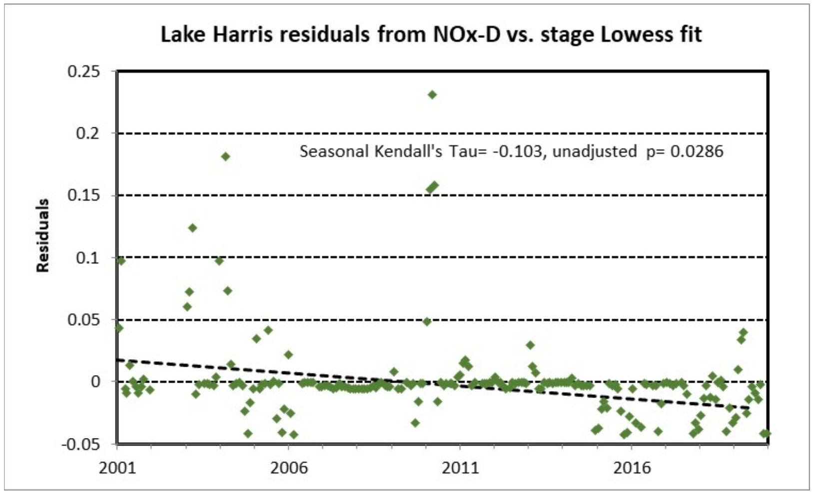


Figure 44. Water quality trends in Lake Harris-Little Harris, 1990-2019.

Daily means and Theil-Sen trend lines.







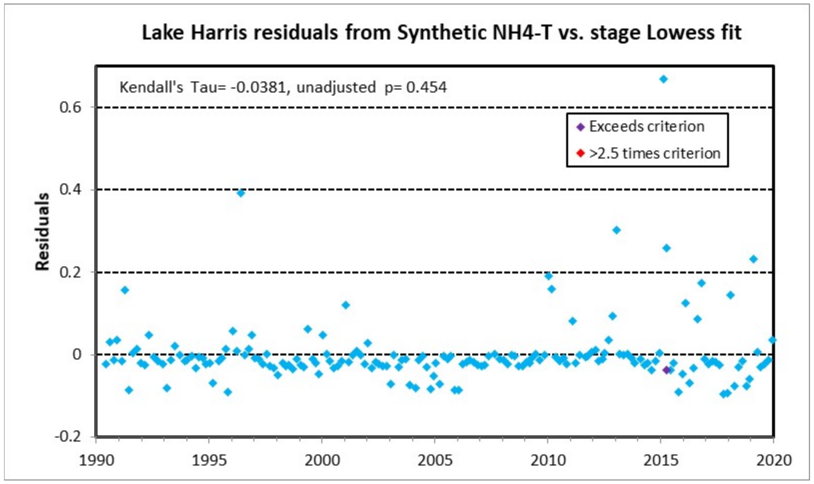


Figure 45. Water quality trends in Lake Harris-Little Harris, 1991-2019 and 2001-2019.

Daily means and Theil-Sen trend lines.

Lake Harris-Little Harris showed improving trends for most water quality variables. For both time periods, there were decreasing trends for both concentrations and masses of TP, TN, chlorophyll-*a*, and TSS, and an increasing trend in Secchi transparency. In most cases, adjusting for changes in water elevations did not substantially affect the trend analyses. Over the 2001-2019 time period, NO_x-D showed no trend but residuals for the Lowess smooth did show a decreasing trend, suggesting water level fluctuations may have masked a decreasing trend. Over the 2001-2019 time period, PO_4-D showed a weak decreasing trend (Table 7), but there was no significant trend in residuals from the Lowess smooth. For the full period of record, Synthetic NH_4-T analyzed methods showed a decreasing trend, but there was no significant trend in residuals from the Lowess smooth – this indicates that the apparent improving trend may have been associated with changes in water elevations over this period rather than representing an independent trend through time. There were only two exceedances of the Florida ammonia standard, in 2015 (only one in the data subset used for the trend analysis in Figure 45, but there was one other in the full data set).

The results of the trend analyses for Lake Harris-Little Harris were similar to the previous trend assessment (Fulton 2018), but this time there were a few more significant improving trends and generally the Kendall’s tau coefficients were larger. This appears to reflect further improvement in water quality since 2016.

The phytoplankton monitoring in Lake Harris ended in August 2018. Lake Harris phytoplankton have generally been dominated by cyanobacteria although diatoms have been more significant than in Apopka, Beauclair, or Dora, particularly during the cool season (Figure 47, Figure 47). Phytoplankton biovolumes in Lake Harris tended to parallel trends in chlorophyll-*a*, particularly with a general decrease from 2007-2012, an increase in 2013-2014, and a subsequent decrease. *Cylindrospermopsis raciborskii* has tended to be the dominant cyanobacterial genus in Lake Harris although it appears to be decreasing in prominence in the last several years. The increase in chlorophyll-*a* in 2013-2014 was apparently due to a bloom of *Cylindrospermopsis*. Unlike in lakes Beauclair and Dora, *Pseudanabaena* was not prominent in Lake Harris (Figure 47).

An algal bloom was observed in Lake Harris in March 2020 (Figure 39). Samples were collected for qualitative assessment of the dominant species and measurement of cyanotoxins. The co-dominant taxa were *Microcystis aeruginosa* and *Cylindrospermopsis raciborskii*. Cyanotoxin concentrations were below detection limits. Chlorophyll-*a* concentration in Lake Harris on March 11 was 46.7 μg/L, well above average. TP concentration was not high on this date (23.6 μg/L) but did increase later in the year. By the next sample date on April 8, chlorophyll-*a* concentration had decreased to 26.2 μg/L.

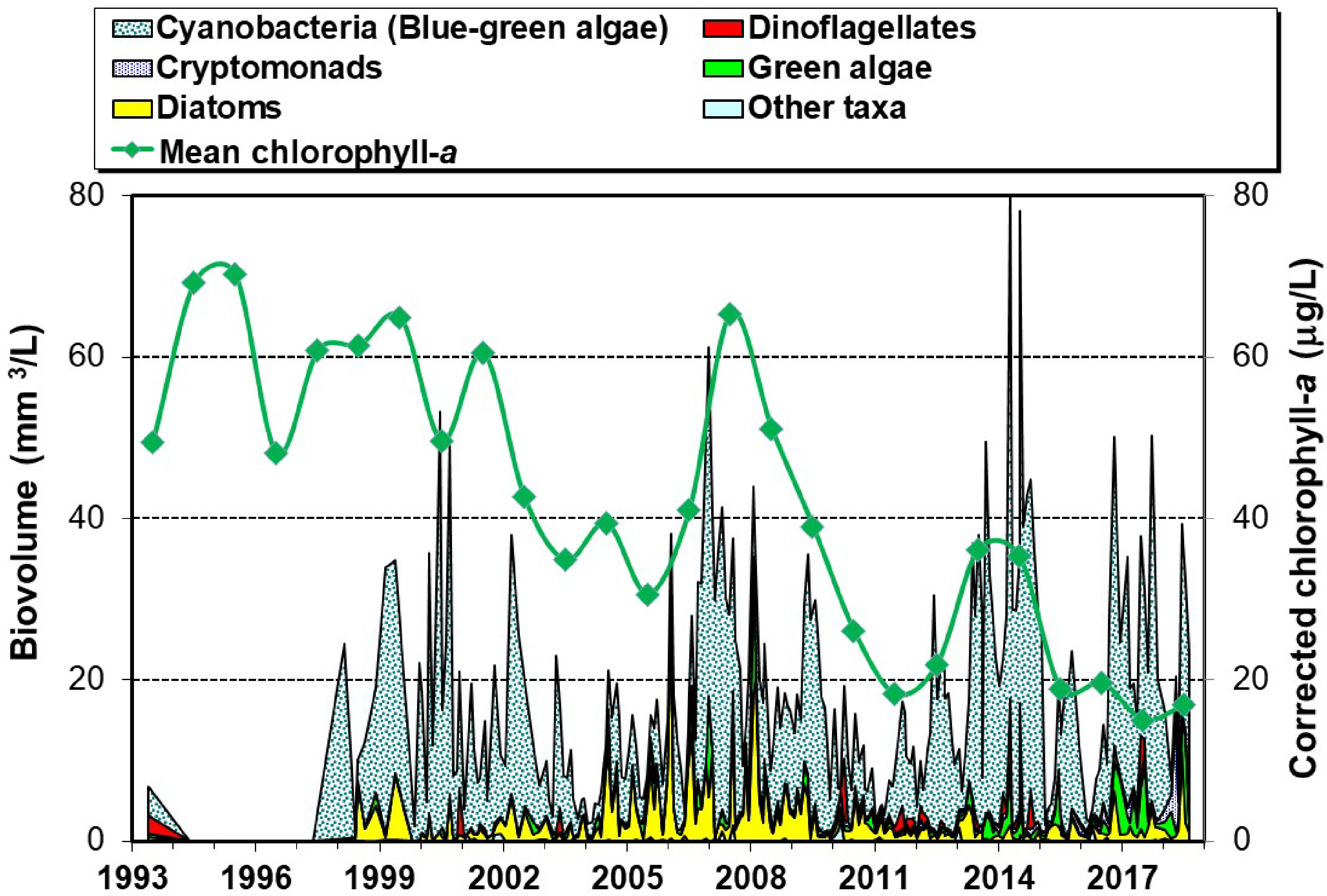


Figure 46. Lake Harris phytoplankton biovolumes and chlorophyll-*a* concentrations.

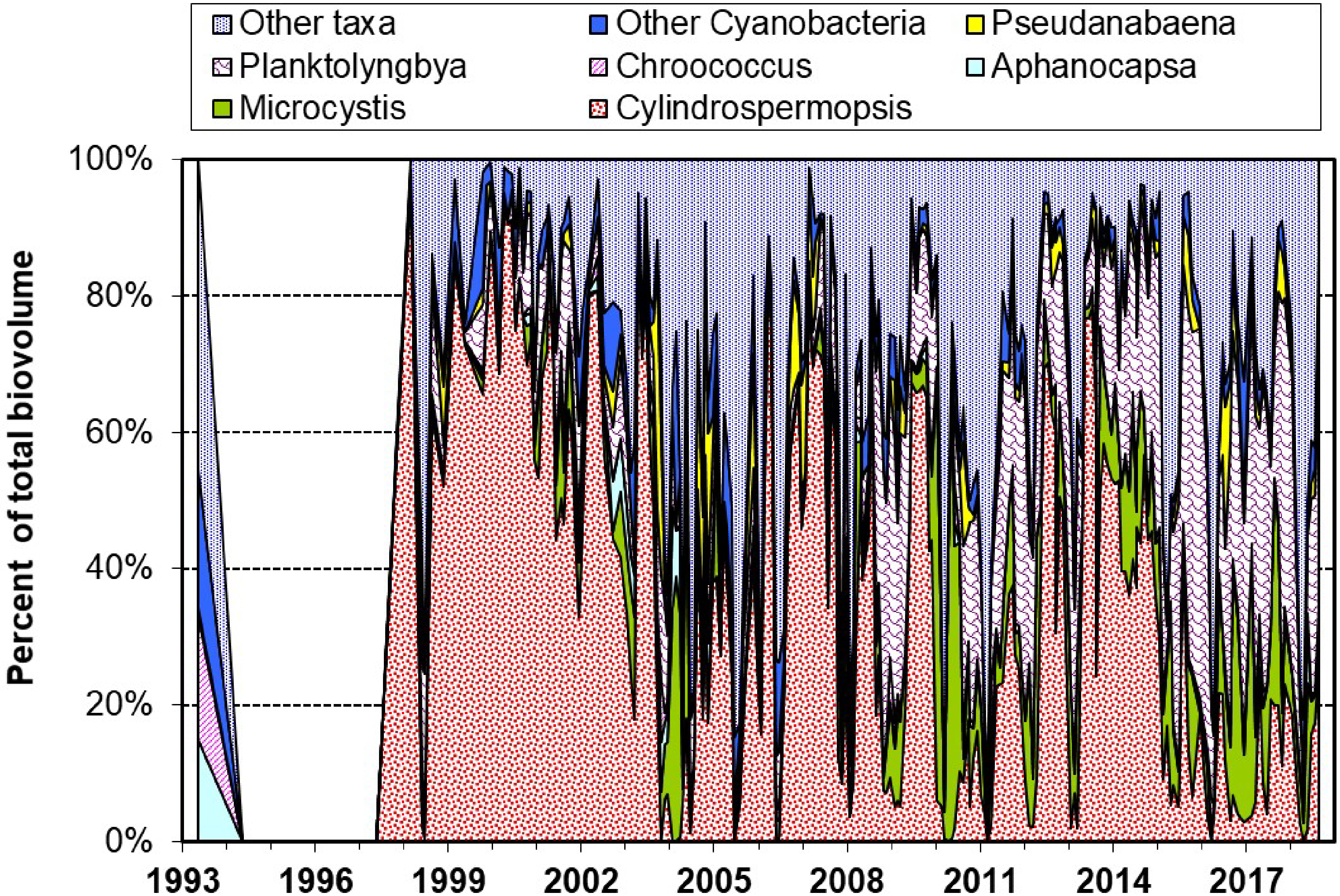


Figure 47. Lake Harris phytoplankton percent composition.

LAKE EUSTIS

The external TP load to Lake Eustis is divided among several sources, most of which have not shown much temporal trend (Figure 48, Appendix Table D- 9). One source that has generally decreased is tributary loads from the upstream lakes Dora and Harris-Little Harris, although they have increased over the last 3 years. These tributary decreases are due to a combination of low flow volumes during droughts in recent years and decreases in TP concentrations in the upstream lakes. The external TP load to the lake has slightly exceeded the adopted TMDL (9.2 MT/year) for the last 3 years but was below the TMDL in 10 of the previous 11 years.

The major external TP sources to Lake Eustis for the last 5 years have been Lake Harris-Little Harris discharges (27%), Lake Dora discharges (23%), urban-residential runoff (14%), atmospheric deposition (14%), septic tank effluents (12%), and natural area runoff (5%).

The external TN load to Lake Eustis is also divided among several sources, with the main temporal trend being a decreased tributary load from the upstream lakes Dora and Harris-Little Harris (Figure 49, Appendix Table D- 10). The major external TN sources to Lake Eustis for the last 5 years been Lake Harris-Little Harris discharges (43%), Lake Dora discharges (33%), septic tank effluents (7%), atmospheric deposition (7%), natural area runoff (5%), and urban-residential runoff (4%).

TP, chlorophyll-*a*, and TN concentrations have been variable in Lake Eustis, but have generally been decreasing since 2008 (Figure 49, Figure 49). In the last 5 years, average concentrations have been 26 μg TP/L and 19 μg chlorophyll-*a*/L. These concentrations are slightly above the TMDL target TP concentration of 25 μg/L and meet the expected chlorophyll-*a* concentration of 20 μg/L at the TP target. Average TN concentrations over the last 5 years were 1.4 mg/L. Trends in concentrations and masses are generally similar in Lake Eustis, although there is some indication of greater increase in concentrations during the 2007-2008 drought (Figure 51, Figure 51). Water quality tended to deteriorate during the 2002 and 2007-2008 droughts.

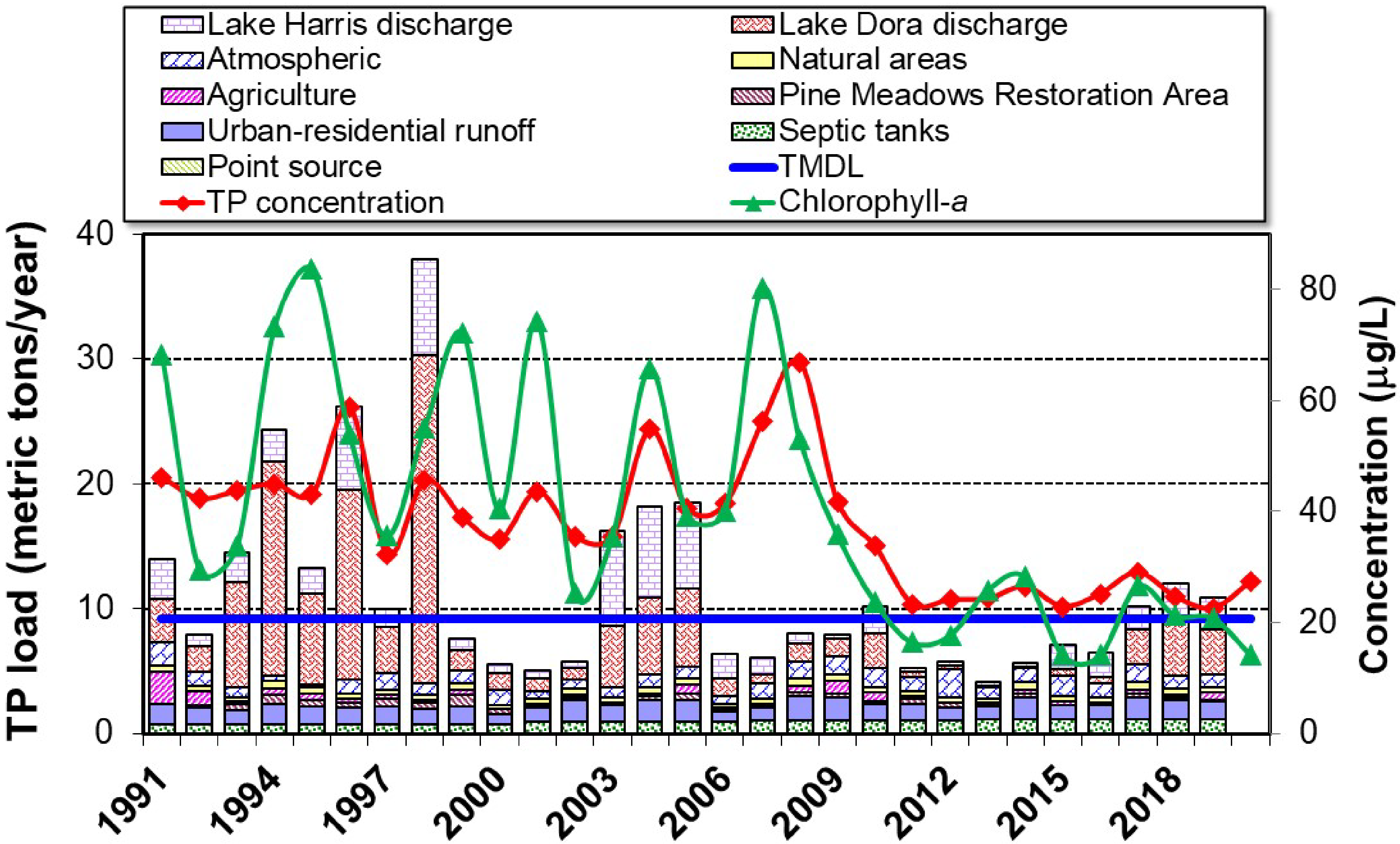


Figure 48. Lake Eustis estimated annual external total phosphorus loads (bars), TMDL loading target, and mean annual concentrations of TP and chlorophyll-*a* (lines).

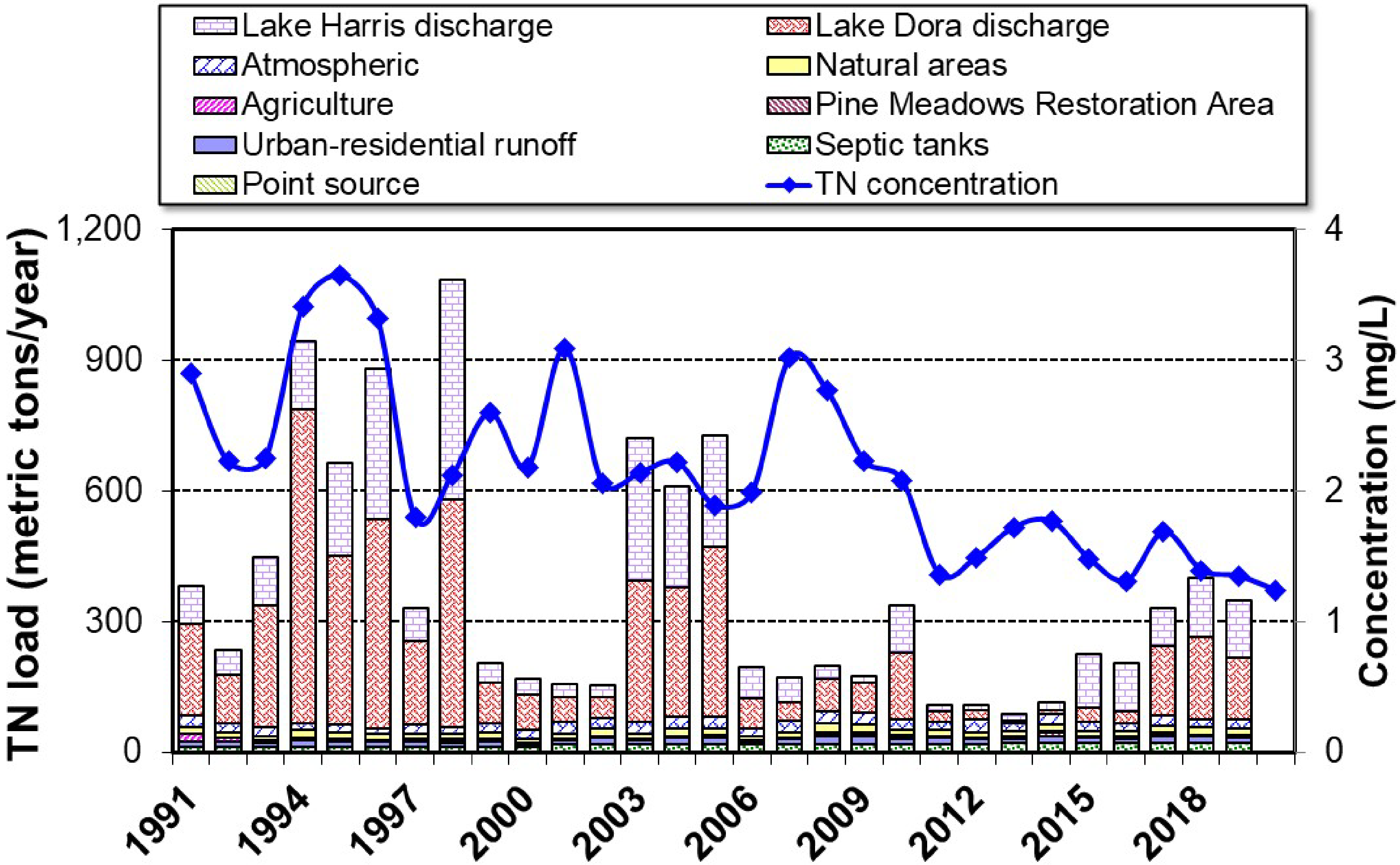


Figure 49. Lake Eustis estimated annual external total nitrogen loads (bars), and mean annual concentrations of TN.

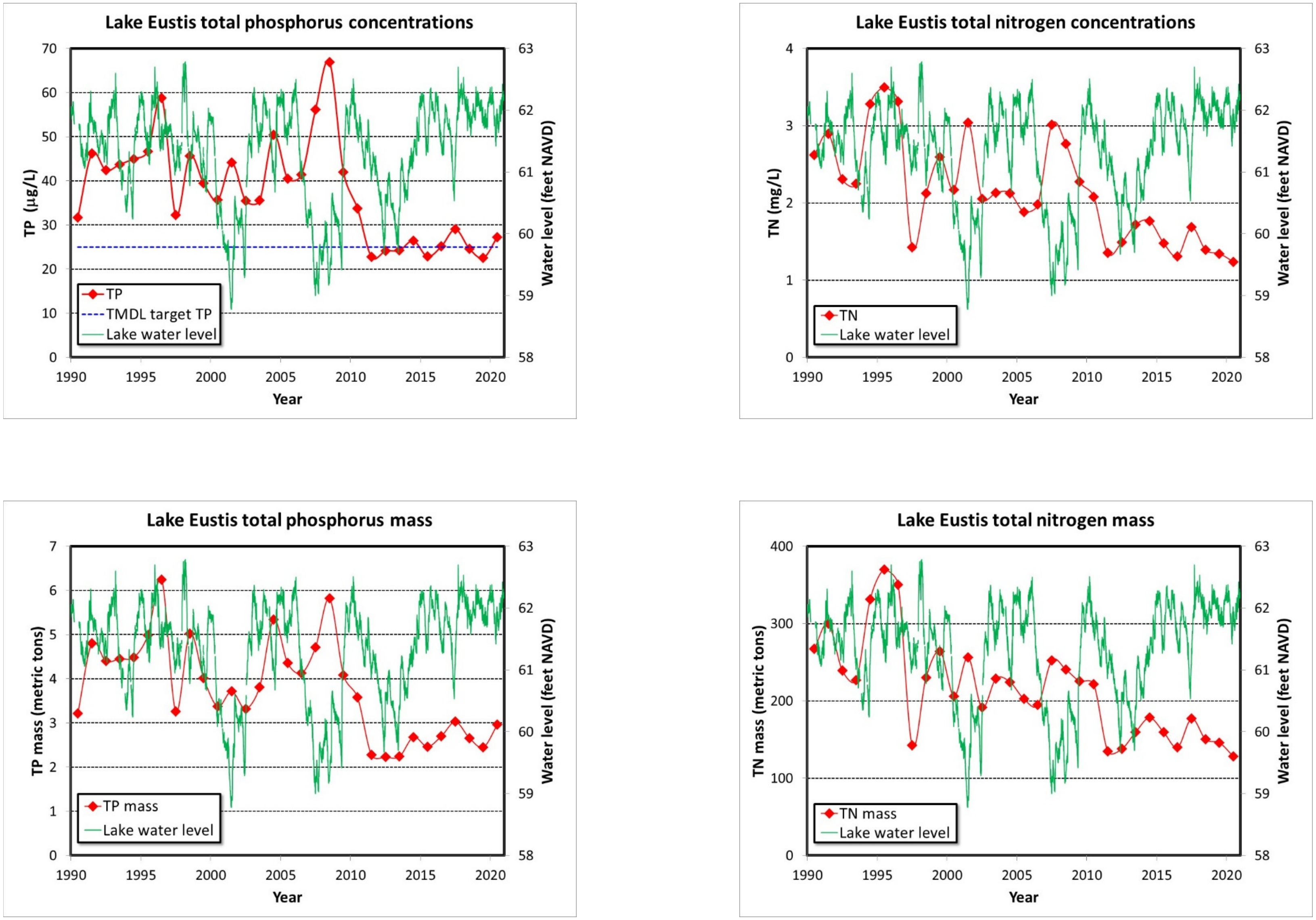


Figure 50. Lake Eustis mean annual total phosphorus and total nitrogen concentrations and masses, and water elevations.

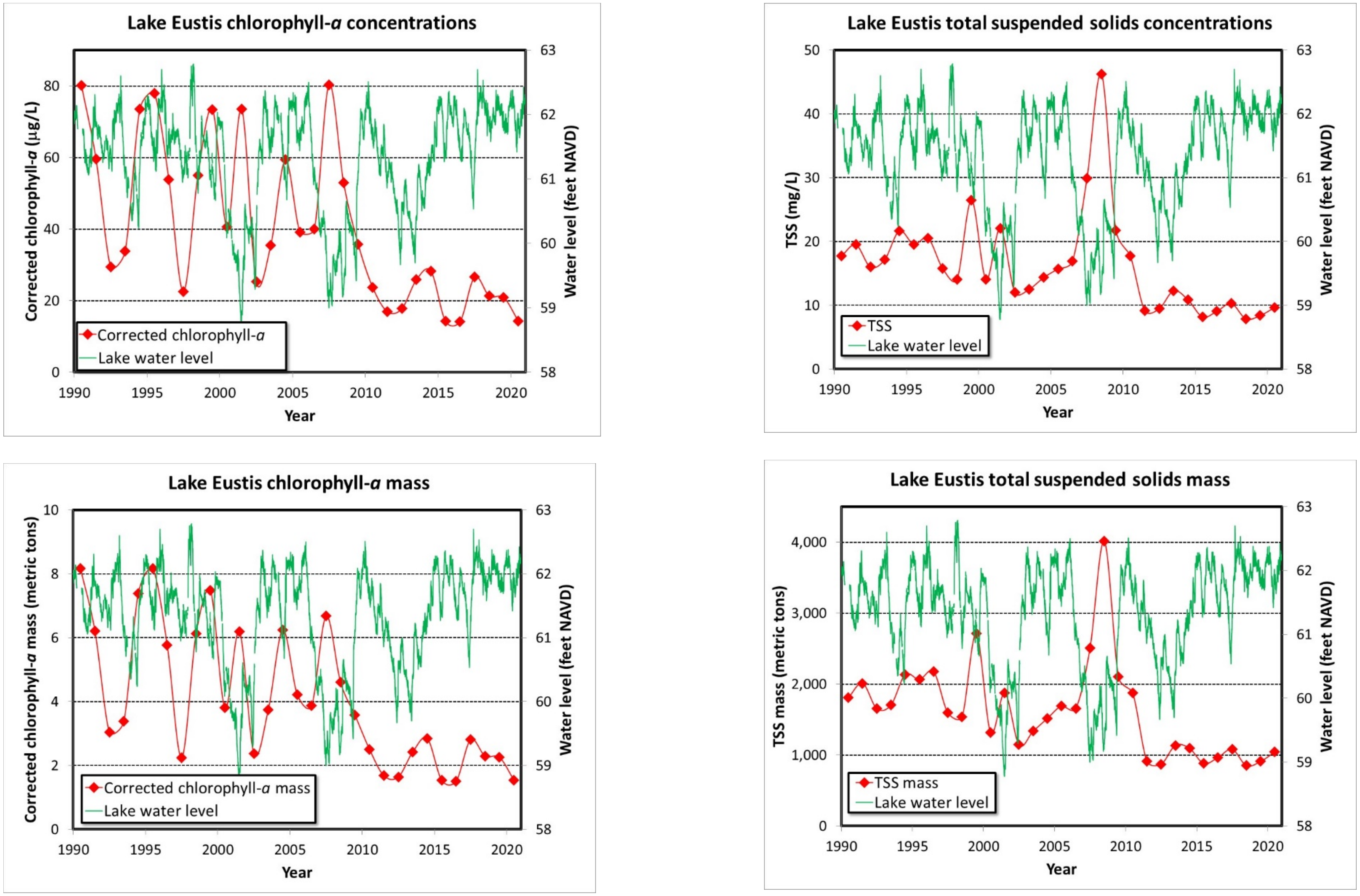


Figure 51. Lake Eustis mean annual total chlorophyll-*a* and total suspended solids concentrations and masses, and water elevations.

Results of water quality trend analyses for Lake Eustis are summarized in Table 8, with more detailed results including Kendall's tau nonparametric correlation coefficient and statistical significance level of the nonparametric Theil-Sen trend lines in Appendix Table B- 5. Time series plots are shown for several parameters in Figure 52 and Figure 53.

Table 8. Water quality trends for Lake Eustis.

Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality, $p<0.05$

(0) No trend, $p \geq 0.05$

(–) Significant decreasing trend, $p<0.05$

(+) Significant increasing trend, $p<0.05$

	1990 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, –	S, –	–	–
TN	–	–	–	–
Chlorophyll-*a*	–	–	S, –	S, –
TSS	–	–	–	0
Secchi	+	+	S, +	S, +
NH_4 [a, d]	S, 0	–		
NH_4 [a, b]	S, –	S, –	S, 0	S, 0
NO_x [a, b]	0	–	S, 0	S, 0
PO_4 [b]	ND [c]	ND	–	0
TP mass	S, –		–	
TN mass	–		–	
Chlorophyll-*a* mass	–		S, –	
TSS mass	–		–	

[a] Synthetic total fraction for 1990-2019; dissolved fraction for 2001-2019

[b] Non-detect methods

[c] ND – No data

[d] NH_4 not analyzed by standard methods for 2001-2019 because >15% below MDL

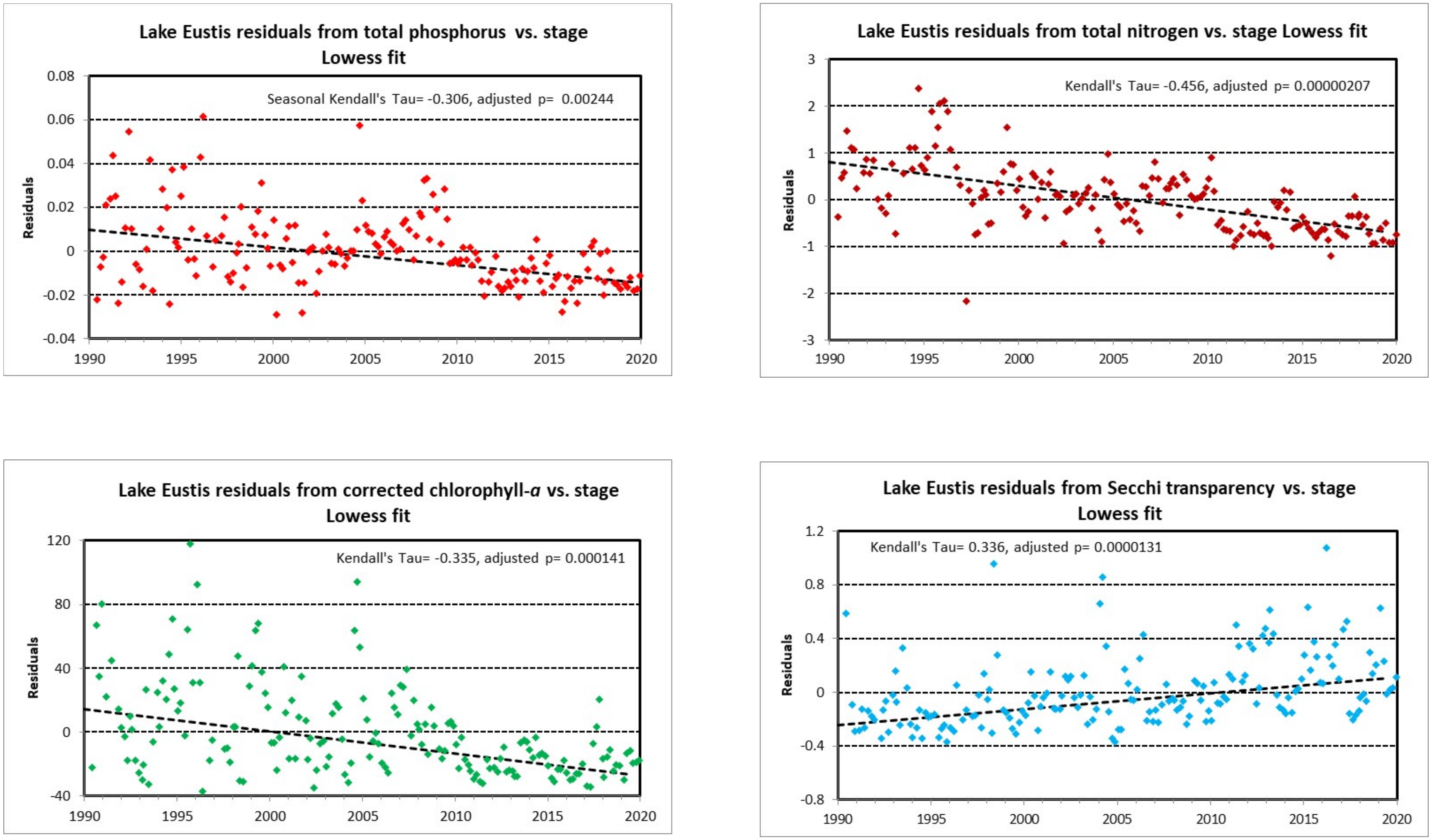


Figure 52. Water quality trends in Lake Eustis, 1990-2019.

Daily means and Theil-Sen trend lines.

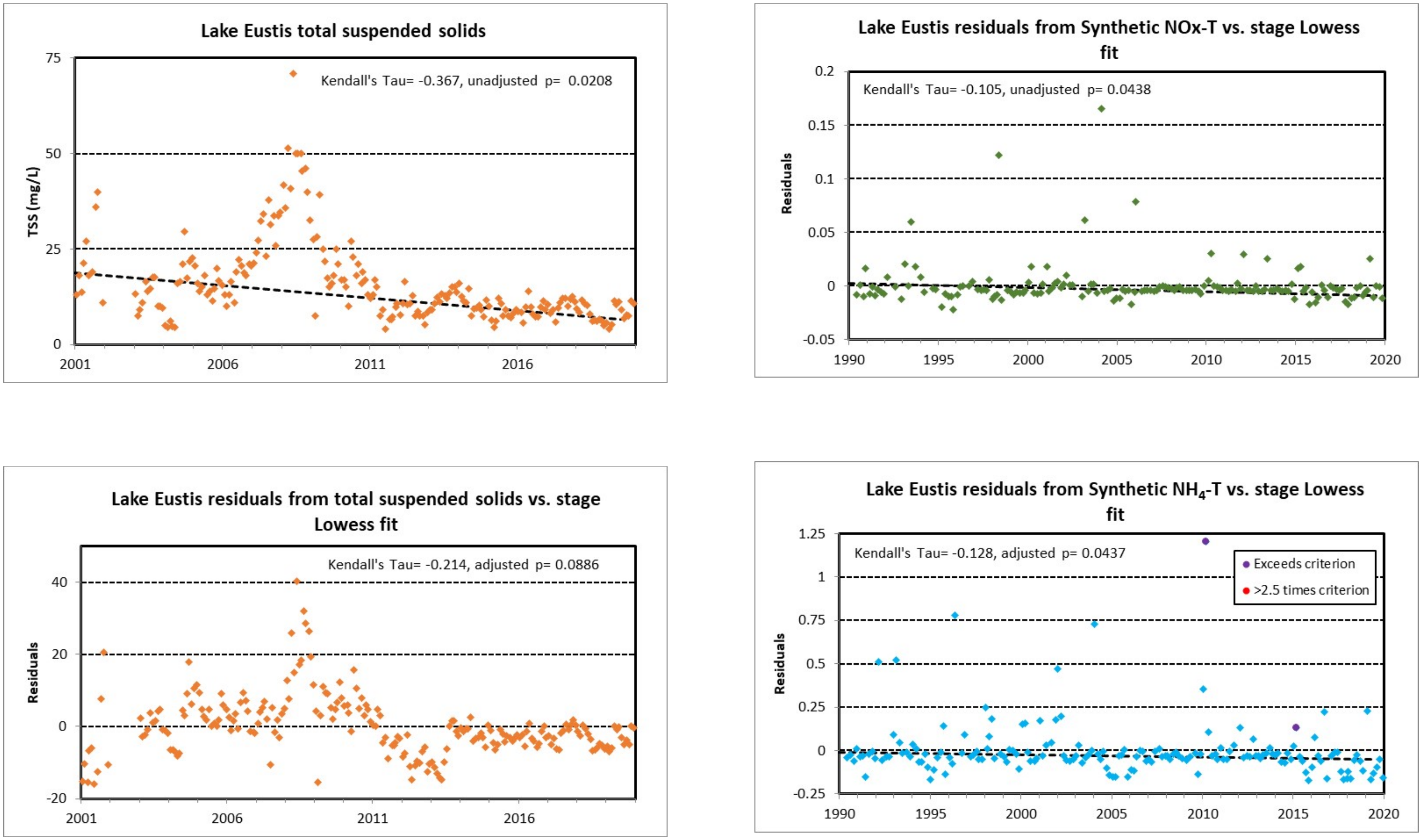


Figure 53. Water quality trends in Lake Eustis, 1990-2019 and 2001-2019.

Daily means and Theil-Sen trend lines.

Lake Eustis showed improving trends for a several water quality variables. For the 1990-2019 time period, there were decreasing trends for both concentrations and masses of TP, TN, chlorophyll-*a*, and TSS, and an increasing trend in Secchi transparency. For 2001-2019, TP, TN, and chlorophyll-*a* concentrations and masses showed decreasing trends, and there was an increasing trend in Secchi transparency. Over the 2001-2019 time period, unadjusted TSS concentrations showed an apparent improving trend, as did TSS masses, but there was no significant trend in residuals from the Lowess smooth – this indicates that the apparent improving trends may have been due to changes in water elevations over this period. There was a significant trend in Synthetic NO_x-T residuals from the Lowess smooth but not in unadjusted Synthetic NO_x-T over the 1990-2019 period. There was a significant trend in unadjusted PO_4-D over the 2001-2019 period, but not in residuals from Lowess smooth. Synthetic NH_4-T showed decreasing trends in residuals from the Lowess smooth over the 1990-2019 period with both standard and non-detect methods, but unadjusted concentrations showed a significant trend only using non-detect methods. There were a few exceedances of the Florida ammonia standard, in 2010 and 2015 (Figure 53).

The results of the trend analyses for Lake Eustis were very similar to the previous trend assessment (Fulton 2018), and generally the Kendall's tau coefficients were also similar. This appears to reflect stable water quality since 2016.

The phytoplankton monitoring in Lake Eustis ended in August 2018. Lake Eustis phytoplankton have generally been dominated by cyanobacteria although diatoms have been more significant than in Beauclair and Dora, particularly during the cool season (Figure 55, Figure 55). Trends in phytoplankton biovolumes in Lake Eustis were similar to those in Lake Harris, with a general decrease from 2007-2012, and a slight increase in 2013-2014. However, there was a larger increase in phytoplankton biovolumes in 2017-2018, which seemed inconsistent with a relatively small increase in chlorophyll-*a*. Again, *Cylindrospermopsis* tended to be the dominant cyanobacterial genus in Lake Eustis. As with lakes Beauclair and Dora, *Pseudanabaena* was prominent in the Phycotech counts (1998-2007), but not in those by BSA (2007-2018) (Figure 55). *Microcystis* and *Planktolyngbya* were more prominent in the BSA analyses than in those by Phycotech.

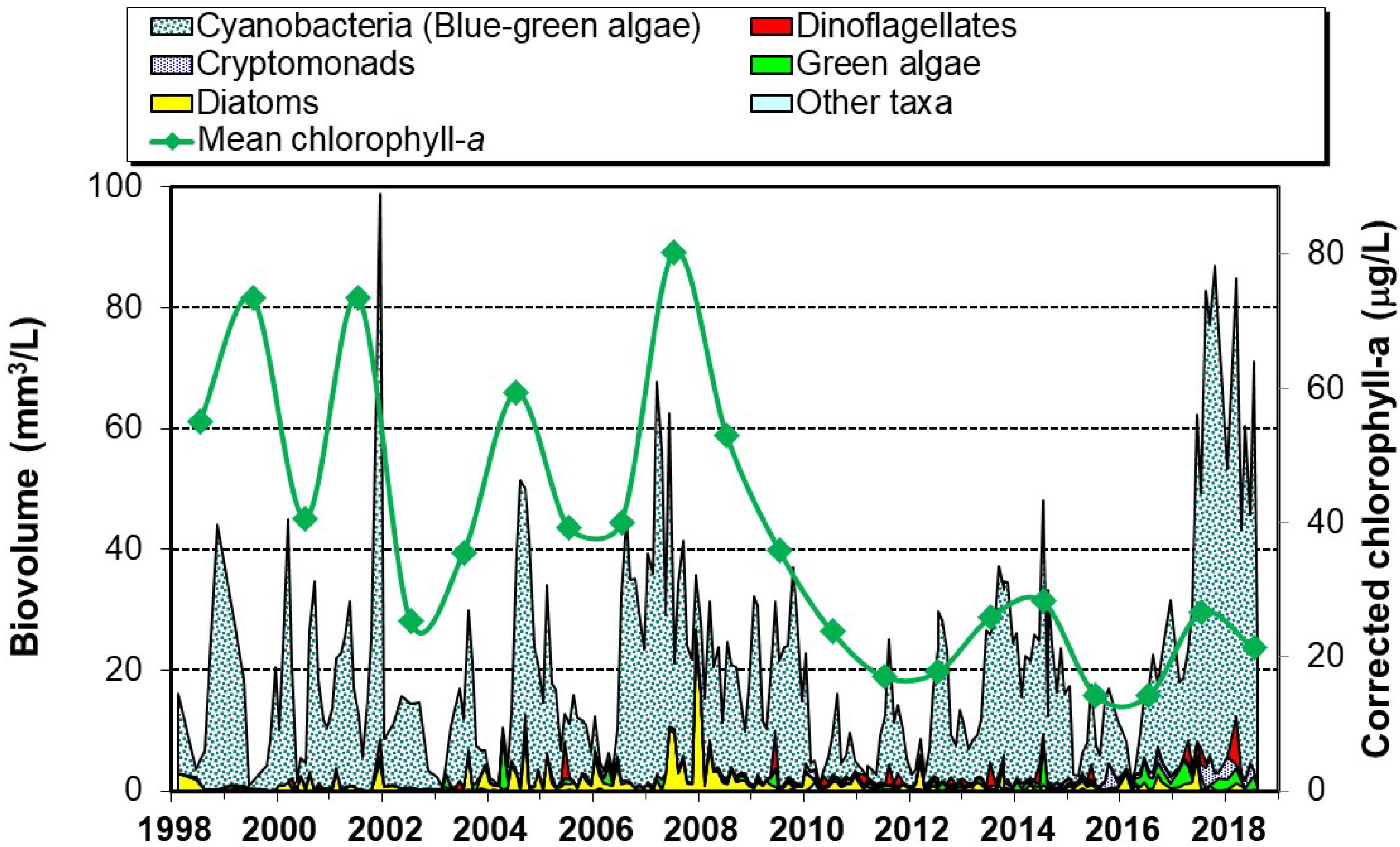


Figure 54. Lake Eustis phytoplankton biovolumes and chlorophyll-*a* concentrations.

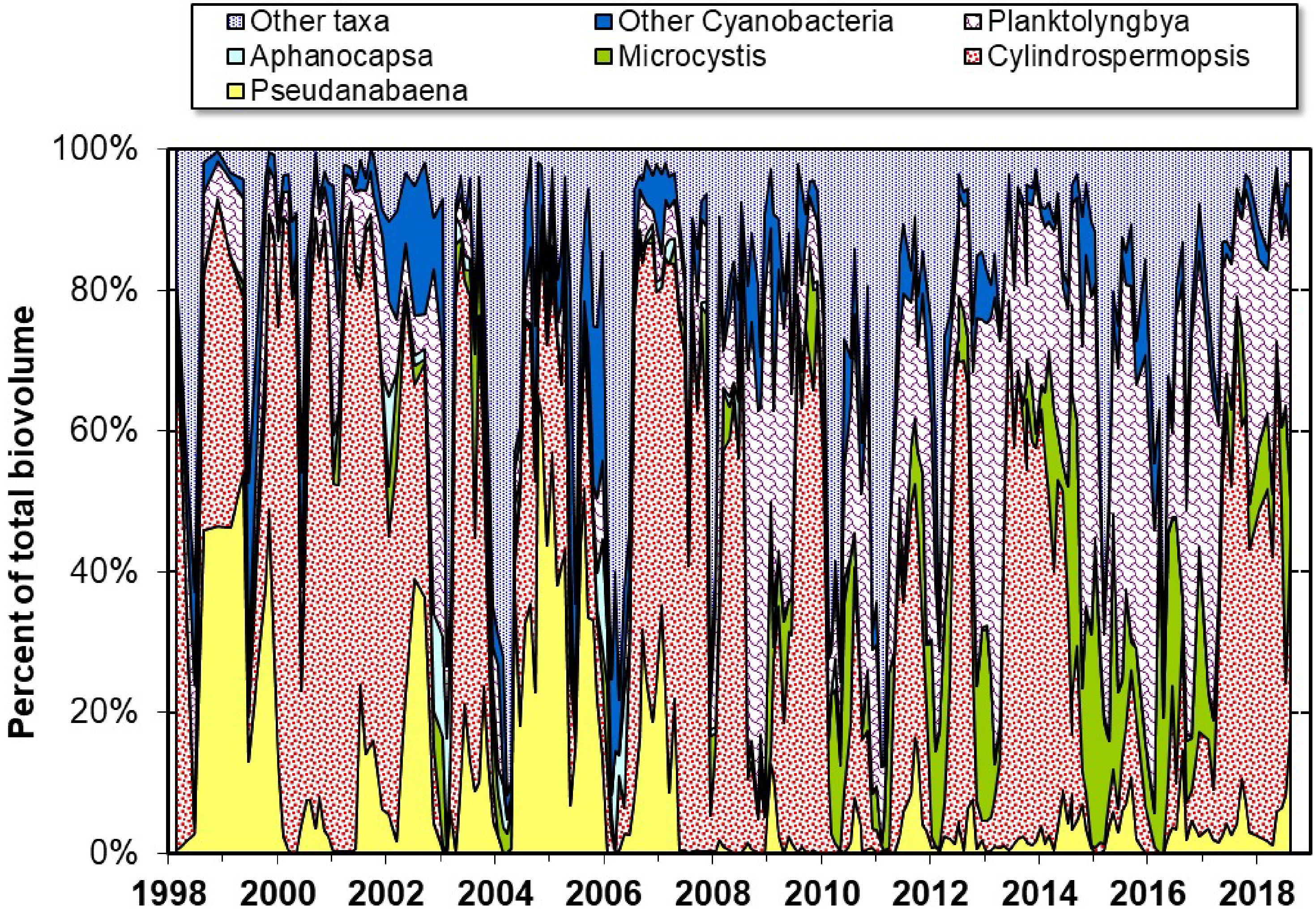


Figure 55. Lake Eustis phytoplankton percent composition.

LAKE GRIFFIN

Estimated external TP loads to Lake Griffin have declined substantially since the acquisition and restoration of muck farms in the Emeralda Marsh Conservation Area (Figure 56, Appendix Table D- 11). Estimated TP loading to Lake Griffin from the operating farms during the period 1984-1993 averaged about 25.5 metric tons/year, approximately 64% of the total external load of about 40 metric tons/year. Over the last 5 years, external TP load to the lake has averaged 13.6 MT/year, of which only about 0.3 MT/year (about 2.4% of the total) has come from the EMCA. The other major external TP source to the lake has been tributary discharges, primarily from Lake Eustis, which have varied substantially depending primarily on the volume of water discharged from upstream in the basin. As with the upstream lakes, external TP loading to Lake Griffin has increased the last 3 years, exceeding the TMDL (12.2 MT/year), due to greater tributary flows. TP loading to the lake was below the adopted TMDL for 10 of the previous 11 years.

Since 2008 discharges from Lake Harris have flowed through Harris Bayou Restoration Area (Lake Harris Conservation Area) into Lake Griffin. I partitioned the nutrient discharges from Harris Bayou to Lake Griffin into those that entered Harris Bayou from Lake Harris and those that came from Harris Bayou. Over the last 5 years, the estimated Harris Bayou TP discharges have averaged 0.8 MT/year, about 6.2% of the total load. The major external TP sources to Lake Griffin for the last 5 years have been tributary discharges from Lake Eustis (41%), atmospheric deposition (22%), stormwater runoff from urban/residential areas (9%), septic tank effluents (8%), Harris Bayou discharges (6%), tributary discharges from Lake Harris (through Harris Bayou, 4%), and runoff from natural areas (4%).

The external TN load to Lake Griffin has also decreased substantially, due primarily to reductions in water flows from upstream (Figure 57, Appendix Table D- 12). The EMCA was always a minor source of TN to Lake Griffin. The major external TN sources to Lake Griffin for the last 5 years have been tributary discharges from Lake Eustis (64%), atmospheric deposition (12%), tributary discharges from Lake Harris (through Harris Bayou, 7%), septic tank effluents (5%), natural area runoff (4%), stormwater runoff from urban/residential areas (3%), and EMCA (3%).

TP and chlorophyll-*a* concentrations have substantially decreased in Lake Griffin (Figure 56). In the late 1990s, average annual concentrations reached 125 μg/L for TP and 300 μg/L for chlorophyll-*a*. In the last 5 years, annual average concentrations have been 34 μg TP/L and 30 μg chlorophyll-*a*/L. These concentrations are very close to the TMDL target TP concentration of 32 μg/L, and equal to the expected chlorophyll-*a* concentration of 30 μg/L at the TP target. These water quality improvements have been attributed primarily to the TP load reduction from the EMCA and secondarily to gizzard shad harvesting from the lake (Fulton et al. 2015). In the fall of 2018, there was a phytoplankton bloom in Lake Griffin, with chlorophyll-*a* concentration reaching 94 μg/L, although TP concentrations were stable during this period. Subsequently, chlorophyll-*a* concentrations have gradually decreased.

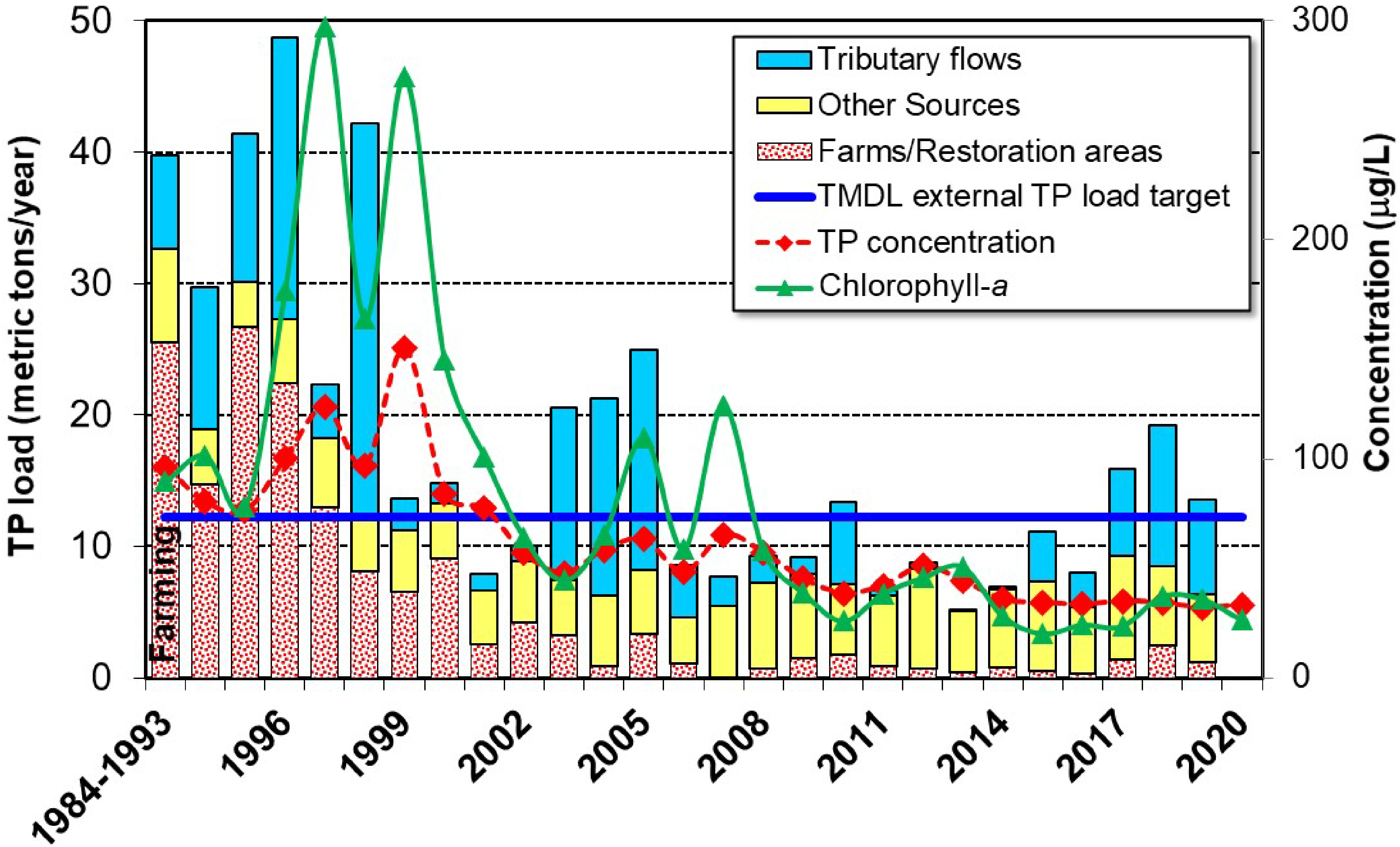


Figure 56. Lake Griffin estimated annual external total phosphorus loads (bars), TMDL loading target, and mean annual concentrations of TP and chlorophyll-*a* (lines).

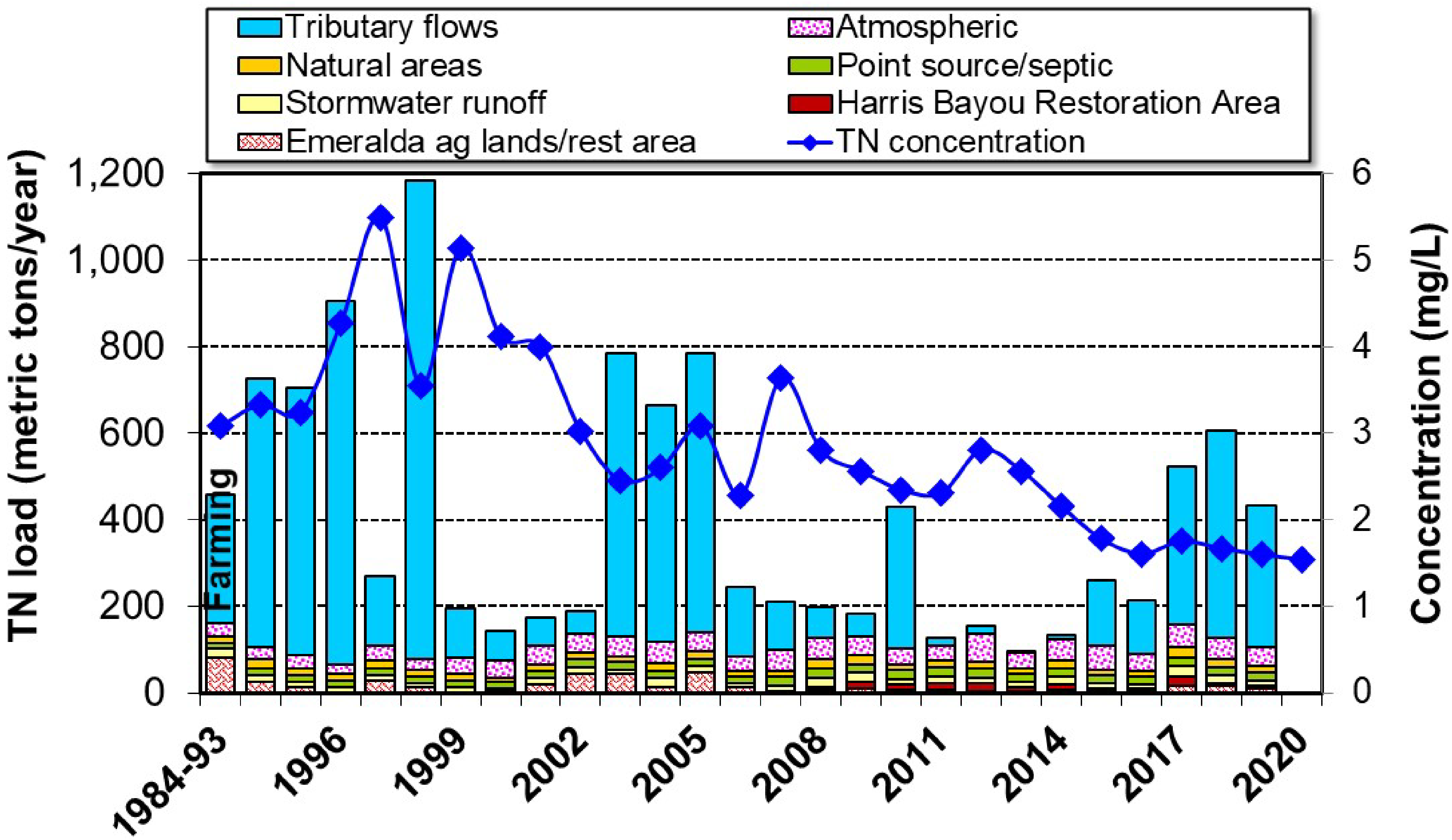


Figure 57. Lake Griffin estimated annual external total nitrogen loads (bars), and mean annual concentrations of TN.

TN concentrations have also decreased in Lake Griffin (Figure 57). Over the last 5 years, average TN concentration in Lake Griffin has been 1.6 mg/L. These decreases were attributed partially to a reduction in nitrogen fixation in the lake, which appears to be limited by phosphorus availability (Fulton et al. 2015). Trends in concentrations and masses of TP, TN, chlorophyll-*a*, and TSS are very similar in Lake Griffin (Figure 59, Figure 59). There was a sharp increase in TP concentrations and mass associated with a drawdown of the lake in 1984 (Figure 58). Unlike several of the other lakes, water quality in Lake Griffin tended to improve during the 2000-2002 drought; this may have been a response to the substantial decrease in external loading. There did seem to be some water quality deterioration during subsequent droughts.

Results of water quality trend analyses for Lake Griffin are summarized in Table 9, with more detailed results including Kendall’s tau nonparametric correlation coefficient and statistical significance level of the nonparametric Theil-Sen trend lines in Appendix Table B-6. Time series plots are shown for several parameters in Figure 60 and Figure 61.

Lake Griffin showed improving trends for several water quality variables. For both time periods, there were decreasing trends for both concentrations and masses of TP, TN, chlorophyll-*a*, and TSS. Secchi transparency increased in both time periods. PO_4-D showed a decreasing trend over the 2001-2019 time period, but not with the residuals from the Lowess smooth – this indicates that the apparent improving trend may have been due to changes in water elevations over this period. The plots for PO_4-D and residuals appeared very similar (Figure 61); the statistical analysis may be suspect since over 92% of the PO_4-D measurements were below MDL. NH_4 showed decreasing trends in both time periods when analyzed using non-detect methods, but there was not a significant trend for the 1983-2019 period using standard methods, in which the p-values were corrected for serial correlation. There were many exceedances of the Florida ammonia standard (Figure 61).

The results of the trend analyses for Lake Griffin were very similar to the previous trend assessment (Fulton 2018), and generally the Kendall’s tau coefficients were also similar. This appears to reflect stable water quality since 2016.

The seasonal variability in NH_4 is much larger than the long-term temporal trend, with large increases in concentrations occurring in several winters (Figure 62, Figure 62). These high winter concentrations of NH_4 often resulted in exceedances of the Florida ammonia standard. Most of the other basin lakes showed occasional high concentrations of NH_4, usually in the winter-spring, but not as frequently as Lake Griffin. The high NH_4 concentrations in Lake Griffin tend to be associated with low winter minimum concentrations of chlorophyll, particulate nitrogen, and organic nitrogen, although not with large decreases in their concentrations from summer or fall peaks. Nor are they associated with low dissolved oxygen concentrations (typically measured at a depth of 0.5 m). The high NH_4 concentrations may be due to releases from decomposing organic matter in the lake sediments and low uptake rates because of low phytoplankton biomass.

Table 9. Water quality trends for Lake Griffin.

Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality, $p<0.05$

(0) No trend, $p\geq0.05$

(–) Significant decreasing trend, $p<0.05$

(+) Significant increasing trend, $p<0.05$

	1983 - 2019		**2001 - 2019**	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	–	–	S, –	S, –
TN	–	–	–	S, –
Chlorophyll-*a*	–	–	–	S, –
TSS	–	–	–	–
Secchi	+	+	+	+
NH_4 [a, d]	S, 0	0		
NH_4 [a, b]	S, –	S, –	S, –	S, –
NO_x [a, b]	S, 0	S, 0	S, 0	S, 0
PO_4 [a, b]	ND [c]	ND	–	0
TP mass	S, –		S, –	
TN mass	–		S, –	
Chlorophyll-*a* mass	–		–	
TSS mass	–		–	

[a] Synthetic total fraction for 1983-2019; dissolved fraction for 2001-2019

[b] Non-detect methods

[c] ND – No data

[d] NH_4 not analyzed by standard methods for 2001-2019 because >15% below MDL

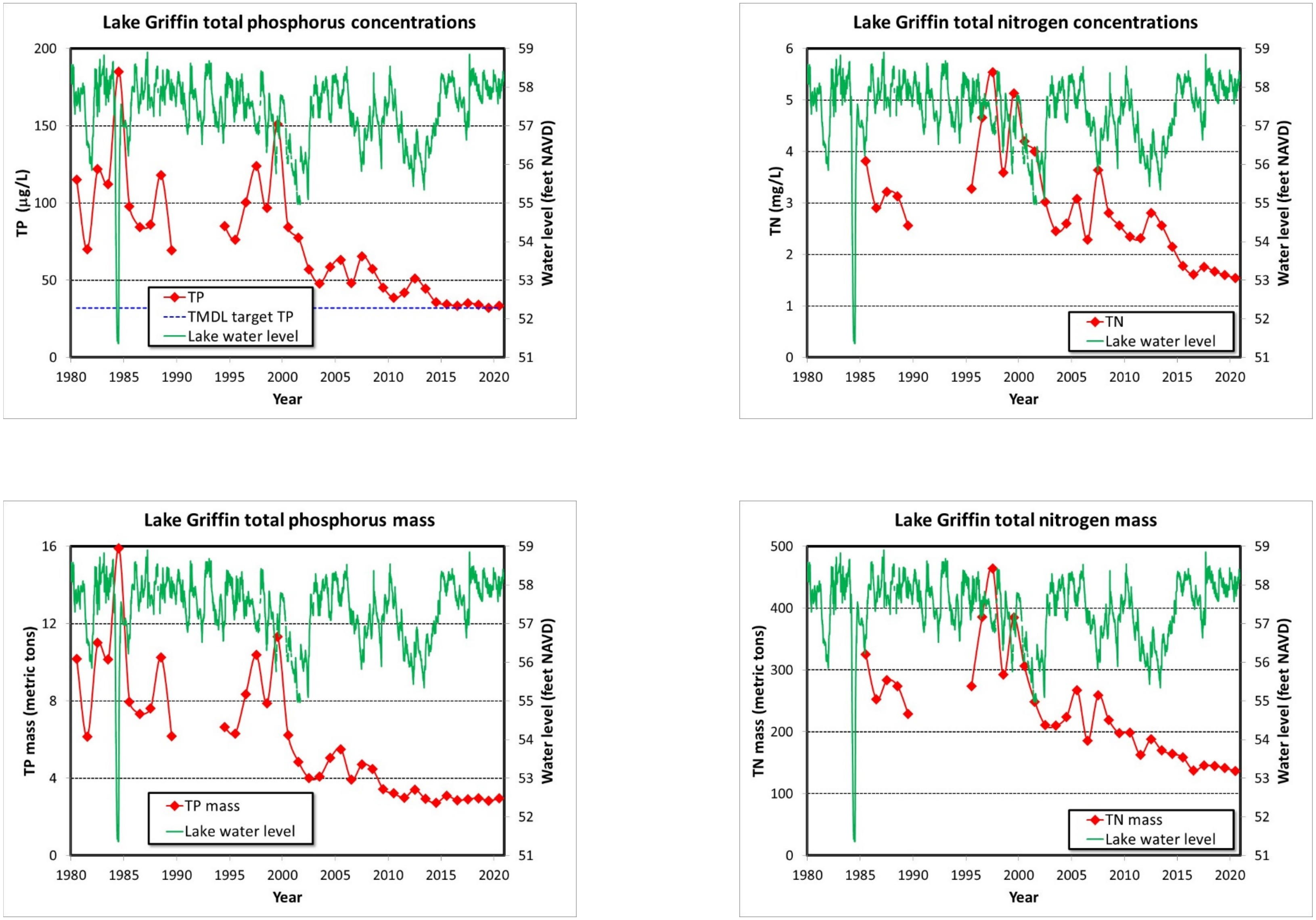


Figure 58. Lake Griffin mean annual total phosphorus and total nitrogen concentrations and masses, and water elevations.

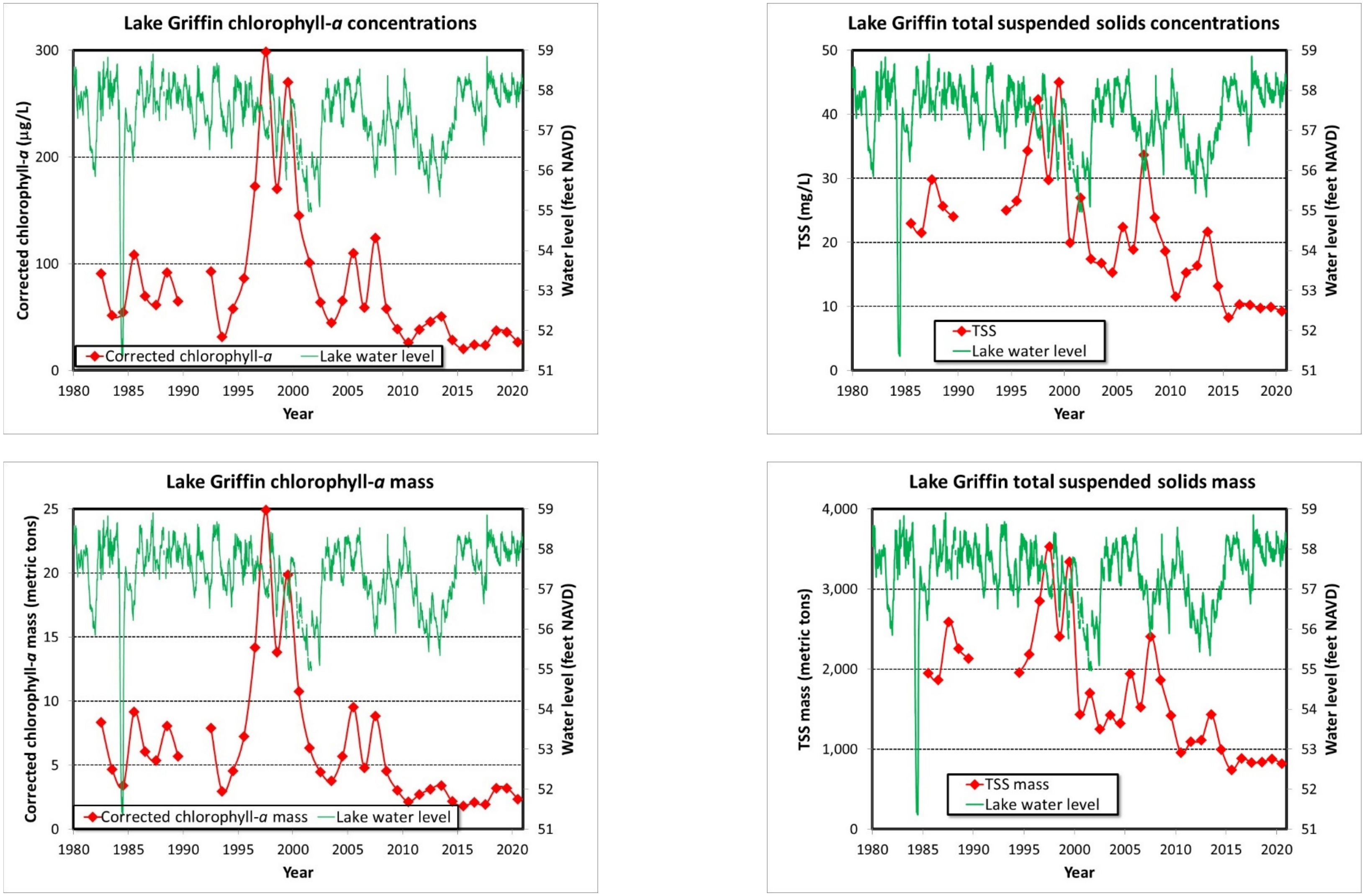


Figure 59. Lake Griffin mean annual total chlorophyll-*a* and total suspended solids concentrations and masses, and water elevations.

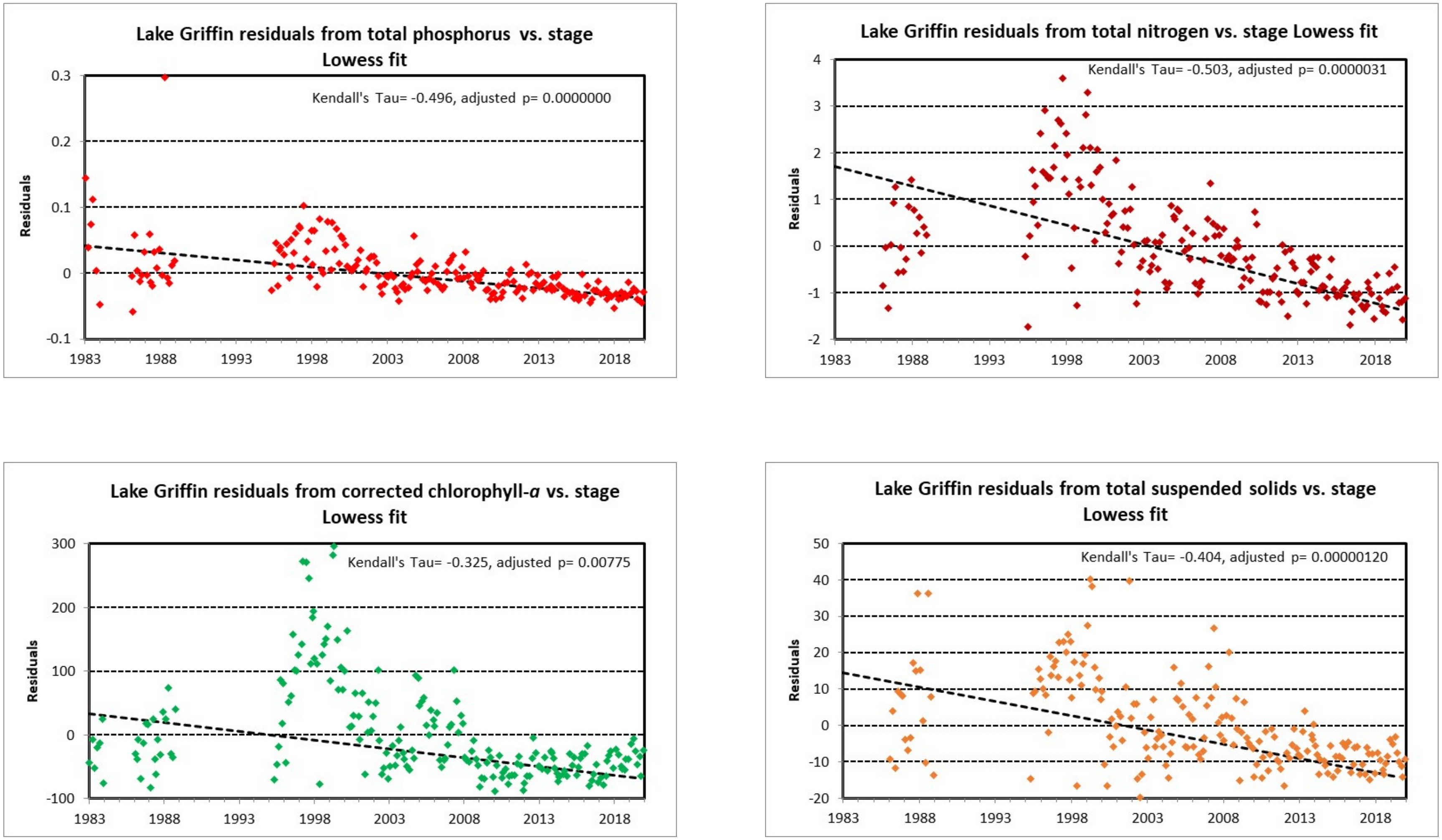


Figure 60. Water quality trends in Lake Griffin, 1983-2019.

Daily means and Theil-Sen trend lines.

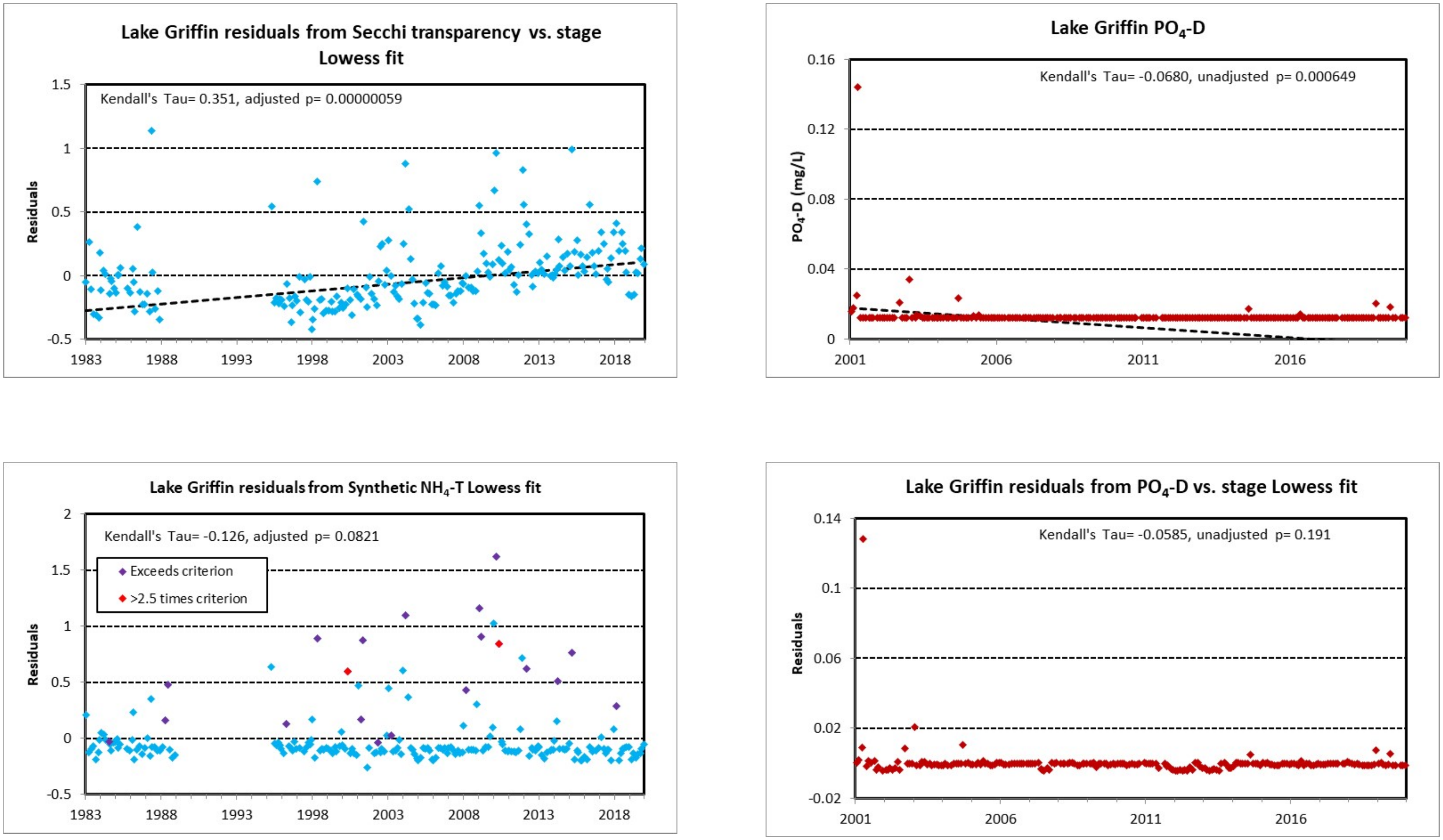


Figure 61. Water quality trends in Lake Griffin, 1983-2019 and 2001-2019.

Daily means and Theil-Sen or ATS trend lines.

Lake Griffin phytoplankton monitoring ended in August 2018. Lake Griffin chlorophyll-*a* and phytoplankton biovolumes substantially decreased from high levels in the late 1990s, although the biovolume decreases are smaller. The phytoplankton have generally been dominated by cyanobacteria although there have been some cool season periods of dominance by diatoms, particularly in recent years (Figure 64, Figure 64). Again, *Cylindrospermopsis* has tended to be the dominant cyanobacterial genus in Lake Griffin. As with several other lakes, *Pseudanabaena* was prominent in the Phycotech counts, particularly from 2001-2007, but not in those by BSA (2007-2018) (Figure 64). As noted previously, Phlips and Schelske (2004) reported both *Cylindrospermopsis* and *Oscillatoria* (alternate genus name *Pseudanabaena*) in Lake Griffin, noting they were very similar in appearance, with the main difference being the presence of heterocysts in *Cylindrospermopsis* and their absence in *Oscillatoria*. As with the other lakes, Phycotech may have identified filaments lacking heterocysts as *Pseudanabaena/Oscillatoria*, while BSA identified them as *Cylindrospermopsis*. Also, like several of the other lakes, *Microcystis* and *Planktolyngbya* were more prominent on the BSA analyses than in those by Phycotech.

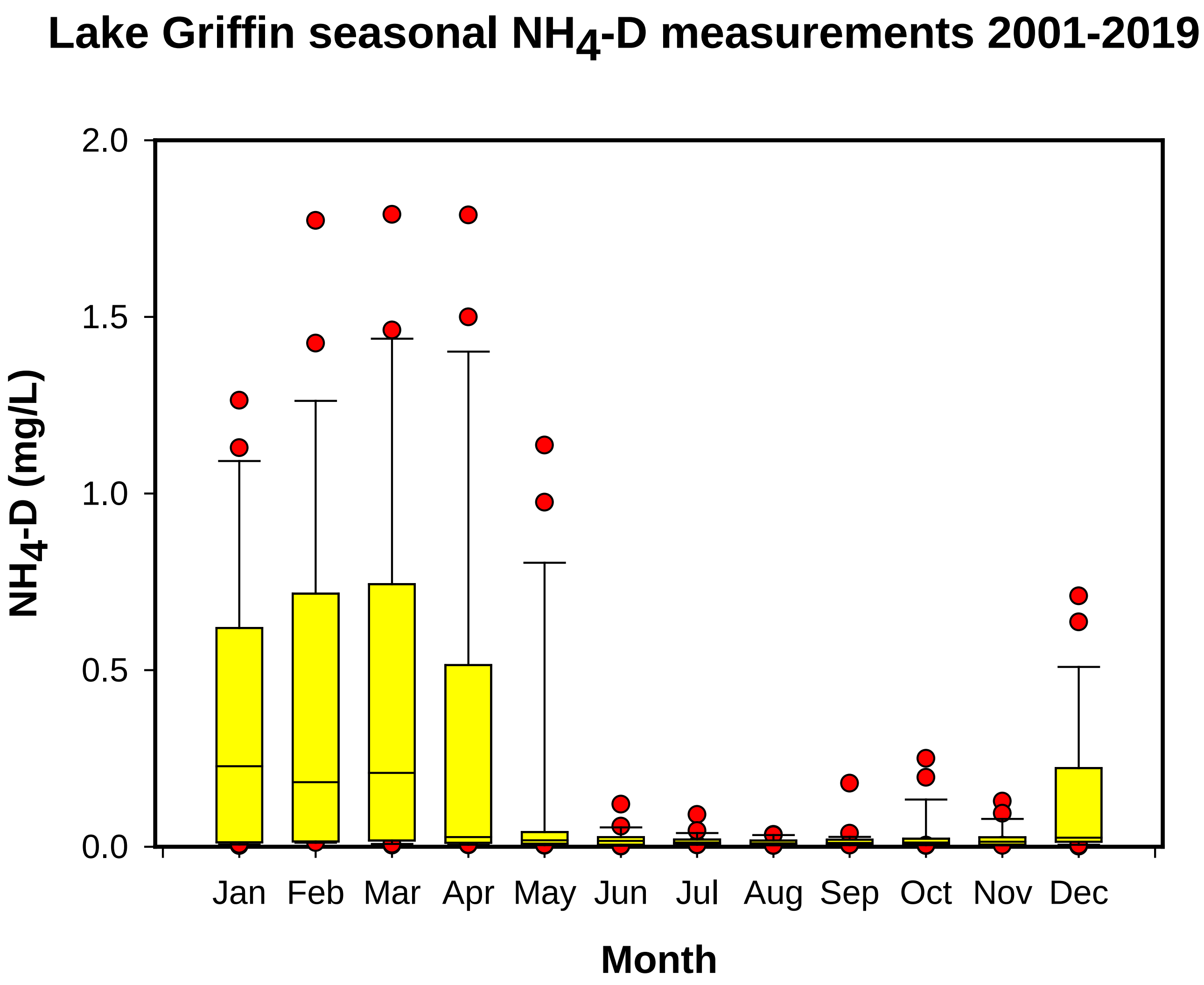


Figure 62. Seasonal NH_4-D measurements in Lake Griffin, 2001-2019.

Box shows median, 25th and 75th percentiles; whiskers above and below show 10th and 90th percentiles; red circles show extreme values.

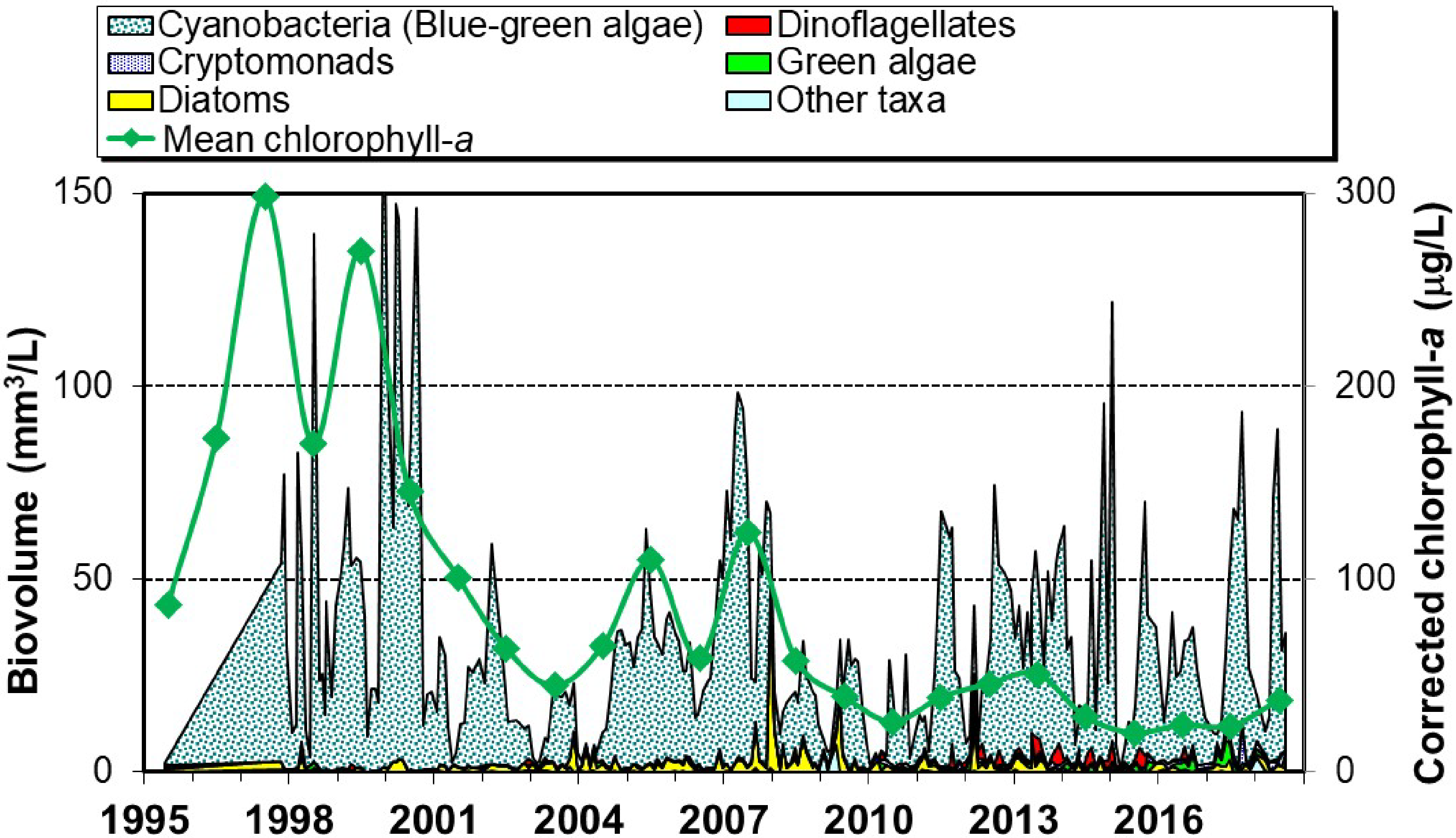


Figure 63. Lake Griffin phytoplankton biovolumes and chlorophyll-*a* concentrations.

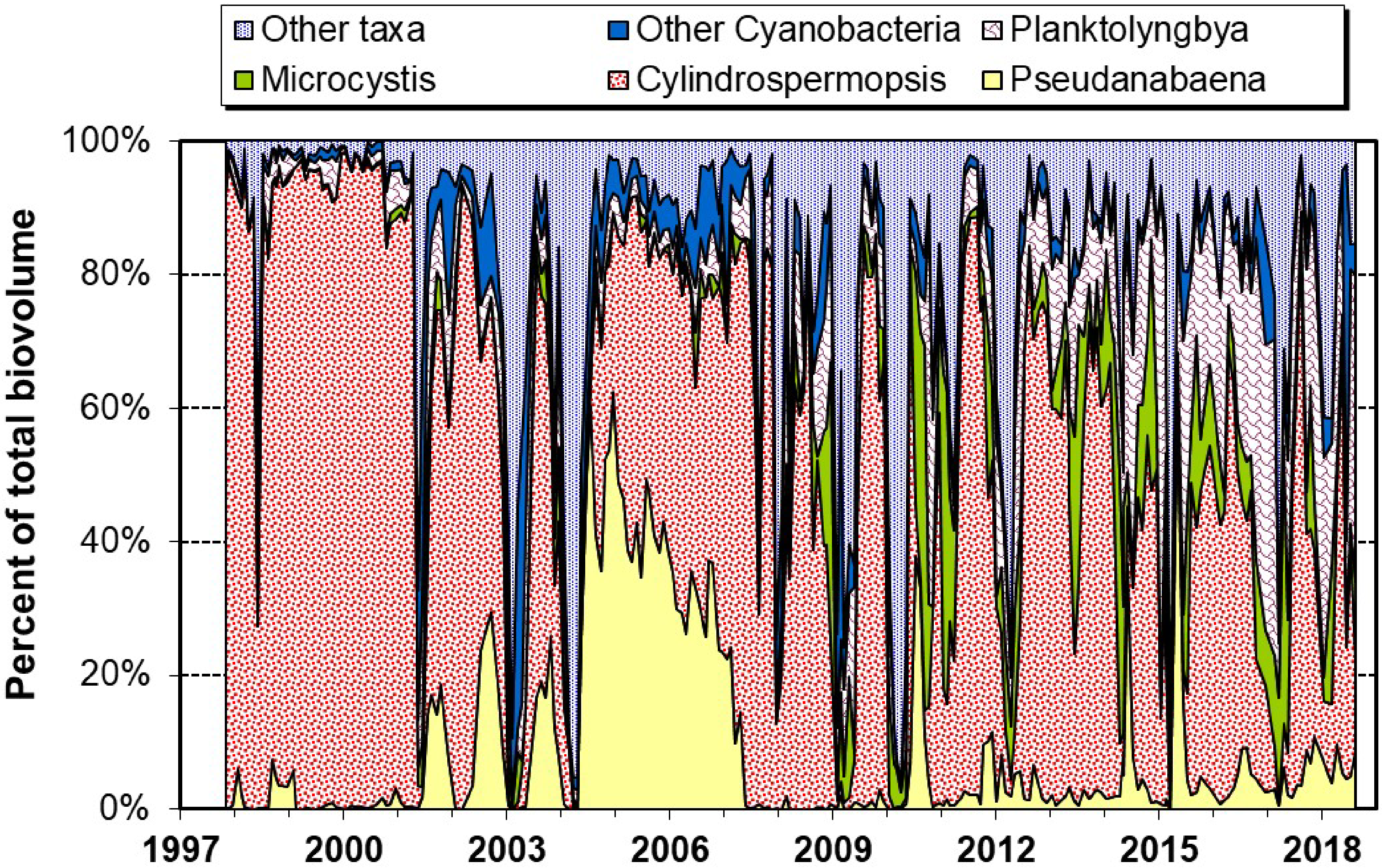


Figure 64. Lake Griffin phytoplankton percent composition.

LAKE YALE

Estimated external TP loads to Lake Yale are divided among several sources, most of which have not shown much temporal trend (Figure 65, Appendix Table D- 13). External TP loading to the lake has been below the adopted TMDL (1.29 MT/year) in only two years since 2004. The major external TP sources to Lake Yale for the last 5 years have been atmospheric deposition (35%), urban-residential runoff (22%), natural area runoff (16%), septic tanks (15%), and agriculture (12%). Failure of the culvert on the Yale-Griffin Canal in 2005 and low lake elevations in later years increased the potential for discharges from Lake Griffin to Lake Yale – there were small discharges in several years. The culvert was replaced in 2014.

External TN loading to Lake Yale has also not shown a clear trend (Figure 66, Appendix Table D- 14). The major external TN sources to Lake Yale for the last 5 years have been atmospheric deposition (35%), natural area runoff (29%), septic tanks (16%), urban-residential runoff (11%), and agriculture (7%).

A recent study (ERD 2017) developed an independent TP and TN budget for Lake Yale, which included estimates of groundwater seepage and internal recycling, but they did not measure sedimentation losses. They concluded that internal recycling accounted for 80% of the TP load to the lake. In my view, a balanced assessment of internal recycling should consider both nutrient releases from the sediments and sedimentation losses. Schelske et al. (2001) measured net nutrient accumulation rates in the sediments in dated cores from Lake Yale.

Lake Yale has not had the historic nutrient loading from muck farms or significant tributary flows like several of the other basin lakes. There was one point source (a citrus processing plant) that contributed significant loading to the lake, but those discharges ceased by the mid-1990s. There also have not been any restoration or major stormwater projects to reduce external loading to Lake Yale. Lake County Water Authority has been considering a whole lake alum treatment for Lake Yale, one of the recommendations of ERD (2017).

Lake Yale underwent a significant deterioration in water quality in the mid-1990s (Figure 66, Figure 66), but that appeared to be primarily due to vegetation management activities in the lake. Lake Yale was treated with herbicides and stocked with grass carp to control *Hydrilla* in the early 1990s. Water quality appeared to deteriorate following the loss of most aquatic vegetation in the lake. In the last 5 years, average concentrations have been 31 μg TP/L and 16 μg chlorophyll-*a*/L. These concentrations exceed the TMDL target TP concentration of 20 μg/L, but only slightly exceed the expected chlorophyll-*a* concentration of 14 μg/L at the TP target. Average TN concentration over the last 5 years was 1.6 mg/L. Chlorophyll-*a*, TN, and perhaps TP concentrations appear to have been decreasing over the last 4 years (one 2020 TP measurement was unusually high, 76 μg/L; the average of the other measurements was 27 μg/L). In recent years there has been an increase in aquatic vegetation, including *Hydrilla*, for which there have been herbicide treatments in the lake. Concentrations and masses of TP, TN, chlorophyll-*a*, and TSS tend to show generally parallel trends, both tending to deteriorate during drought periods, although there appears to be a tendency

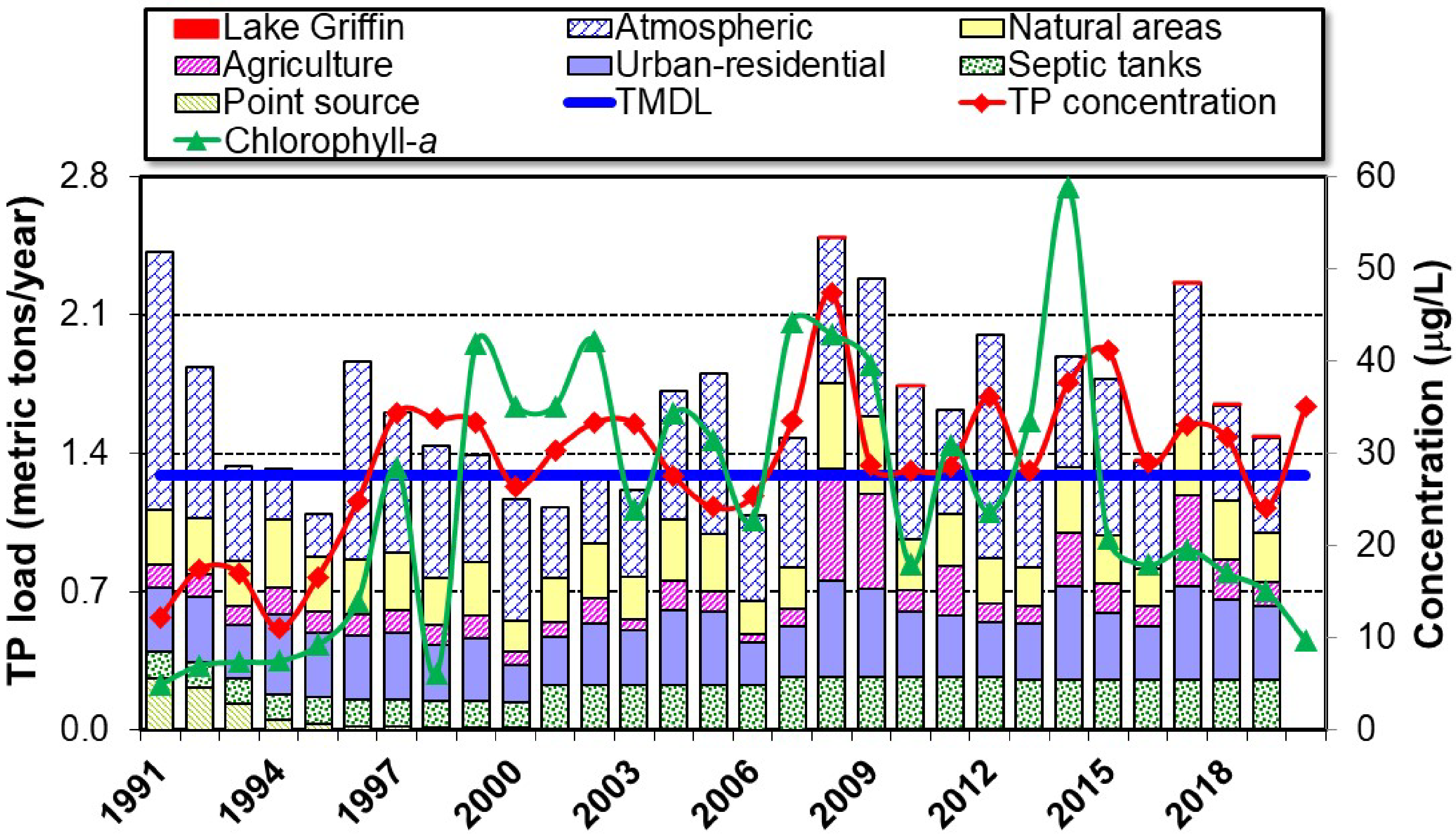


Figure 65. Lake Yale estimated annual external total phosphorus loads (bars), TMDL loading target, and mean annual concentrations of TP and chlorophyll-*a* (lines).

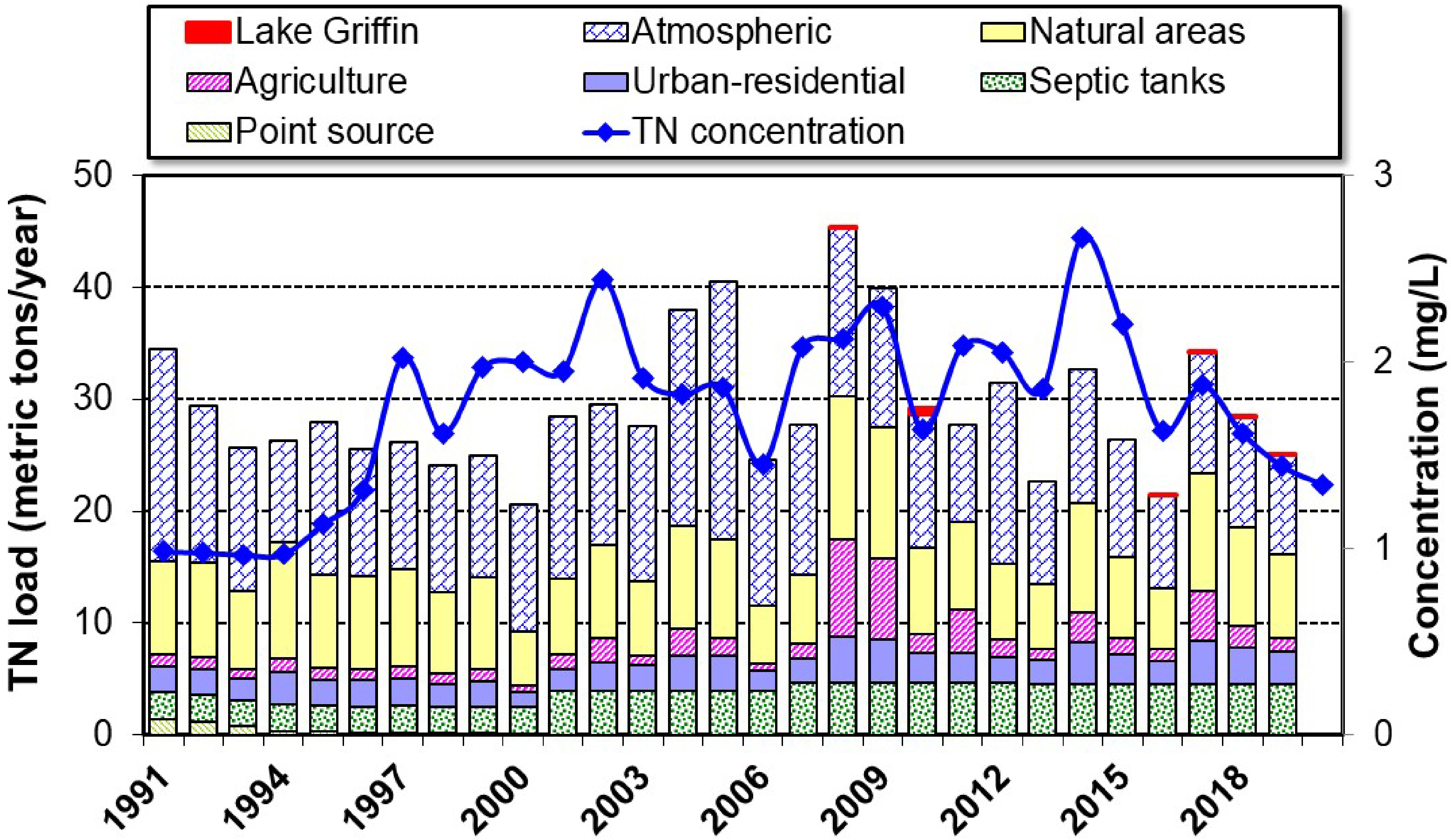


Figure 66. Lake Yale estimated annual external total nitrogen loads (bars), and mean annual concentrations of TN.

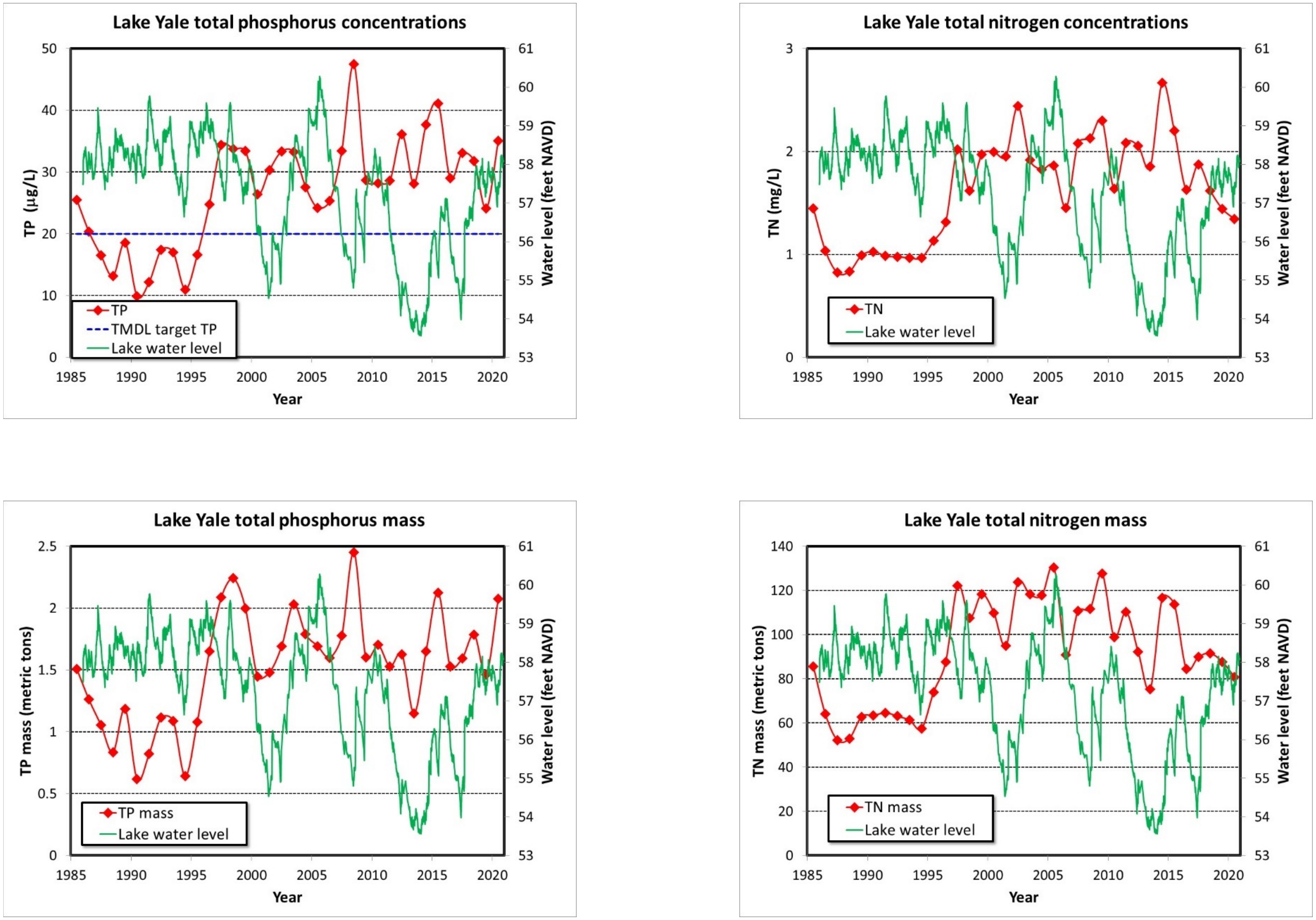


Figure 67. Lake Yale, mean annual total phosphorus and total nitrogen concentrations and masses, and water elevations.

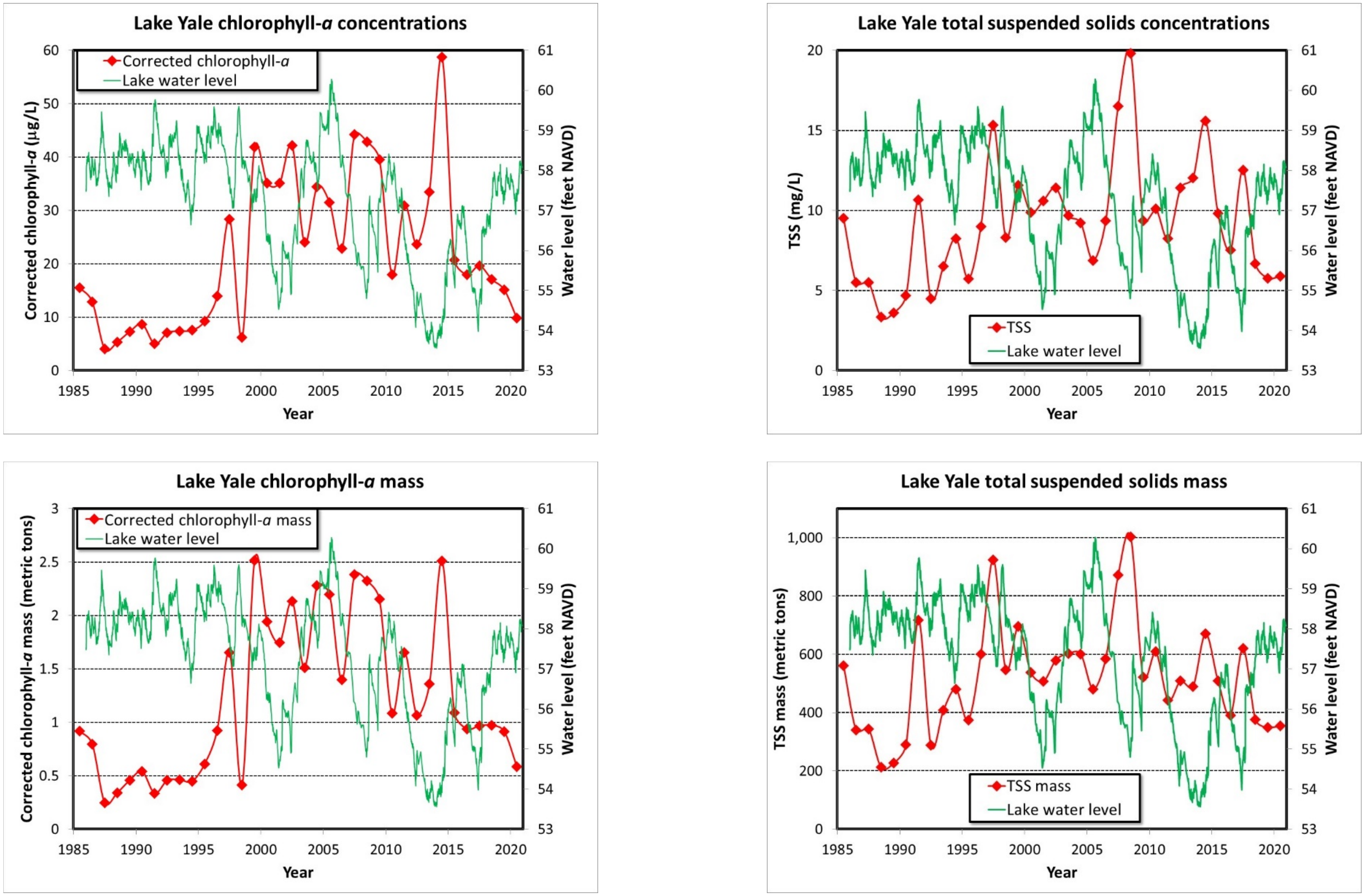


Figure 68. Lake Yale, mean annual chlorophyll-*a* and total suspended solids concentrations and masses, and water elevations.

towards greater increases in concentrations (Figure 68, Figure 68). These plots also indicate that water quality improvements over the last 4 years may be related to water level increases, as concentrations have decreased more than masses.

Results of water quality trend analyses for Lake Yale are summarized in Table 10, with more detailed results including Kendall's tau nonparametric correlation coefficient and statistical significance level of the nonparametric Theil-Sen trend lines in Appendix Table B- 7. Time series plots are shown for several parameters in Figure 69 and Figure 70.

Table 10. Water quality trends for Lake Yale.

Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality, $p<0.05$

(0) No trend, $p\geq0.05$

(–) Significant decreasing trend, $p<0.05$

(+) Significant increasing trend, $p<0.05$

	1986 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	+	S, 0	0	0
TP [b]	+	S, +		
TN	+	S, 0	0	0
Chlorophyll-*a*	+	S, 0	0	–
TSS	+	0	0	0
TSS [b]	+	0		
Secchi	–	S, 0	0	0
NH_4 [a, b]	S, 0	0	0	0
NO_x [a, b]	S, 0	S, 0	0	S, 0
PO_4 [a, b]	ND [c]	ND	–	0
TP mass	S, +		0	
TN mass	S, +		–	
Chlorophyll-*a* mass	S, +		0	
TSS mass	S, 0		0	

[a] Synthetic total fraction for 1986-2019; dissolved fraction for 2001-2019

[b] Non-detect methods

[c] ND – No data

[d] TP and TSS not analyzed by non-detect methods for 2001-2019 because <5% below MDL

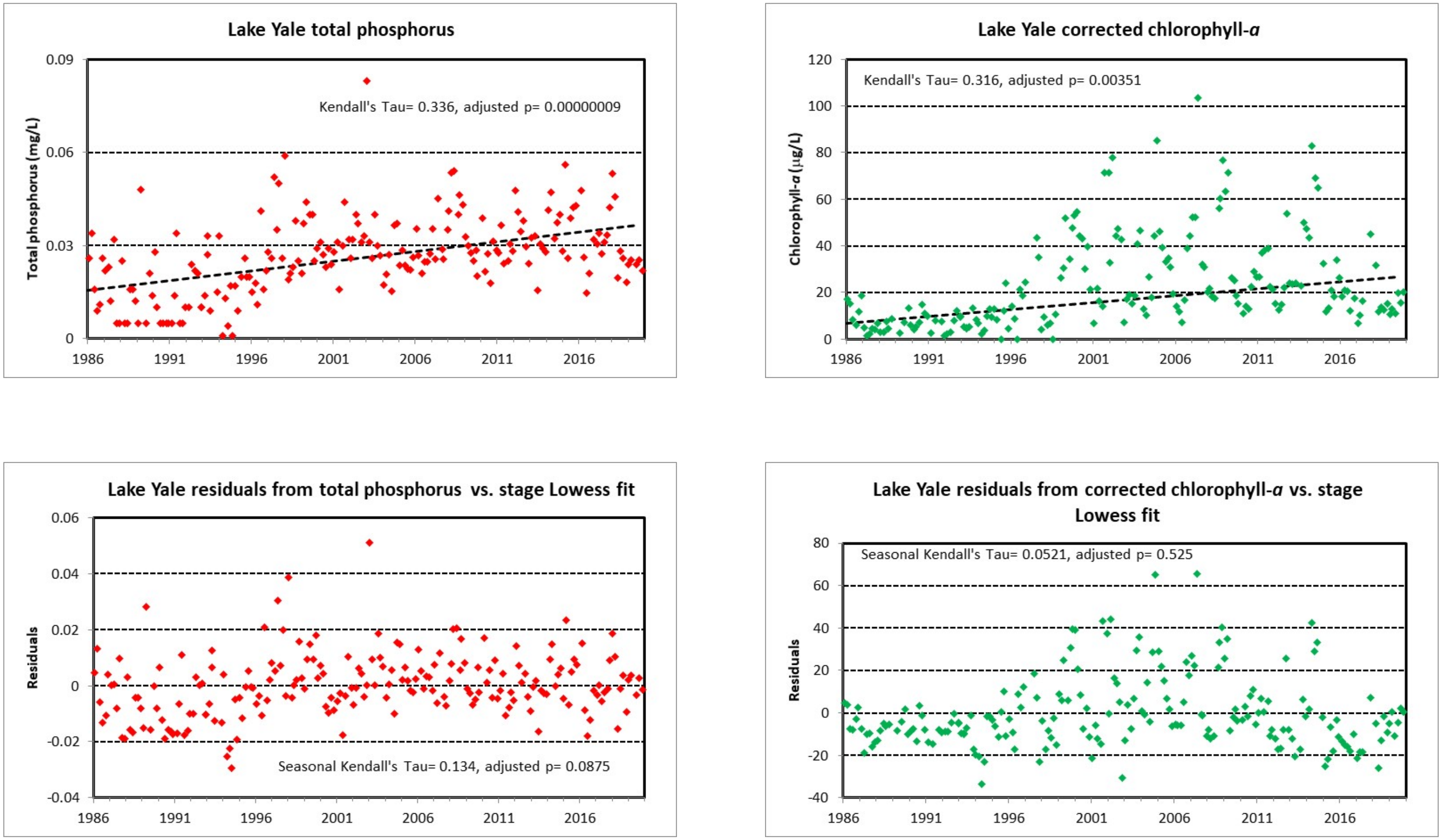


Figure 69. Water quality trends in Lake Yale, 1986-2019.

Daily means and Theil-Sen trend lines.

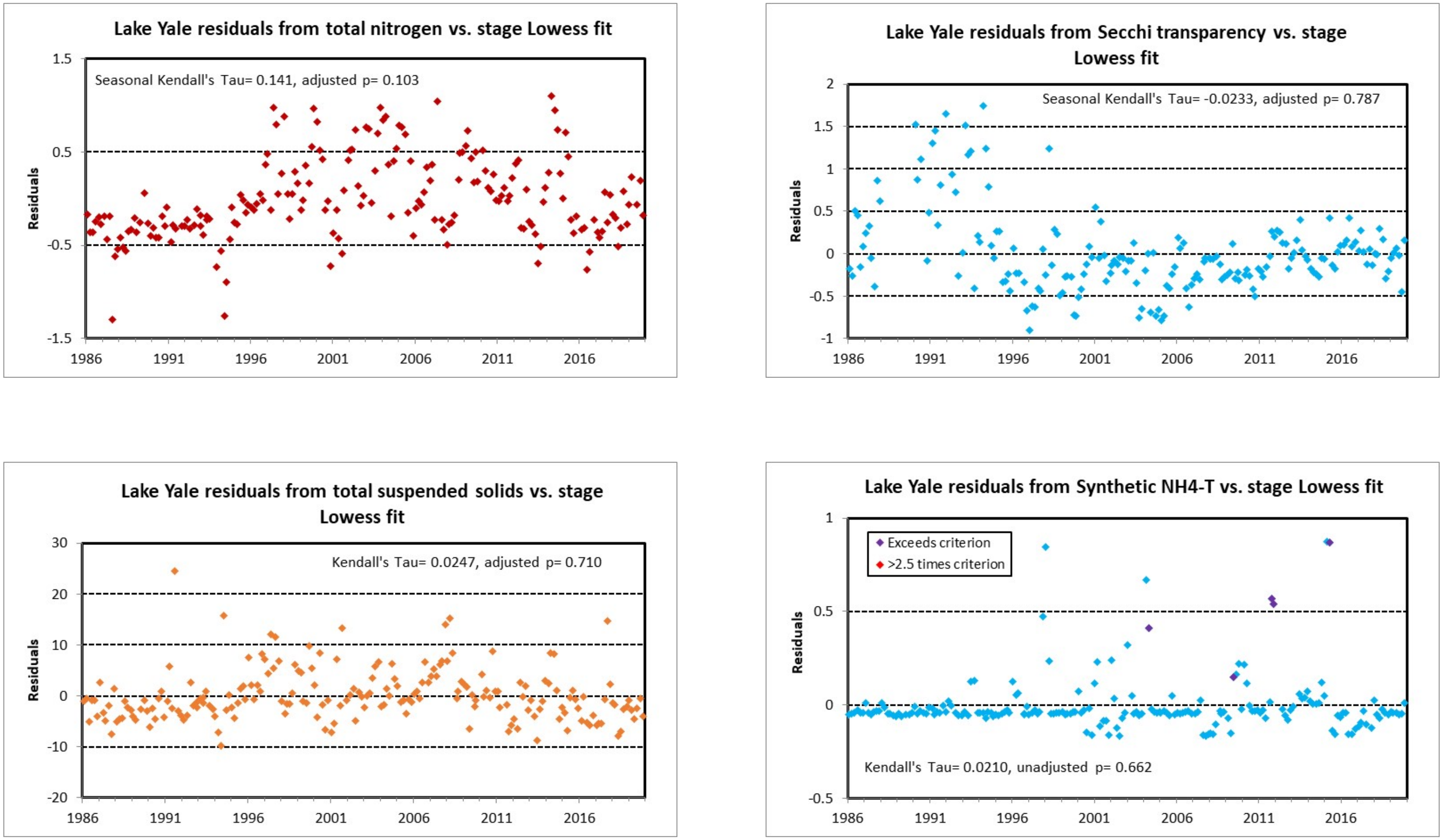


Figure 70. Water quality trends in Lake Yale, 1986-2019.

Daily means and Theil-Sen or ATS trend lines.

Over the 1989-2019 period, Lake Yale showed deteriorating trends for several unadjusted water quality variables, including increases in TP, TN, and chlorophyll-*a* concentrations and masses, TSS concentrations, and a decrease in Secchi transparency. However, there were few significant trends in water elevation-adjusted parameters, suggesting that the apparent deterioration was at least partly due to decreases in water levels. There was a significant increase in water elevation-adjusted TP analyzed using non-detect methods, which do not correct for serial correlation. The initial deterioration that occurred in the mid-1990s did not appear related to water levels, but the lake has undergone a series of severe droughts since 2000 which do appear to have degraded water quality (Figure 68, Figure 68). Over the 2001-2019 period, there were significant decreasing trends in chlorophyll-*a* water elevation-adjusted residuals and in TN mass, and several other water elevation-adjusted concentrations and masses showed weak improving trends, with p-values of around 0.1 (Appendix Table B-7). Unadjusted PO_4-D showed a decreasing trend over 2001-2019, but there was no trend in the water elevation-adjusted residuals. There were several exceedances of the Florida ammonia standard (Figure 70).

In the previous assessment (Fulton 2018), over the full period several of the water elevation-adjusted parameters showed deteriorating trends, and there were no improving trends over the 2001-2016 period. Those contrasts with the current assessment suggest that water quality has started to improve in the last few years.

Phytoplankton monitoring in Lake Yale stopped after July 2018. The phytoplankton in Lake Yale have generally been dominated by cyanobacteria although there have often been cool season periods of dominance by other taxa, usually by diatoms (Figure 72, Figure 72). There was a particularly large biovolume peak in November 2017 that was not mirrored in the chlorophyll-*a* data. Again, *Cylindrospermopsis* tended to be the dominant cyanobacterial genus in Lake Yale. Unlike several other lakes, *Pseudanabaena* was not prominent in the counts from either analyst (Figure 72). There have been some periods of significant representation by *Microcystis* in Lake Yale, usually in the cool season. There have been a few reports of severe, but short-lived, *Microcystis* blooms in Lake Yale, although no samples were collected during those blooms.

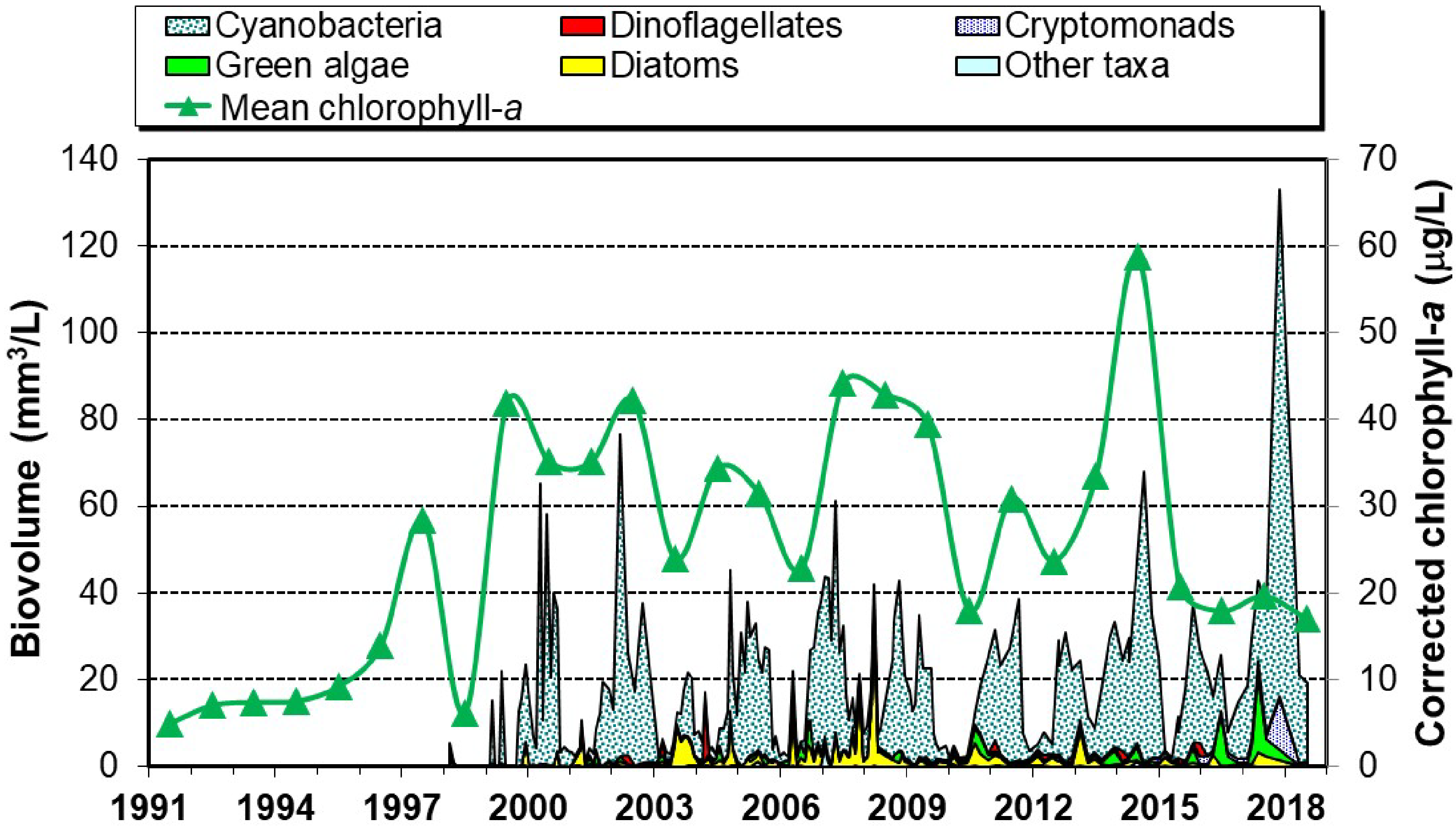


Figure 71. Lake Yale phytoplankton biovolumes and chlorophyll-*a* concentrations.

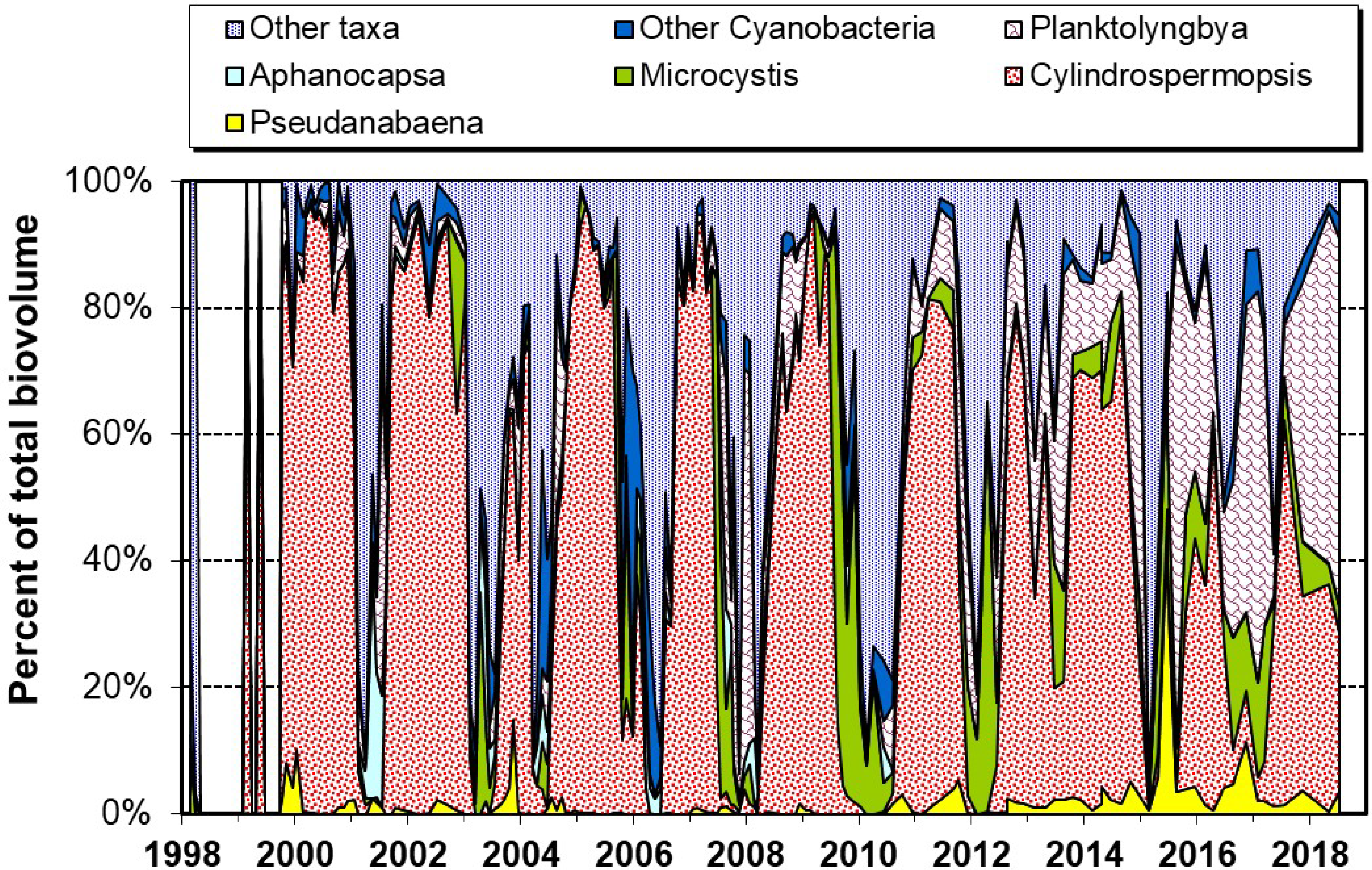


Figure 72. Lake Yale phytoplankton percent composition.

LAKE WEIR

Estimated external TP loads to Lake Weir are divided among several sources, most of which have not shown much temporal trend (Figure 73, Appendix Table D- 15). TP loading to the lake has been above the PLRG developed for the lake by SJRWMD (1.23 MT/year) for the last 13 years. FDEP adopted TMDLs for Lake Weir in 2017. The TMDLs are 1.667 MT TP/year and 27.432 MT TN/year (Rhew 2017a). Estimated external TP loading exceeded the TMDL in 6 of the last 13 years. However, our TP loading estimates are not directly comparable to the TMDL because our loading estimates for septic tanks are substantially lower than FDEP's (see next paragraph). If we used FDEP's septic tank loading estimate, then the TP TMDL would also have been exceeded the last 13 years. Estimated TN loading has exceeded the TMDL in 7 of the last 13 years (Figure 74, Appendix Table D- 16).

The methods used by SJRWMD and FDEP for estimating external nutrient loading were similar. The major difference was in estimated loads from septic tanks. As discussed previously, the SJRWMD estimates were based on counts of contributing septic systems in a 200-m zone around the lake and associated wetlands, with loading rates based on literature. FDEP estimated loads from counts of contributing septic systems in the entire watershed and used the ArcGIS-based Nitrate Load Estimation Toolkit (ArcNLET), developed by Florida State University (Rhew 2017a). ArcNLET currently simulates only nitrogen. For the Lake Weir TMDL, FDEP estimated septic tank TP loading by applying a TN:TP concentration ratio of 6.05:1. This ratio was calculated based on ground water TN and TP data collected from wells located in WBID 2790 (the Lake Weir outlet). Comparing the two septic tank load estimates, the SJRWMD estimates for recent years are 312 kg TP/year and 5,513 kg TN/year, while the FDEP estimates are 974 kg TP/year and 5,893 kg TN/year. So, the FDEP estimate is about 3.1 times higher for TP, and about 1.1 times higher for TN. FDEP's approach may erroneously assume that TP is as mobile in groundwater as TN, resulting in an overestimate of TP loading. However, septic tank nutrient loading to Lake Weir was estimated to have increased progressively from the 1980s to the 1980s (Crisman et al. 1992). Soils in the watershed may be nutrient saturated from long usage of septic tanks, so the SJRWMD methods could underestimate loading from this source.

The major external TP sources to Lake Weir (SJRWMD estimates, Figure 73) for the last 5 years have been atmospheric deposition (58%), septic tanks (21%), urban-residential runoff (10%), and natural area runoff (9%). The main reason for exceedance of the PLRG in recent years has been an increase in estimated atmospheric deposition, due to increases in TP concentrations in wet and dry deposition measured at wet-dry collector stations near Lake Apopka (these stations were used to estimate atmospheric deposition for all the basin lakes). A secondary reason for the increase in estimated TP loading to Lake Weir has been an increase in the number septic tanks around the lake.

The major external TN sources to Lake Weir (SJRWMD estimates, Figure 74) for the last 5 years have been atmospheric deposition (55%), septic tanks (22%), natural area runoff (15%), and urban-residential runoff (6%).

Lake Weir has not had the historic nutrient loading from muck farms or significant tributary flows like several of the other basin lakes. There also have not been any restoration or stormwater projects to reduce external loading to Lake Weir.

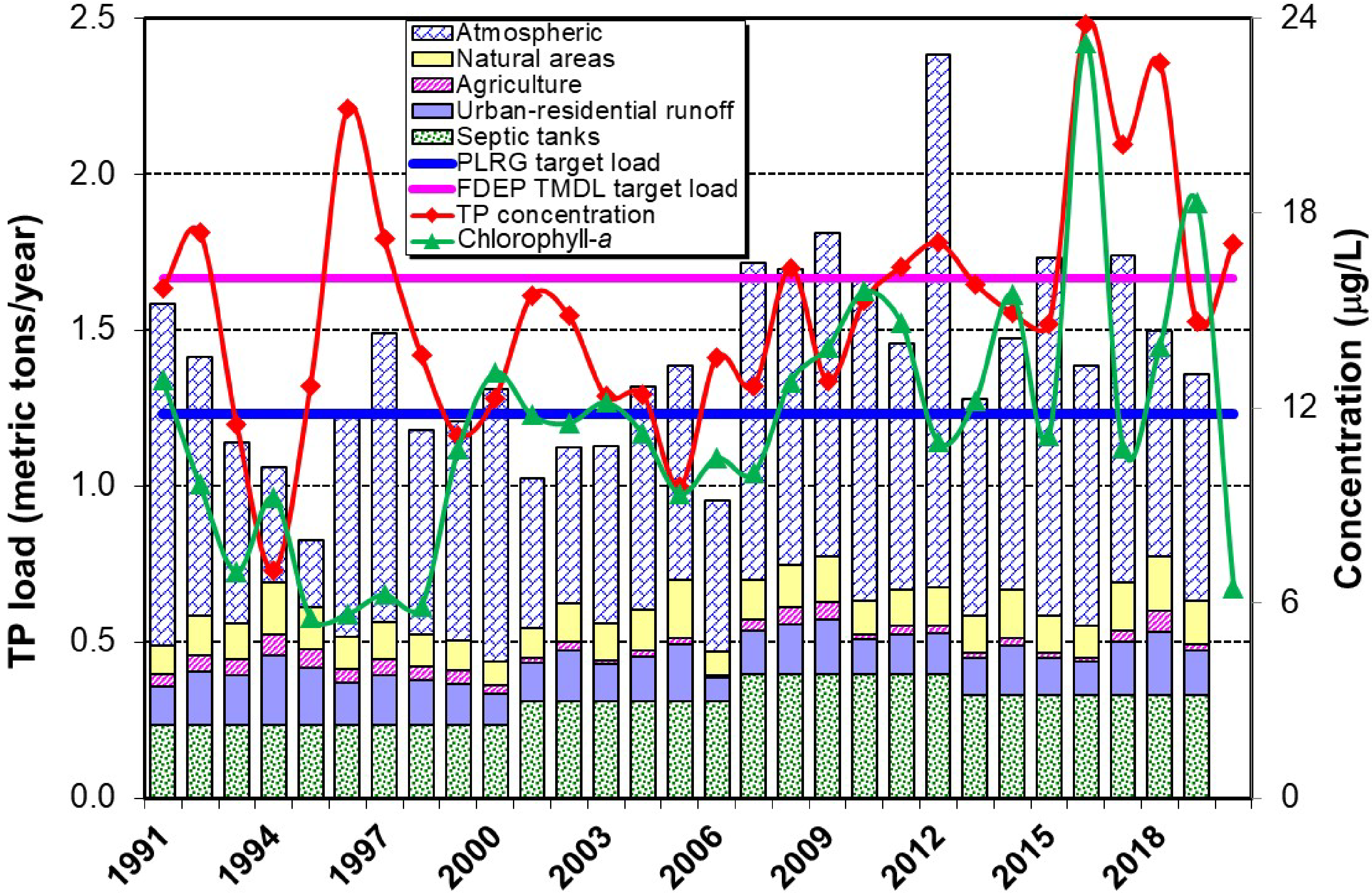


Figure 73. Lake Weir estimated annual external total phosphorus loads (bars), TMDL and PLRG loading targets, and mean annual concentrations of TP and chlorophyll-*a* (lines).

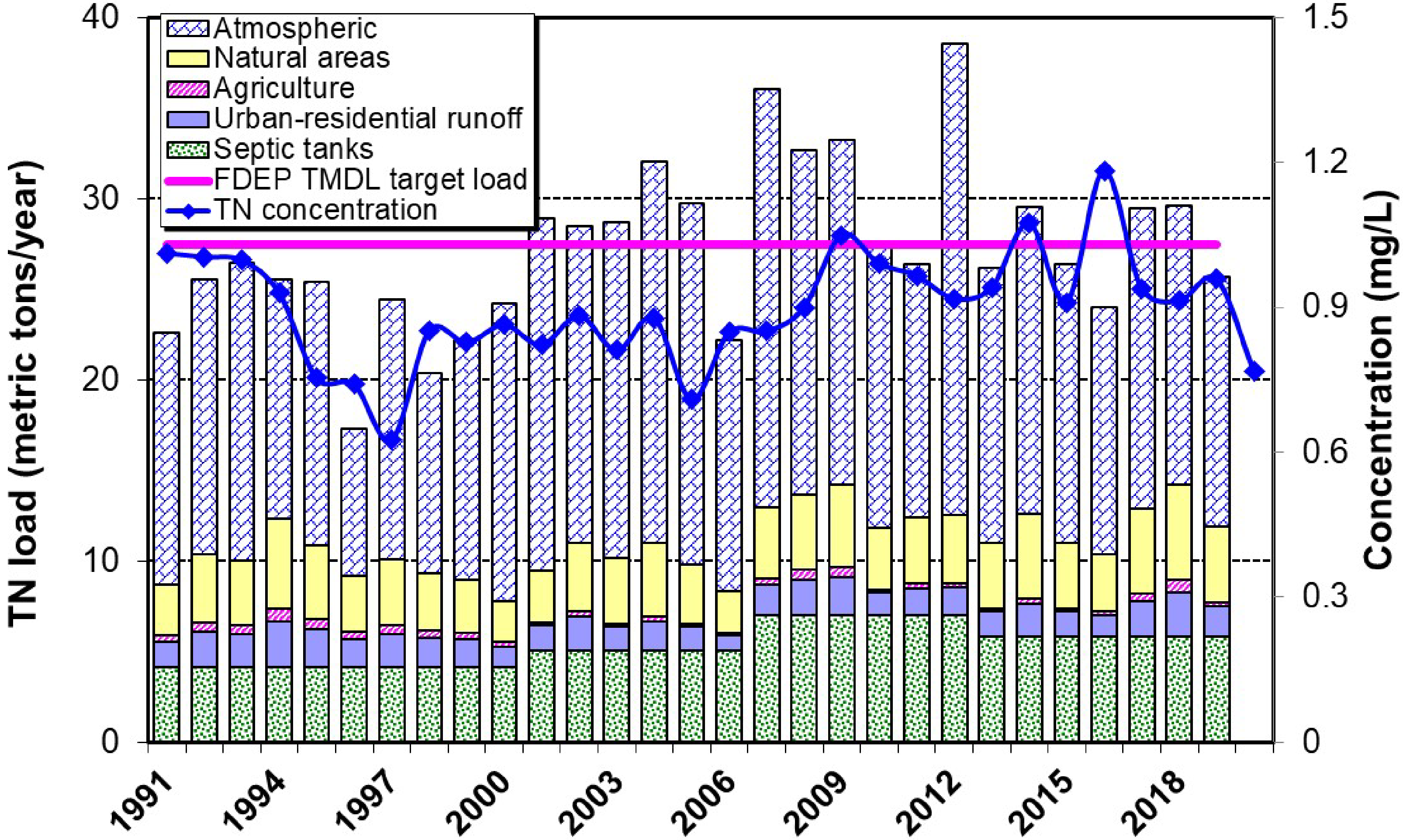


Figure 74. Lake Weir estimated annual external total nitrogen loads (bars), TMDL loading target and mean annual concentrations of TN (lines).

As discussed previously, area residents have been concerned that flocks of gulls roosting in Lake Weir may be contributing nutrients to the lake. In Christmas Bird Counts (CBC), the primary gull species occurring on Lake Weir has been the Ring-billed gull (*Larus delawarensis*). In the CBC, Ring-billed gull numbers ranged as high as 15,000; with several years in the range from 3-5 thousand. Reported excretion rates of different bird species (Portnoy 1990; Manny et al. 1994; Scherer et al. 1995; Fleming and Fraser 2001; Rip et al. 2006; Hahn et al. 2007) were plotted vs. bird weight (Figure 75). I excluded reported excretion rates for two species, a TP excretion rate for Great black-backed gulls which was a high outlier, and TP and TN excretion rates reported for Canada goose, because it would be expected that herbivorous geese would have different excretion rates than largely carnivorous gulls. For the other species, I developed linear regressions of excretion rate vs. gull weight and used the regressions to predict an excretion rate for a reported body weight for Ring-billed gulls (590 g, taken from http://identify.whatbird.com/). Figure 76 compares estimates of maximum nutrient loads due to excretion from a range of gull population sizes to the external nutrient loads to Lake Weir for the period 2007-2019. The estimated TP nutrient loads from gull excretion assume that the birds feed away from the lake and come back to roost on the lake at night, and that the specified number of gulls are present all year long. Under these assumptions, a year-round population of 10,000 Ring-billed gulls would excrete about 59% of the external TP load, but only about 7% of the external TN load.

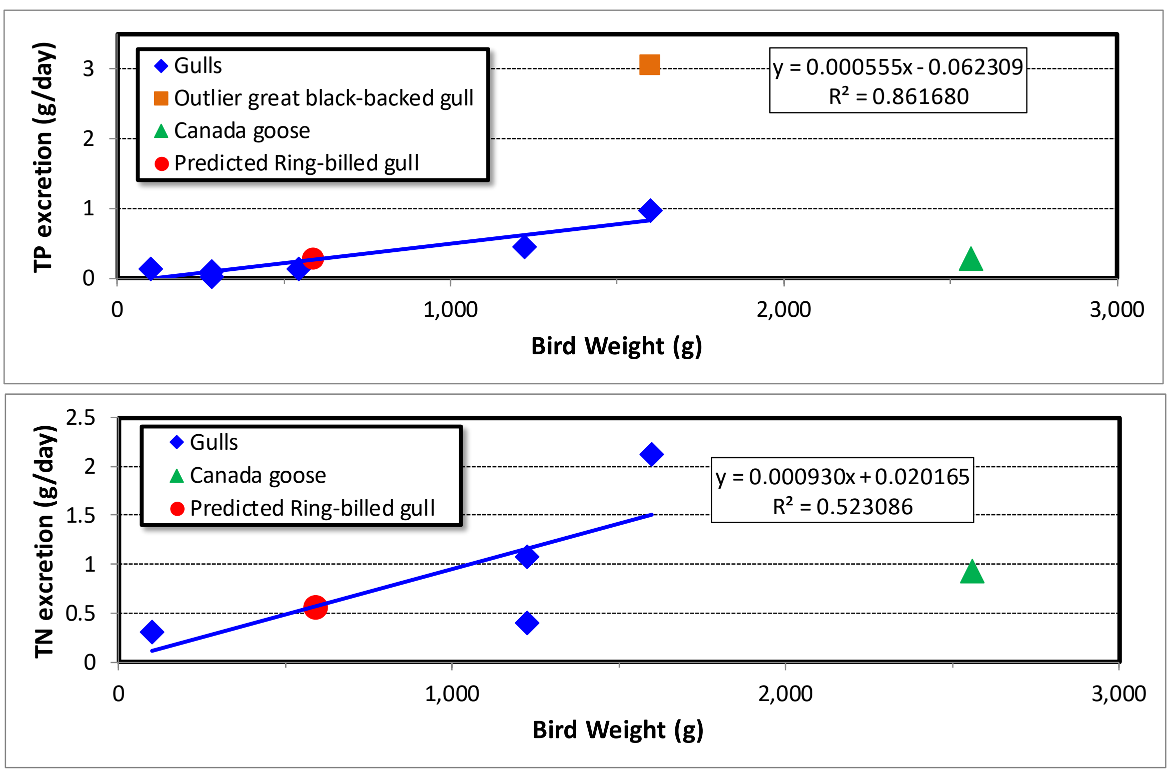
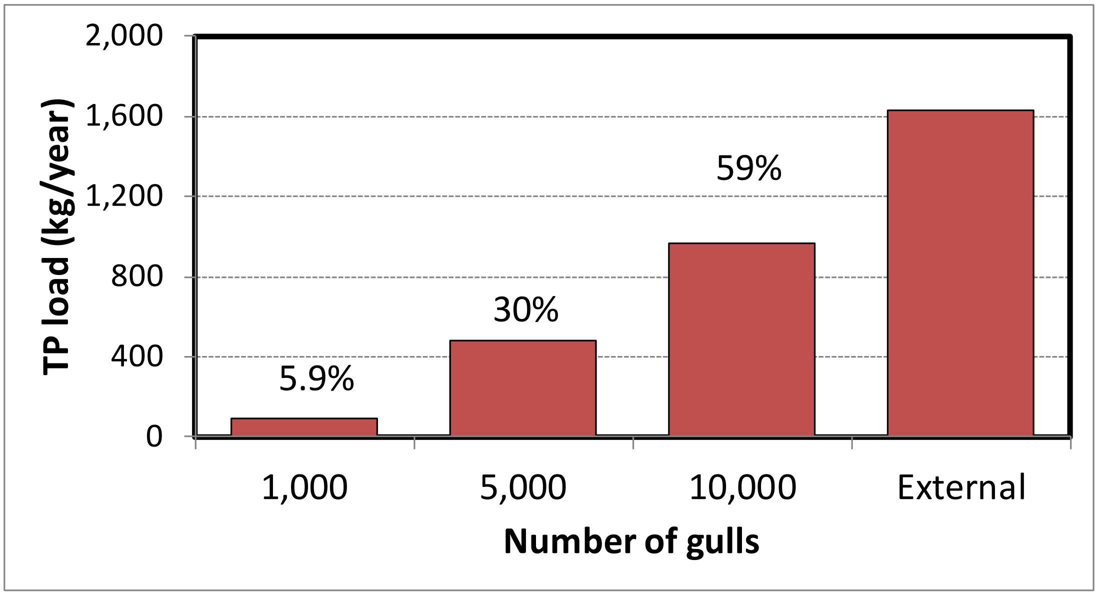


Figure 75. Reported waterfowl total phosphorus and total nitrogen excretion rates vs. bird weight.



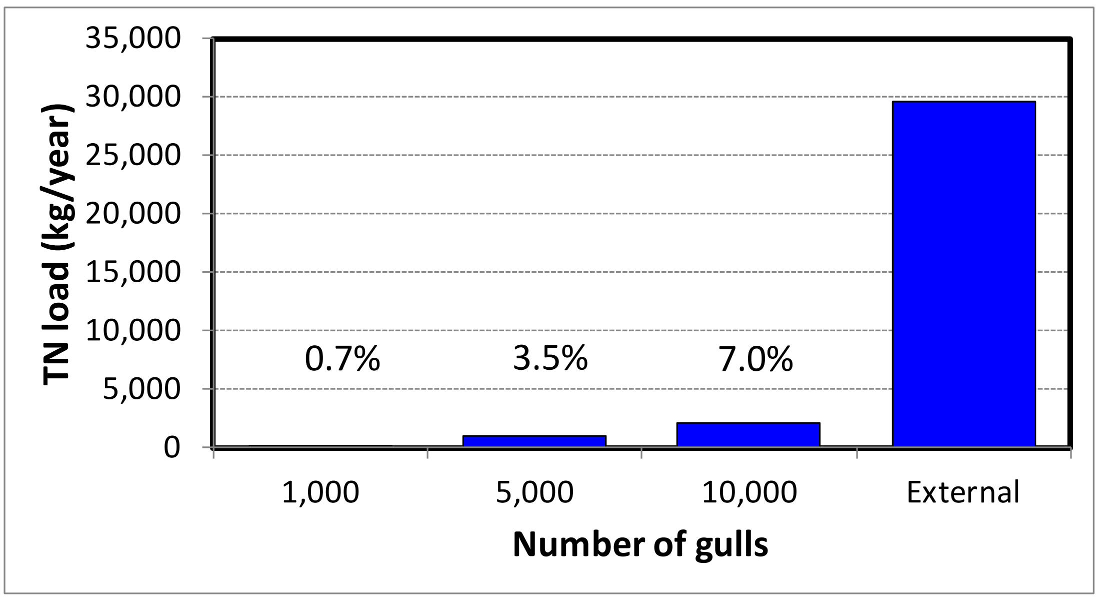


Figure 76. Comparison of maximum nutrient loads from excretion by Ring-billed gull populations to estimated external nutrient loading to Lake Weir from all other sources.

In the last 5 years, average concentrations for Lake Weir have been 20 μg TP/L and 15 μg chlorophyll-*a*/L. These concentrations exceed the PLRG target TP concentration of 14 μg/L, (and the TMDL target concentration of 10 μg/L) and the expected chlorophyll-*a* concentration of 8 μg/L at the PLRG TP target. Average TN concentration in Lake Weir over the last 5 years has been 1.0 mg/L, which exceeds the TMDL target concentration of 0.68 mg/L. Comparison of trends in concentrations and masses show some evidence of greater increases in concentrations during low-water periods, most clearly for TN (Figure 78, Figure 78).

During the period 1999-2003, most of the SJRWMD samples for chlorophyll-*a* were sent to the FDEP laboratory for analysis. The FDEP measurements were substantially lower than SJRWMD analyses or contemporaneous measurements by the Lakewatch program, so I excluded the FDEP measurements from my analysis. I filled in the missing period in Figure 73 and Figure 78 with chlorophyll-*a* data obtained from Lakewatch, but for the trend analyses I used only data analyzed by the SJRWMD laboratory.

Results of water quality trend analyses for Lake Weir are summarized in Table 11, with more detailed results including Kendall's tau nonparametric correlation coefficient and statistical significance level of the nonparametric Theil-Sen trend lines in Appendix Table B- 8. Time series plots are shown for several parameters in Figure 79 and Figure 80.

Lake Weir showed deteriorating trends for several water quality variables for unadjusted data, but not for water elevation-adjusted data, over both the entire period of record and 2001-2019. These deteriorating trends for only unadjusted data included TP, TN, and chlorophyll-*a*, over the entire period of record and 2001-2019, and TP and Secchi transparency for 2001-2019 (Table 11). These contrasts between unadjusted data and stage-adjusted residuals indicate that the apparent deteriorating trends in water quality may have been due to water elevation decreases over the analysis period. There were increasing trends in TSS over both the entire period and 2001-2019. There was a decreasing trend in Secchi transparency over the entire period of record. Lake Weir showed perhaps the largest water elevation effects on the trend analyses of all the lakes. Lake Weir has had a substantial decrease in water elevations over the period of water quality sampling – nearly 7 feet at its lowest point (Figure 77), which resulted in a decrease of over 35% in estimated lake volume. For the entire period, synthetic NH_4-T showed a decreasing trend for unadjusted data, but no trend in residuals from the Lowess smooth (Figure 80). Unlike in the other lakes, there were no exceedances of the Florida ammonia standard in Lake Weir.

These trend assessment results were generally similar to the results of the last assessment (Fulton 2018), although there were a few more significant trends this time, reflecting a continued deterioration in water quality.

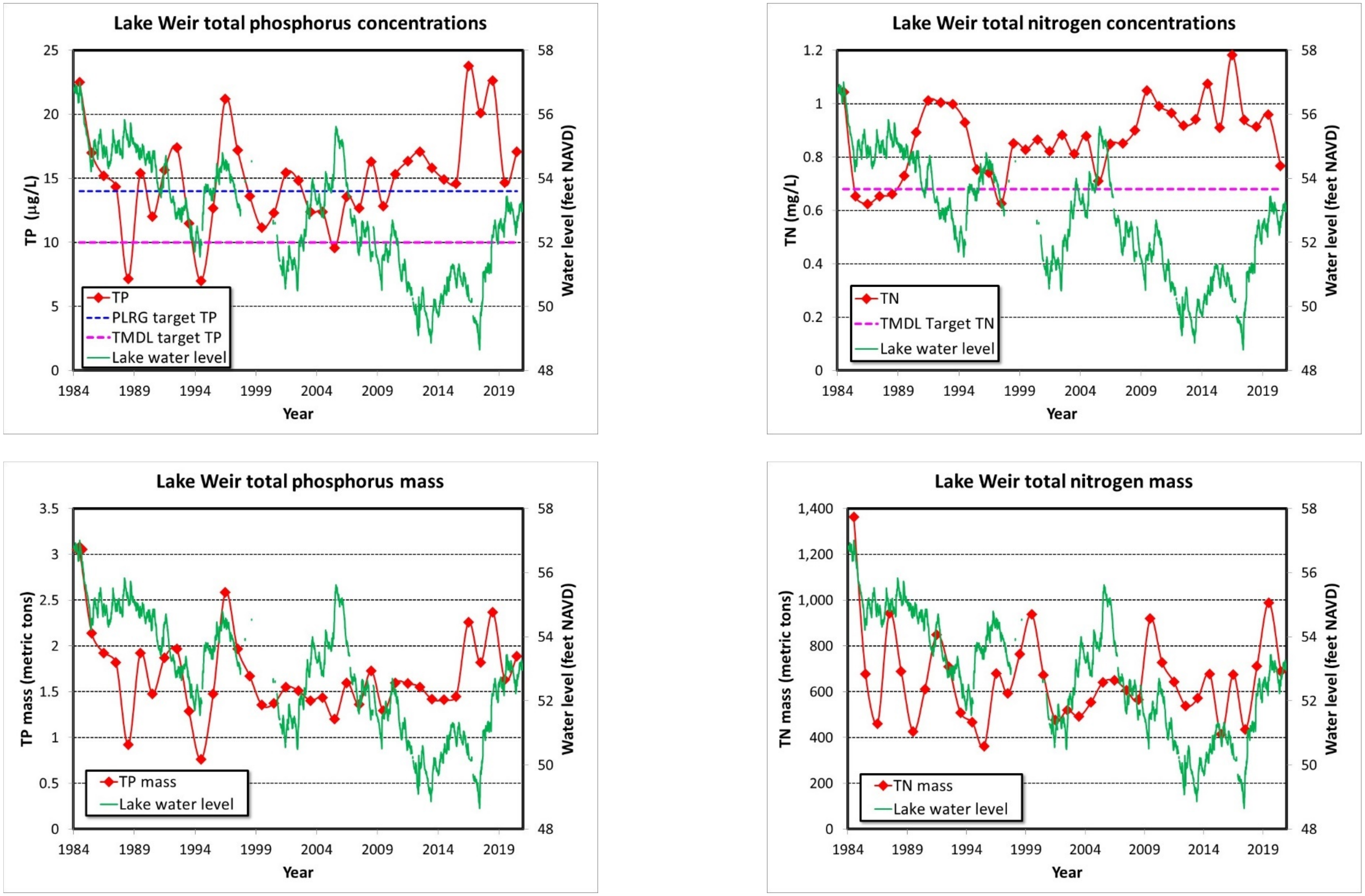


Figure 77. Lake Weir, mean annual total phosphorus and total nitrogen concentrations and masses, and water elevations.

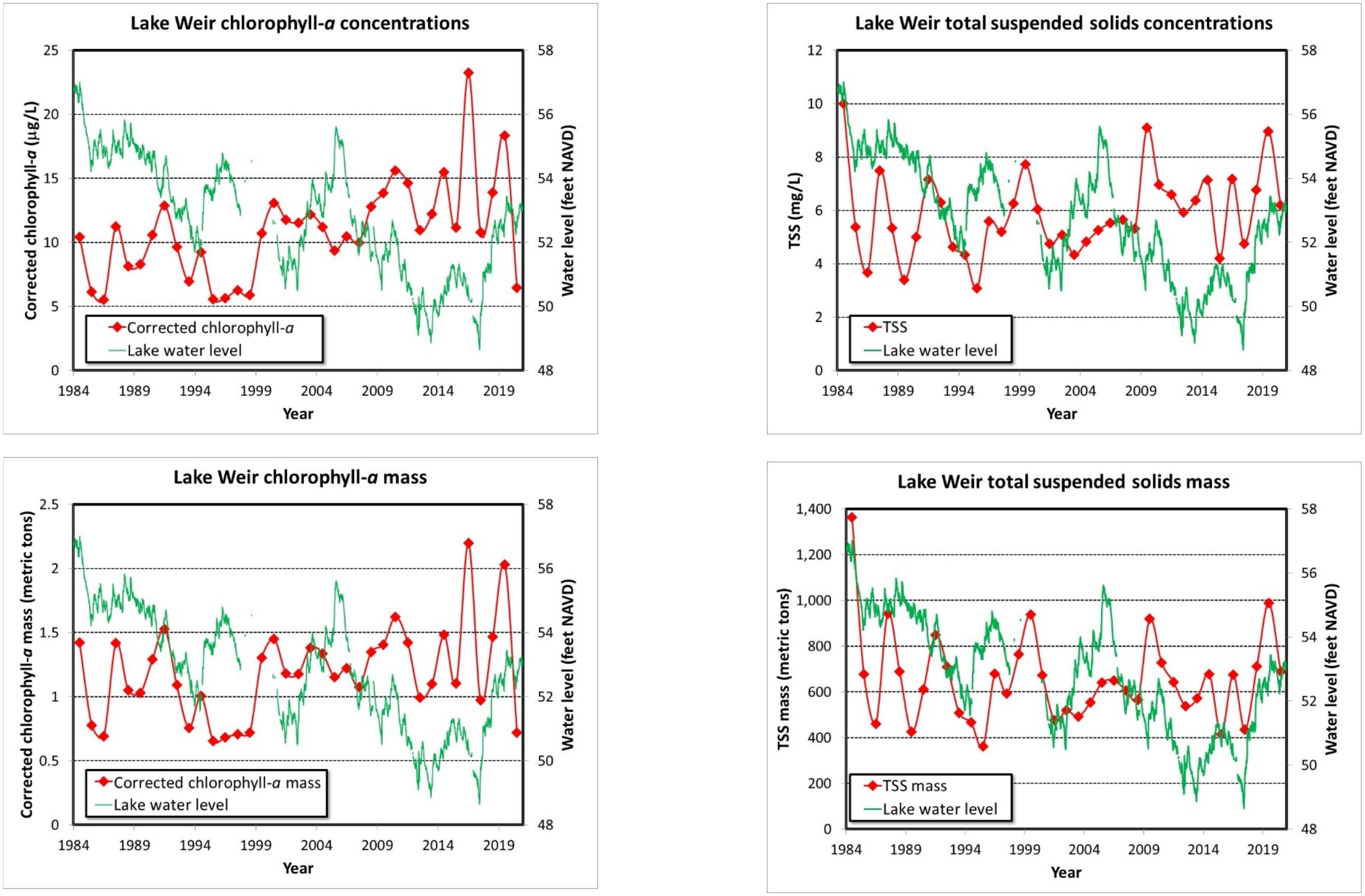


Figure 78. Lake Weir, mean annual chlorophyll-*a* and total suspended solids concentrations and masses, and water elevations.

Table 11. Water quality trends for Lake Weir.

Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality, $p<0.05$

(0) No trend, $p \geq 0.05$

(–) Significant decreasing trend, $p<0.05$

(+) Significant increasing trend, $p<0.05$

	1984 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, +	S, 0	+	0
TP [b]	+	0	+	0
TN	+	0	+	+
Chlorophyll-*a*	+	S, 0	0	0
TSS	+	+	+	+
TSS [b]	+	+	+	+
Secchi	–	–	–	0
NH_4 [a, b]	–	0	S, 0	0
NO_x [a, b]	ND [c]	ND	S, 0	0
PO_4 [a, b]	ND	ND	S, 0	0
TP mass	S, 0		0	
TN mass	0		0	
Chlorophyll-*a* mass	+		0	
TSS mass	0		+	

[a] Synthetic total fraction for 1984-2019; dissolved fraction for 2001-2019

[b] Non-detect methods

[c] ND – No data

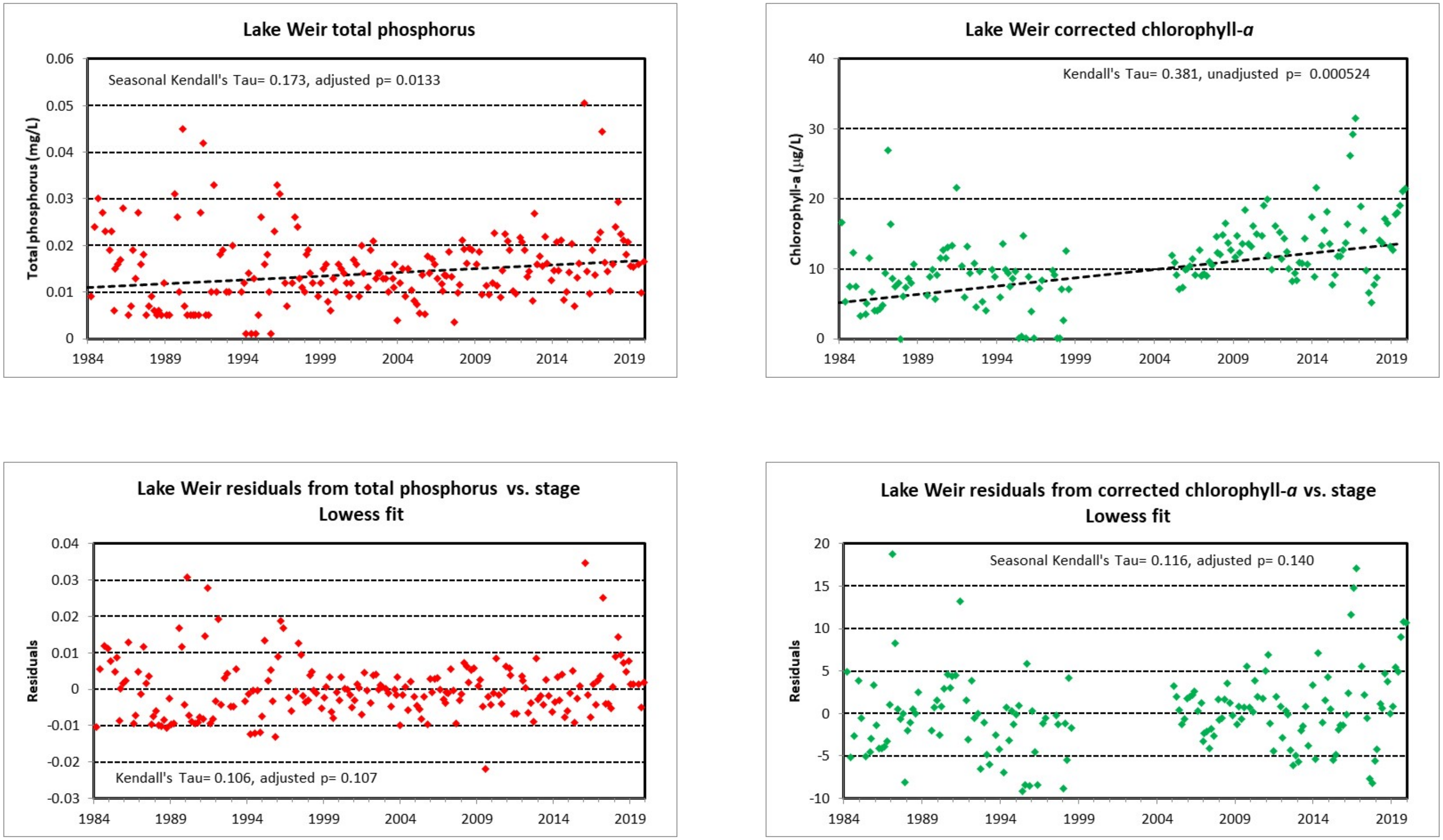


Figure 79. Water quality trends in Lake Weir, 1984-2019.

Daily means and Theil-Sen trend lines.

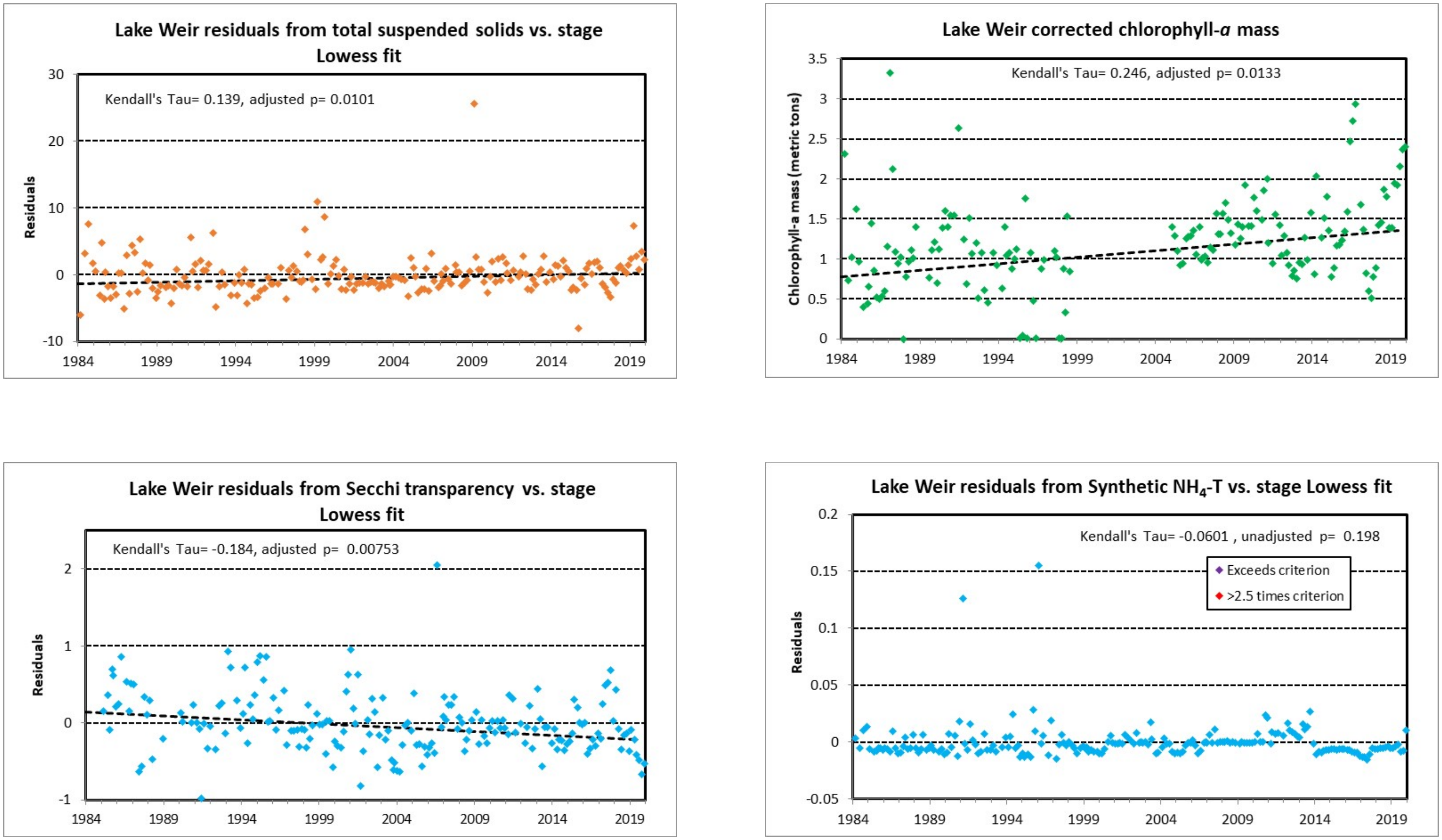


Figure 80. Water quality trends in Lake Weir, 1984-2019.

Daily means and Theil-Sen or ATS trend lines.

There is a shorter period of record for phytoplankton data from Lake Weir than for the other lakes. Phytoplankton sampling ended in Lake Weir after August 2018. There may be an increasing trend in phytoplankton biovolume, with clear peaks in both biovolume and chlorophyll in 2016 (Figure 81). As with the other lakes, the phytoplankton in Lake Weir tend to be dominated by cyanobacteria, although there is more consistent representation of other taxa, including diatoms, dinoflagellates, and green algae (Figure 82, Figure 82). Again, *Cylindrospermopsis* tended to be the dominant cyanobacterial genus in Lake Weir, although there were periods in which *Microcystis* and *Planktolyngbya* were prominent. Only the first three samples in Lake Weir were analyzed by Phycotech and *Pseudanabaena* was prominent in them. *Pseudanabaena* was prominent in only one of the later samples analyzed by BSA.

There was a reported algal bloom (dominated by *Microcystis*) and associated fish kill in Lake Weir in April 2015. This was not apparent in our closest sample, which was collected about two weeks before the reported bloom event. There was a bloom of the green alga *Botryococcus* in Lake Weir in February 2019 (Figure 83). *Botryococcus* is brown-red in color and accumulates large amounts of lipids. This lipid production makes it buoyant and causes it to accumulate along shorelines. *Botryococcus* is being investigated for commercial bioproduction of fuels. Chlorophyll-*a* concentration was not unusually high in our February 2019 sample, but that was at a center lake station and the bloom was primarily concentrated near the shoreline.

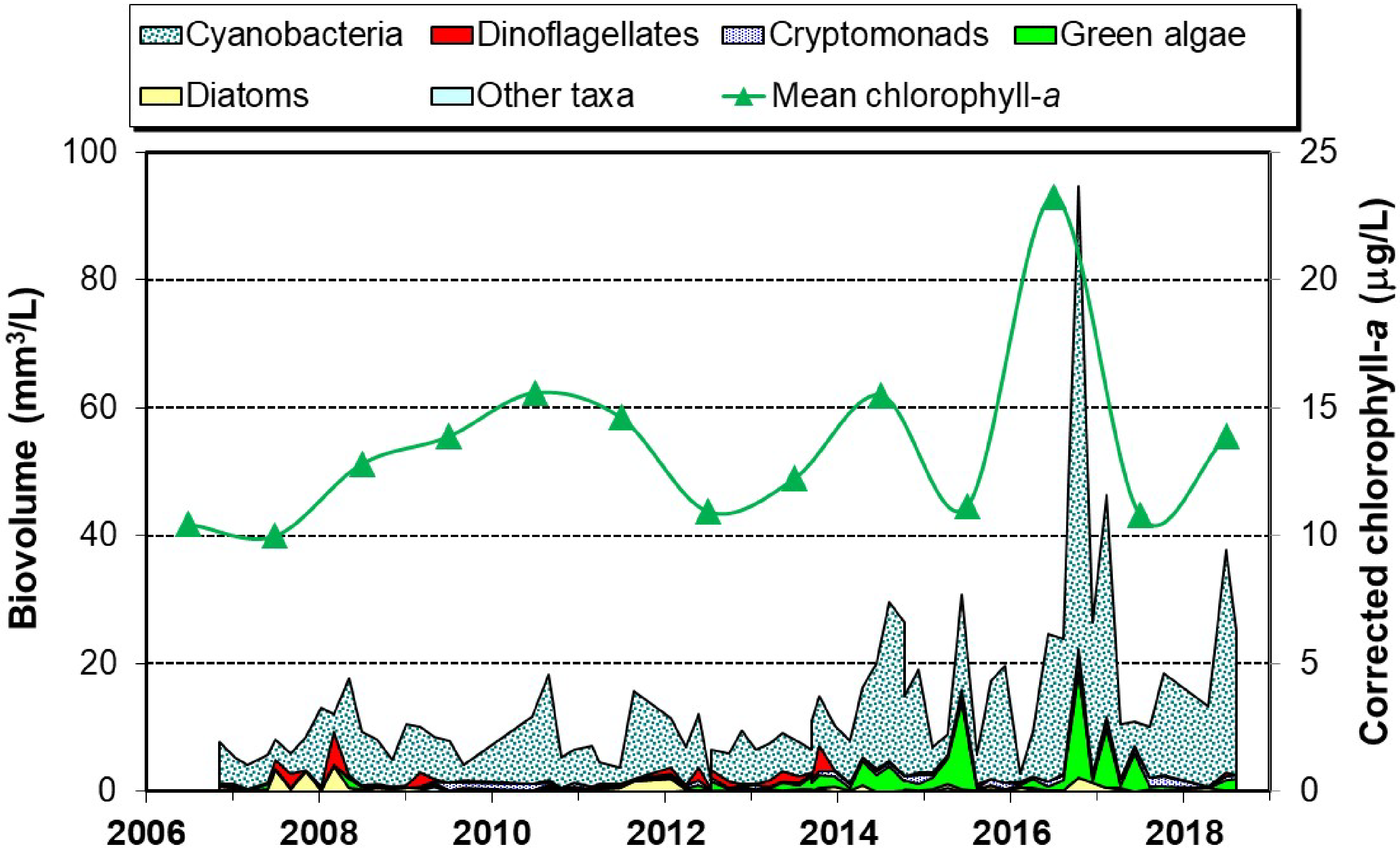


Figure 81. Lake Weir phytoplankton biovolumes and chlorophyll-*a* concentrations.

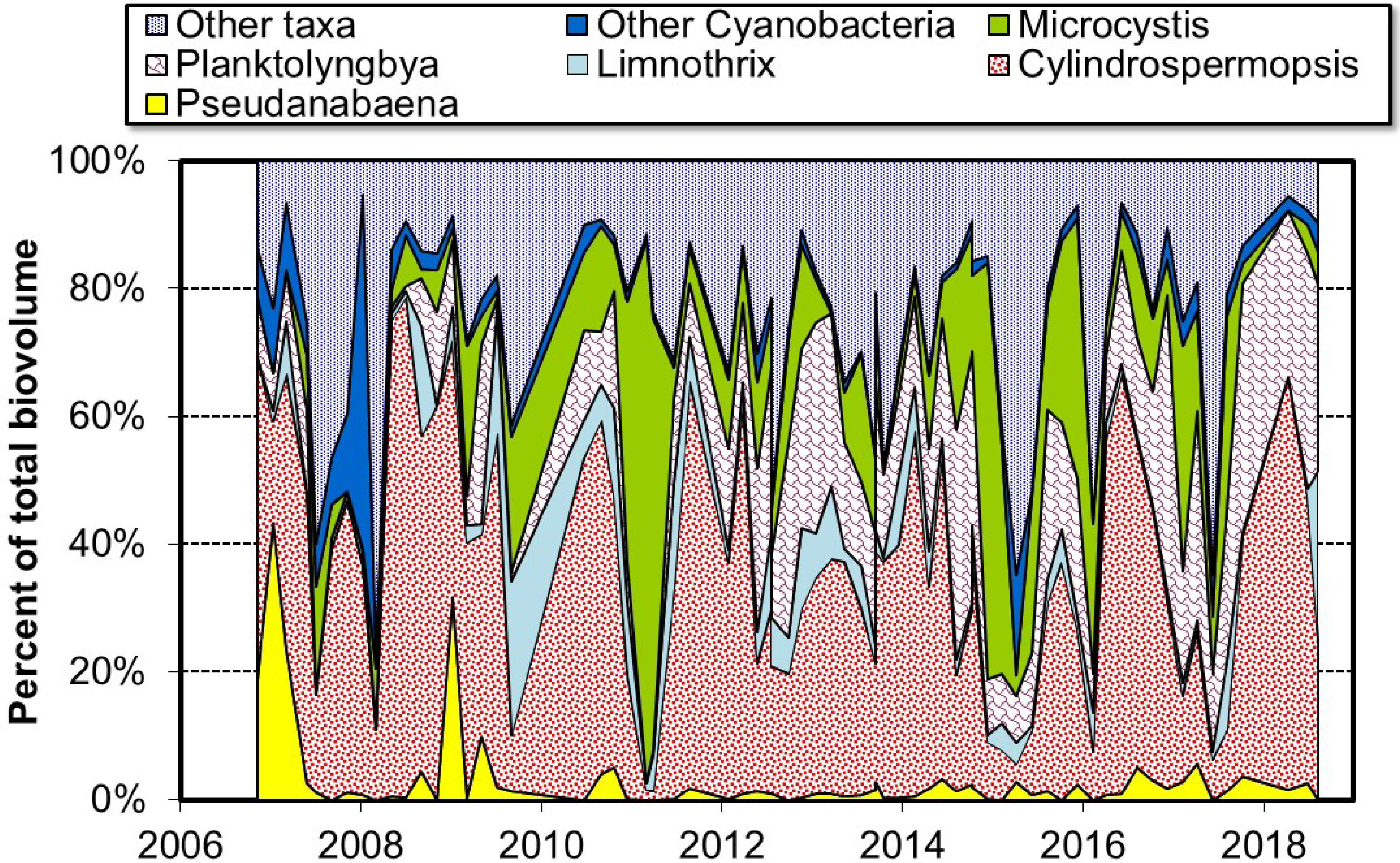


Figure 82. Lake Weir phytoplankton percent composition.

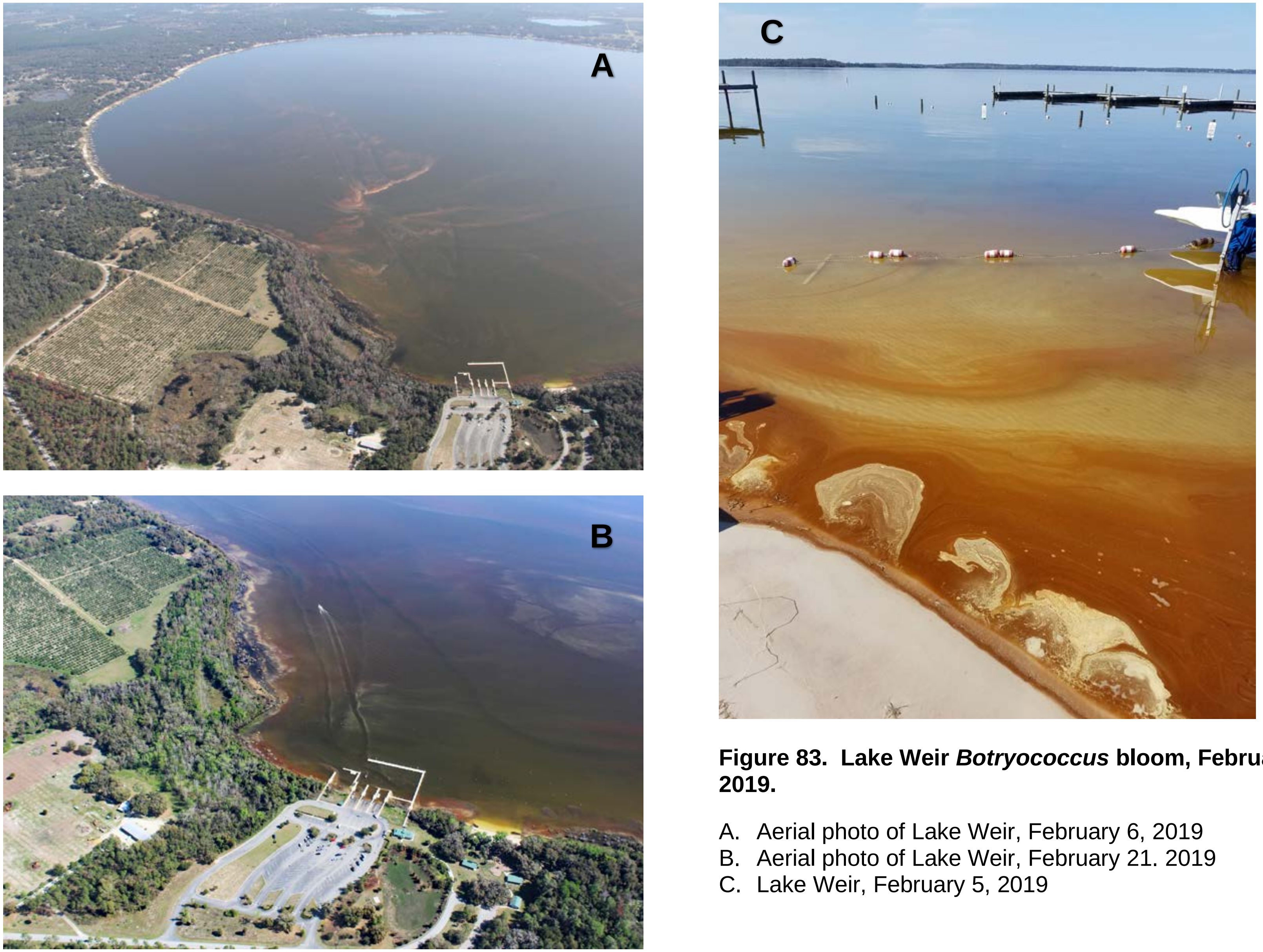


Figure 83. Lake Weir *Botryococcus* bloom, February 2019.

A. Aerial photo of Lake Weir, February 6, 2019
B. Aerial photo of Lake Weir, February 21. 2019
C. Lake Weir, February 5, 2019

LAKE DENHAM

We have not developed annual external nutrient budgets for Lake Denham. However, FDEP developed external nutrient budgets for the lake for the years 2000-2012, using very similar methods as we have used (Rhew 2017b). The FDEP nutrient budgets also included internal sources. A muck farm in the Lake Denham watershed accounted for about 44% of the TP surface runoff, and about 29% of the TN surface runoff. Lake County Water Authority has recently purchased and plans to restore the Lake Denham muck farm. We have little information on operations of the muck farm in the Lake Denham watershed, although notes from a BMAP meeting indicate it had ceased commercial production by 2015.

SJRWMD conducted an experimental gizzard shad removal project in Lake Denham from 1990 to 1992 (Godwin et al. 2011). Over that period there appeared to be a substantial decrease in TP and chlorophyll-*a* concentrations in the lake (Figure 85, Figure 85). However, average TP and TN concentrations in the lake have always exceeded the TMDL target concentrations. Comparison of trends in concentrations and masses show some evidence of greater increases in concentrations during low-water periods, most clearly for TN and TSS.

In the last 5 years, TP concentration for Lake Denham has averaged 91 μg/L, and chlorophyll-*a* concentration has averaged 64 μg /L. These concentrations exceed the TMDL target TP concentration of 40 μg/L, and the TMDL target chlorophyll-*a* concentration of 26.8 μg/L. Average TN concentration in Lake Denham over the last 5 years has been 2.1 mg/L, which also exceeds the TMDL target concentration of 1.1 mg/L.

Results of water quality trend analyses for Lake Denham are summarized in Table 12, with more detailed results including Kendall's tau nonparametric correlation coefficient and statistical significance level of the nonparametric Theil-Sen trend lines in Appendix Table B-9. Time series plots are shown for several parameters in Figure 86.

There were significant decreasing trends in Lake Denham for TP and water elevation-adjusted residuals for the both the entire period of record and for 2001-2019. TSS showed decreasing trends for the entire period of record, but not for 2001-2019. TN showed a significant decreasing trend in mass over the entire period of record, but not in concentrations, although the trend in the water elevation-adjusted residuals was marginally nonsignificant ($p=0.0863$, Appendix Table B- 9), suggesting a weak trend when accounting for water level changes. NH_4-D showed a significant decreasing trend over the 2001-2019 period, but not for the water elevation-adjusted residuals. There were several exceedances of the Florida ammonia standard in Lake Denham (Figure 86).

Phytoplankton sampling ended in Lake Denham after August 2018. There is no apparent trend in phytoplankton biovolumes (Figure 87). As with the other lakes, the phytoplankton generally has been dominated by cyanobacteria. Again, *Cylindrospermopsis* tended to be the dominant cyanobacterial genus in Lake Denham (Figure 88). As with some of the other lakes, *Microcystis* became more prominent after the change in analysts to BSA in 2007.

Table 12. Water quality trends for Lake Denham.

Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality, $p<0.05$

(0) No trend, $p \geq 0.05$

(–) Significant decreasing trend, $p<0.05$

(+) Significant increasing trend, $p<0.05$

	1989 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, –	–	–	–
TN	0	0	0	0
Chlorophyll-*a*	S, 0	S, 0	0	0
TSS	–	–	0	0
Secchi	0	0	0	0
NH_4 [a, b]	0	0	–	0
NO_x [a, b]	0	0	0	S, 0
PO_4 [b]	0	0	0	0
TP mass	–		0	
TN mass	–		0	
Chlorophyll-*a* mass	S, 0		0	
TSS mass	–		0	

[a] Dissolved fraction for both time periods.

[b] Non-detect methods

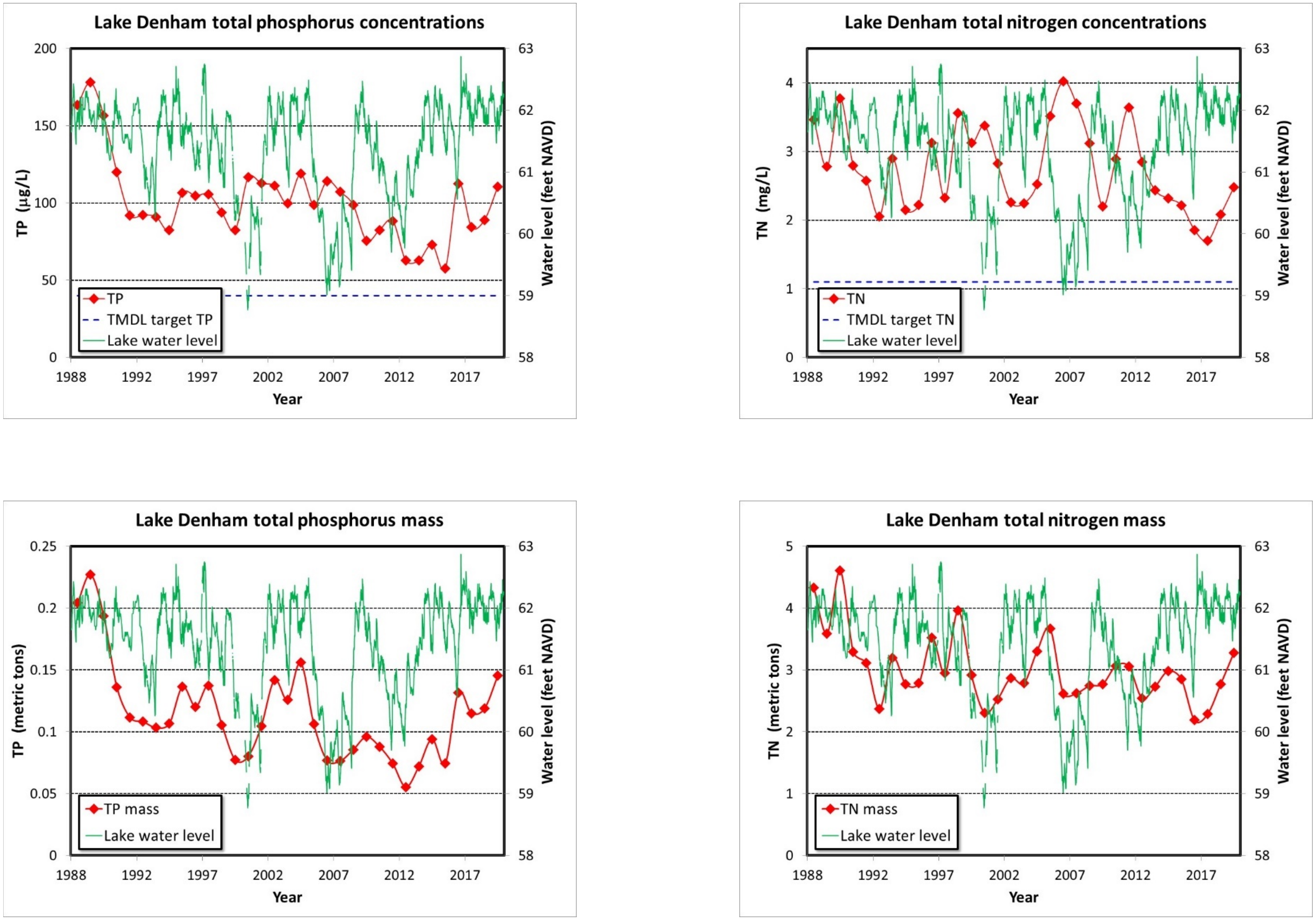


Figure 84. Lake Denham, mean annual total phosphorus and total nitrogen concentrations and masses, and water elevations.

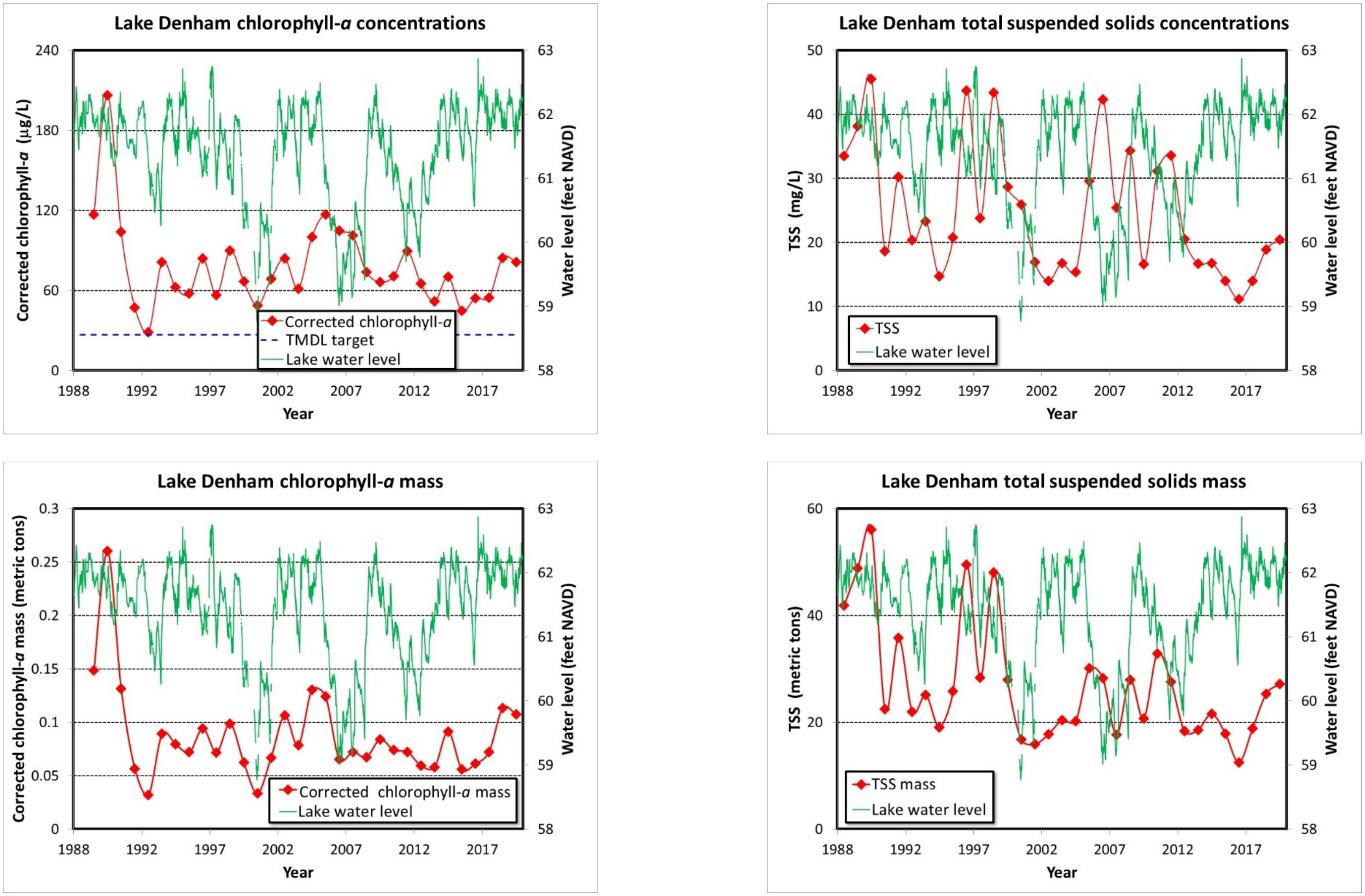


Figure 85. Lake Denham, mean annual chlorophyll-*a* and total suspended solids concentrations and masses, and water elevations.

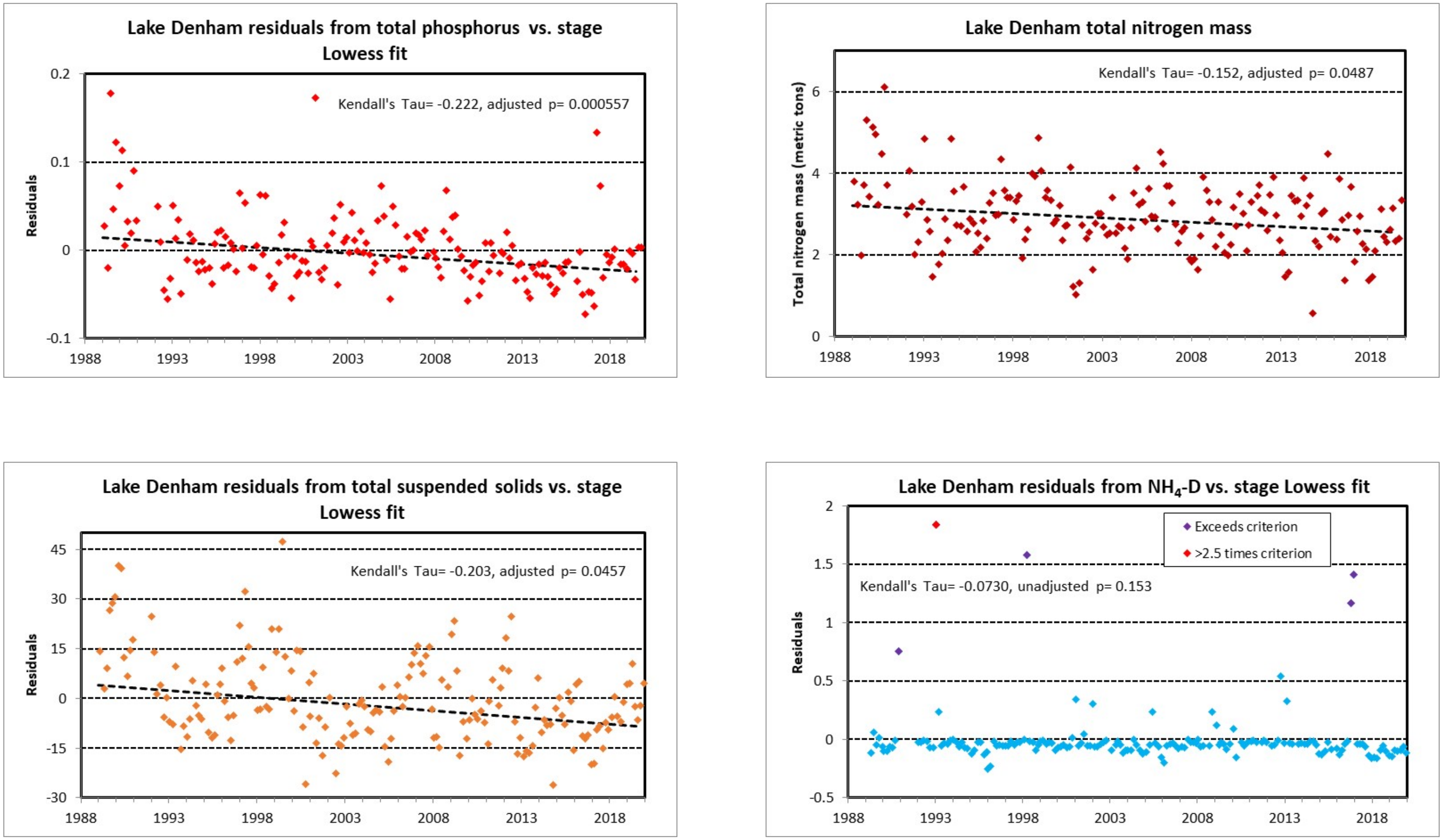


Figure 86. Water quality trends in Lake Denham, 1989-2019.

Daily means and Theil-Sen or ATS trend lines.

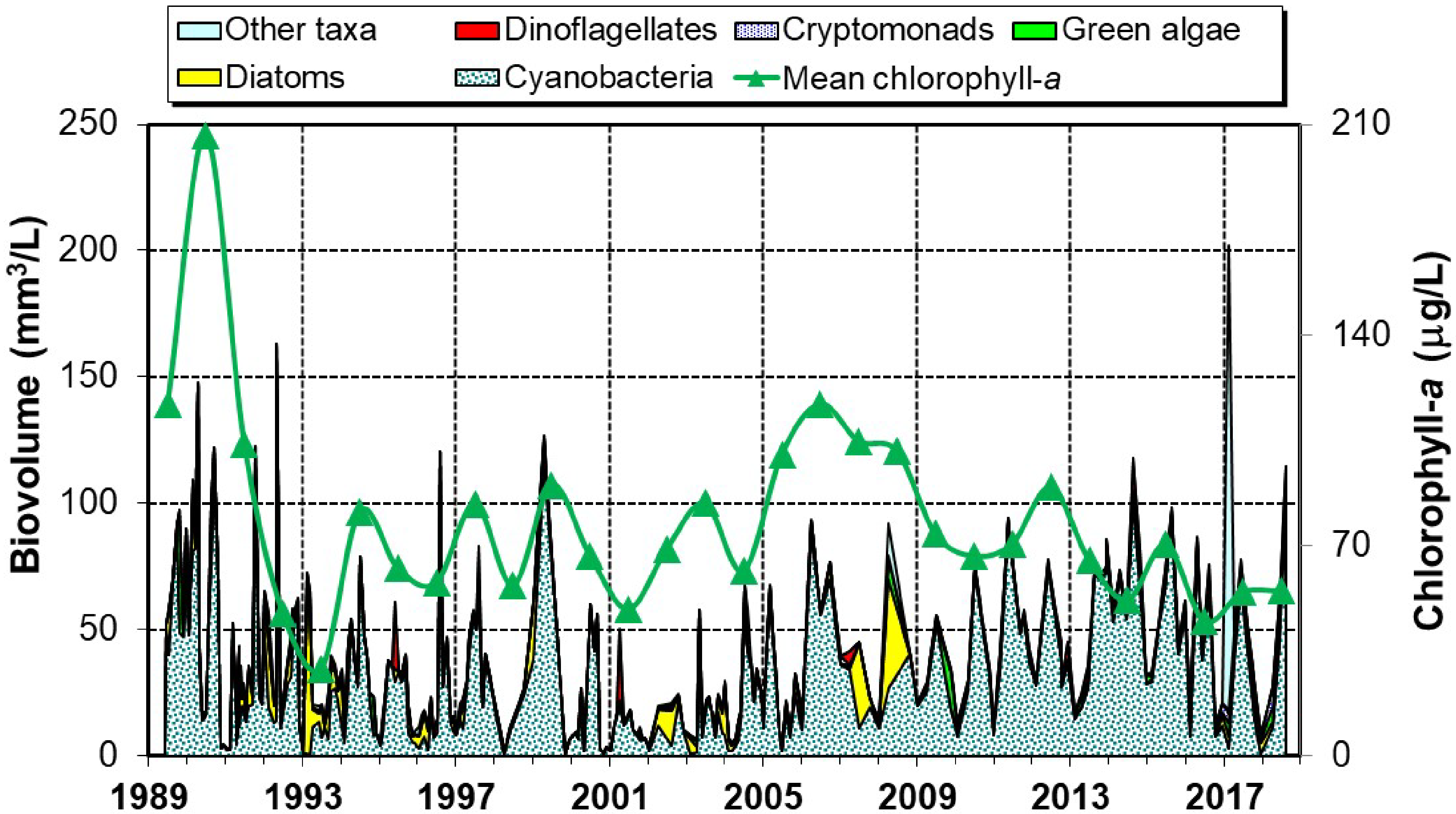


Figure 87. Lake Denham phytoplankton biovolumes and chlorophyll-*a* concentrations.

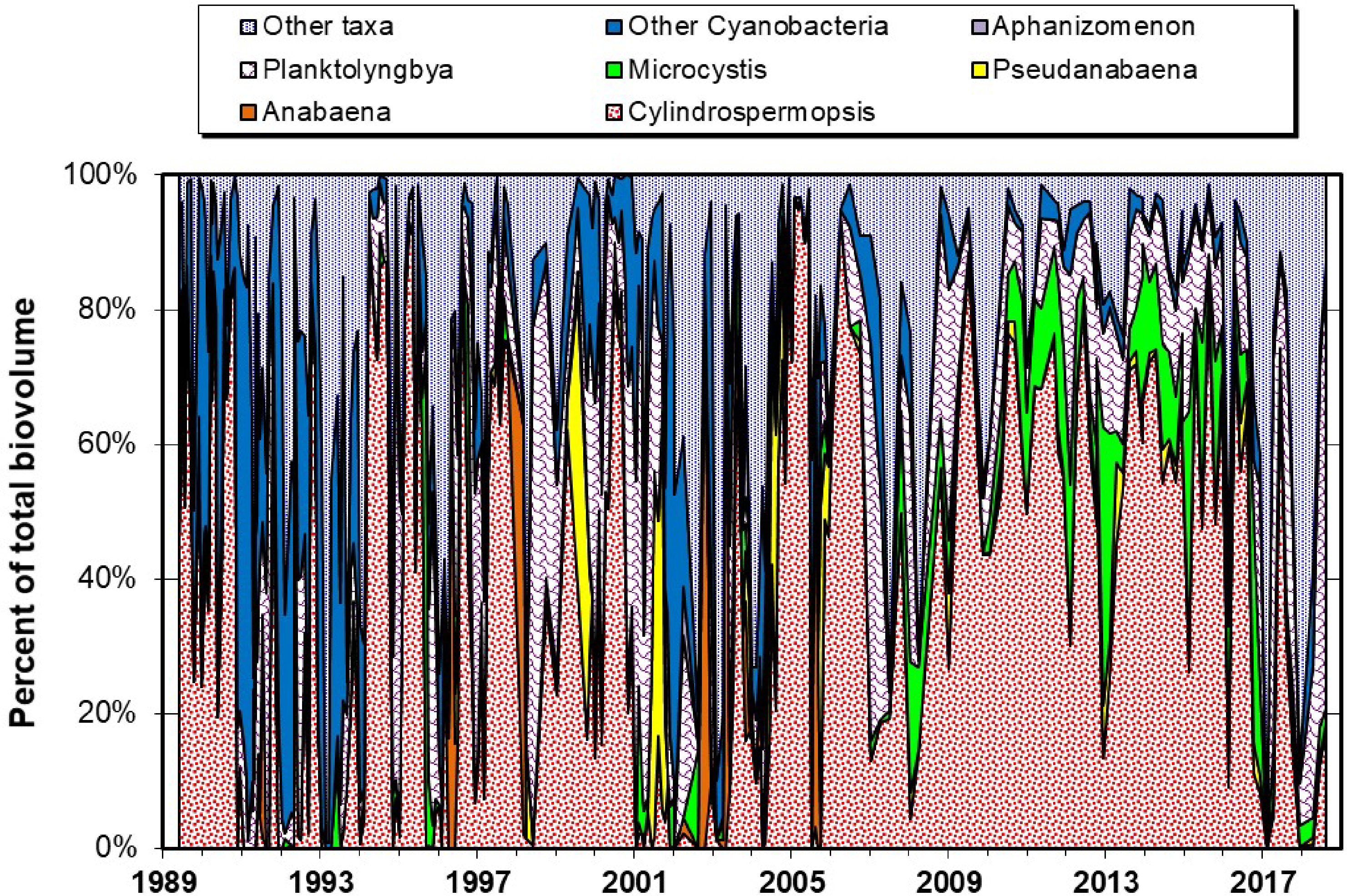


Figure 88. Lake Denham phytoplankton percent composition.

LAKE CARLTON

We have not developed complete annual external nutrient budgets for Lake Carlton. Although external loads have been estimated for the watershed sources for Lake Carlton, we do not have estimates of nutrient flows in water exchanges with adjacent Lake Beauclair. Anderson and Hughes (1977) found frequent flow reversals between the two lakes, with flows approaching 200 cfs. Concentrations and masses show rather similar trends in Lake Carlton other than greater increases in concentrations during the 2007 drought (Figure 90, Figure 90).

In the last 5 years, TP concentration for Lake Carlton has averaged 47 μg/L, and chlorophyll-*a* concentration has averaged 54 μg /L. These concentrations exceed the TMDL target TP concentration of 32 μg/L, and an expected chlorophyll-a concentration of 30 μg/L at the TP target. Average TN concentration in Lake Carlton over the last 5 years has been 2.0 mg/L.

Results of water quality trend analyses for Lake Carlton are summarized in Table 13, with more detailed results including Kendall's tau nonparametric correlation coefficient and statistical significance level of the nonparametric Theil-Sen trend lines in Appendix Table B-10. Water quality trends for Lake Carlton. Time series plots are shown for several parameters in Figure 91.

Lake Carlton showed improving trends for TP and masses of TP, TN, and chlorophyll-*a*. There were no significant trends for TN and chlorophyll-*a* concentrations, but water elevation-adjusted trends were significant, suggesting that water level fluctuations masked a decreasing trend, which is also supported by the decreasing trend in masses for these parameters. Conversely, unadjusted Secchi transparency showed an improving trend, but there was no significant trend in water elevation-adjusted residuals, although the p-value for the residuals trend was just above significance ($p=0.0604$). NH_4-D showed a significant decreasing trend in both the unadjusted measurements and in the residuals when analyzed by non-detect methods, but not for the residuals analyzed by standard methods. This discrepancy may have been partially due to the absence of correction of the p-value for serial correlation in the non-detect methods, but the Kendall's tau was also larger for the residuals in the non-detect analysis. There was only one exceedance of the Florida ammonia standard in Lake Carlton (Figure 91).

There have been no projects to reduce nutrient loading in the Lake Carlton watershed. The water quality improvements in the lake are probably due to unquantified water exchanges with adjacent Lake Beauclair, so Lake Carlton also benefits from the restoration projects upstream of lake Beauclair.

Table 13. Water quality trends for Lake Carlton.

Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality, $p<0.05$

(0) No trend, $p\geq 0.05$

(–) Significant decreasing trend, $p<0.05$

(+) Significant increasing trend, $p<0.05$

	2002 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, –	S, –
TN	S, 0	–
Chlorophyll-*a*	S, 0	–
TSS	0	0
Secchi	+	0
NH_4	S, –	0
NH_4 [a]	S, –	S, –
NO_x [a]	S, 0	S, 0
PO_4 [a]	0	0
TP mass	S, –	
TN mass	S, –	
Chlorophyll-*a* mass	S, –	
TSS mass	0	

[a] Non-detect methods

Phytoplankton sampling ended in Lake Carlton after August 2018. The decrease in chlorophyll-*a* in the lake was not paralleled by a decrease in phytoplankton biovolumes (Figure 92). Measured biovolumes substantially increased after the change in analysts from Phycotech to BSA in 2007. As with the other lakes, the phytoplankton in Lake Carlton tend to be dominated by cyanobacteria. Again, *Cylindrospermopsis* tended to be the dominant cyanobacterial genus in Lake Carlton (Figure 93). As with several other lakes, *Pseudanabaena* was prominent in the Phycotech counts, but not in those by BSA.

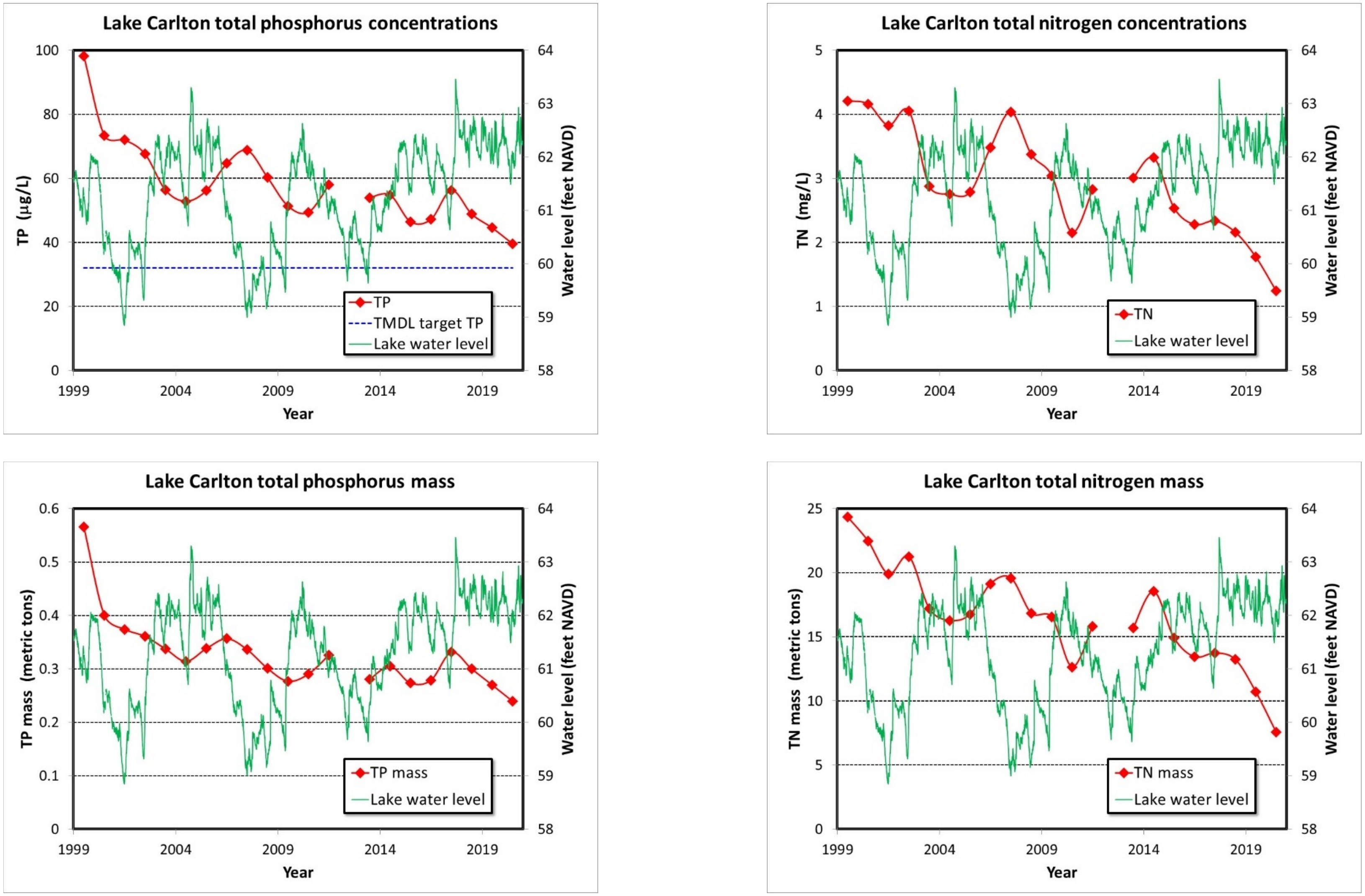
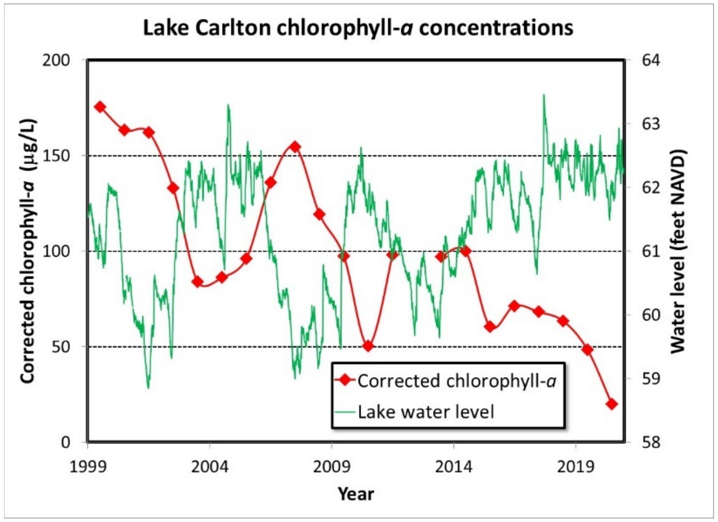
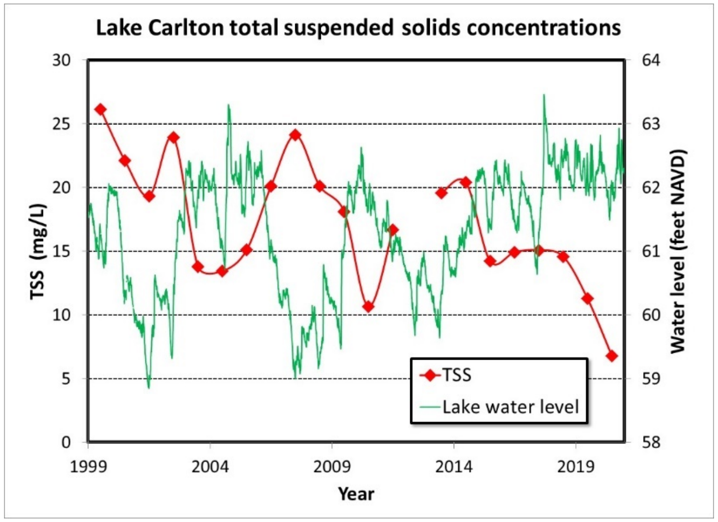
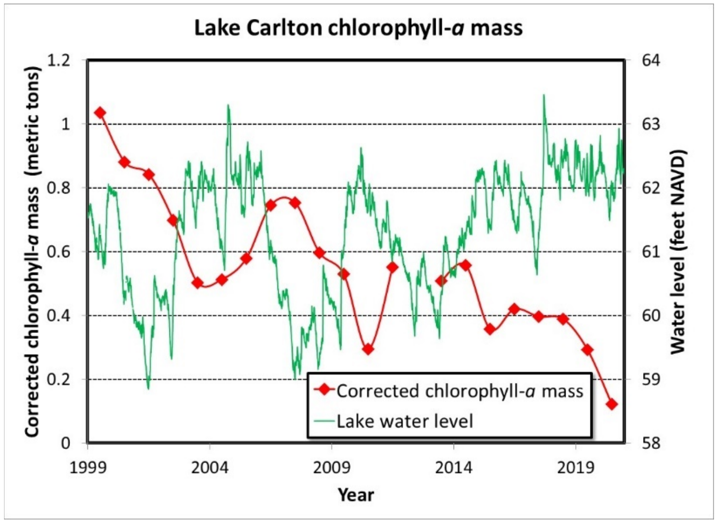


Figure 89. Lake Carlton, mean annual total phosphorus and total nitrogen concentrations and masses, and water elevations.







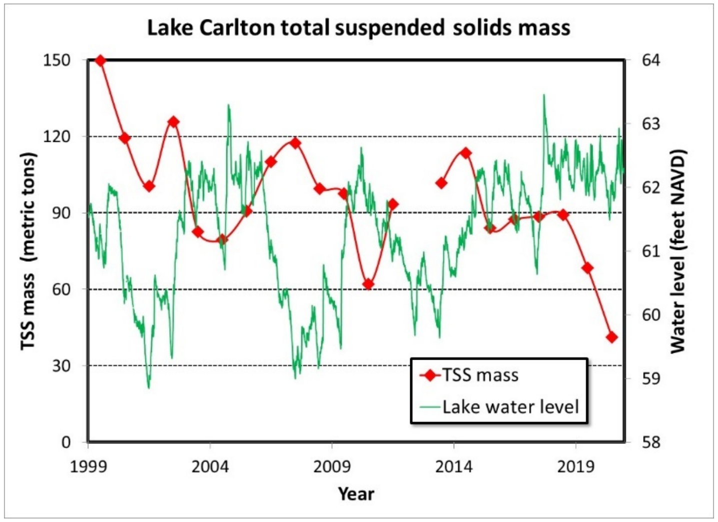


Figure 90. Lake Carlton, mean annual chlorophyll-*a* and total suspended solids concentrations and masses, and water elevations.

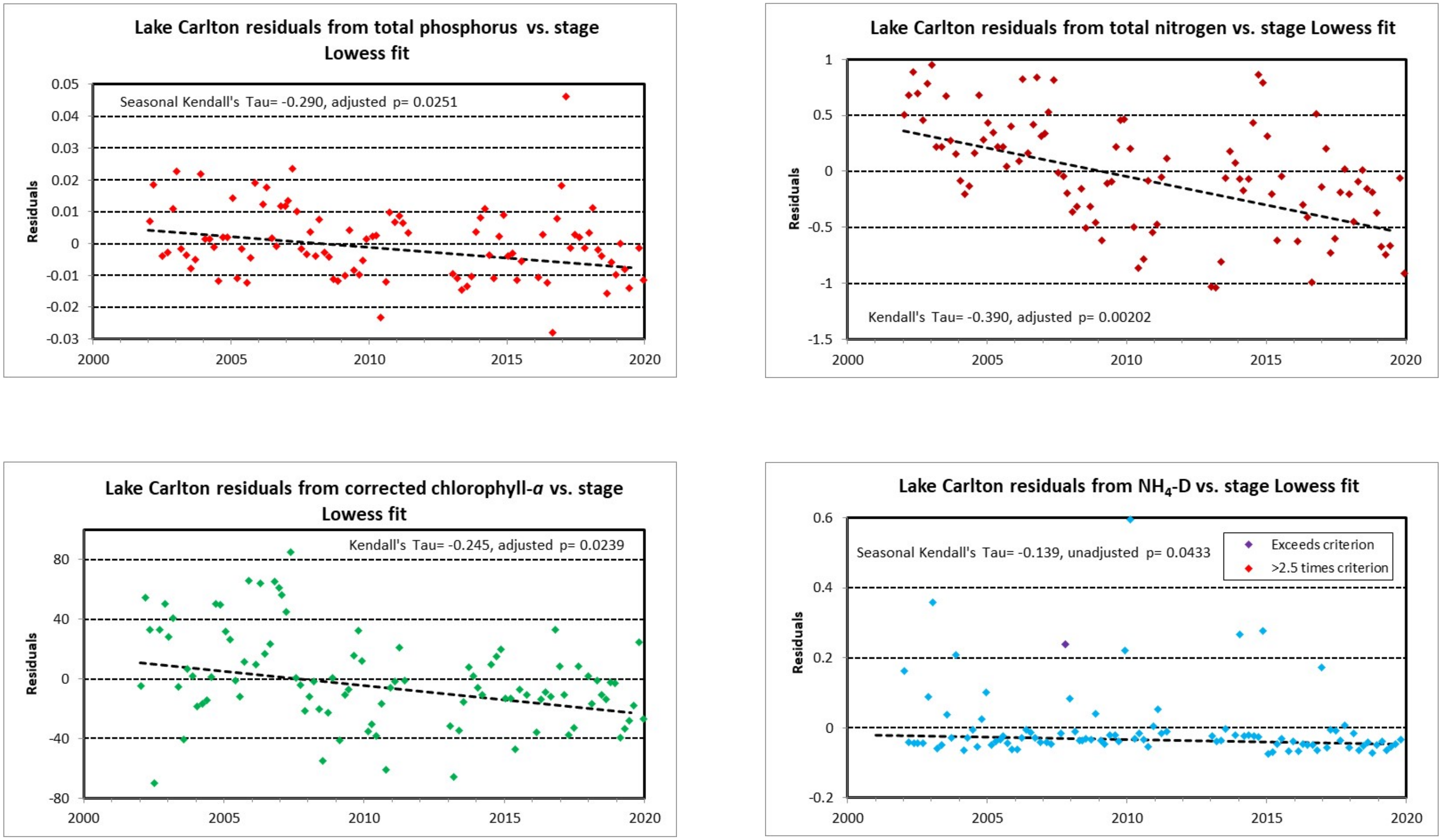


Figure 91. Water quality trends in Lake Carlton, 2002-2019.

Daily means and Theil-Sen or ATS trend lines.

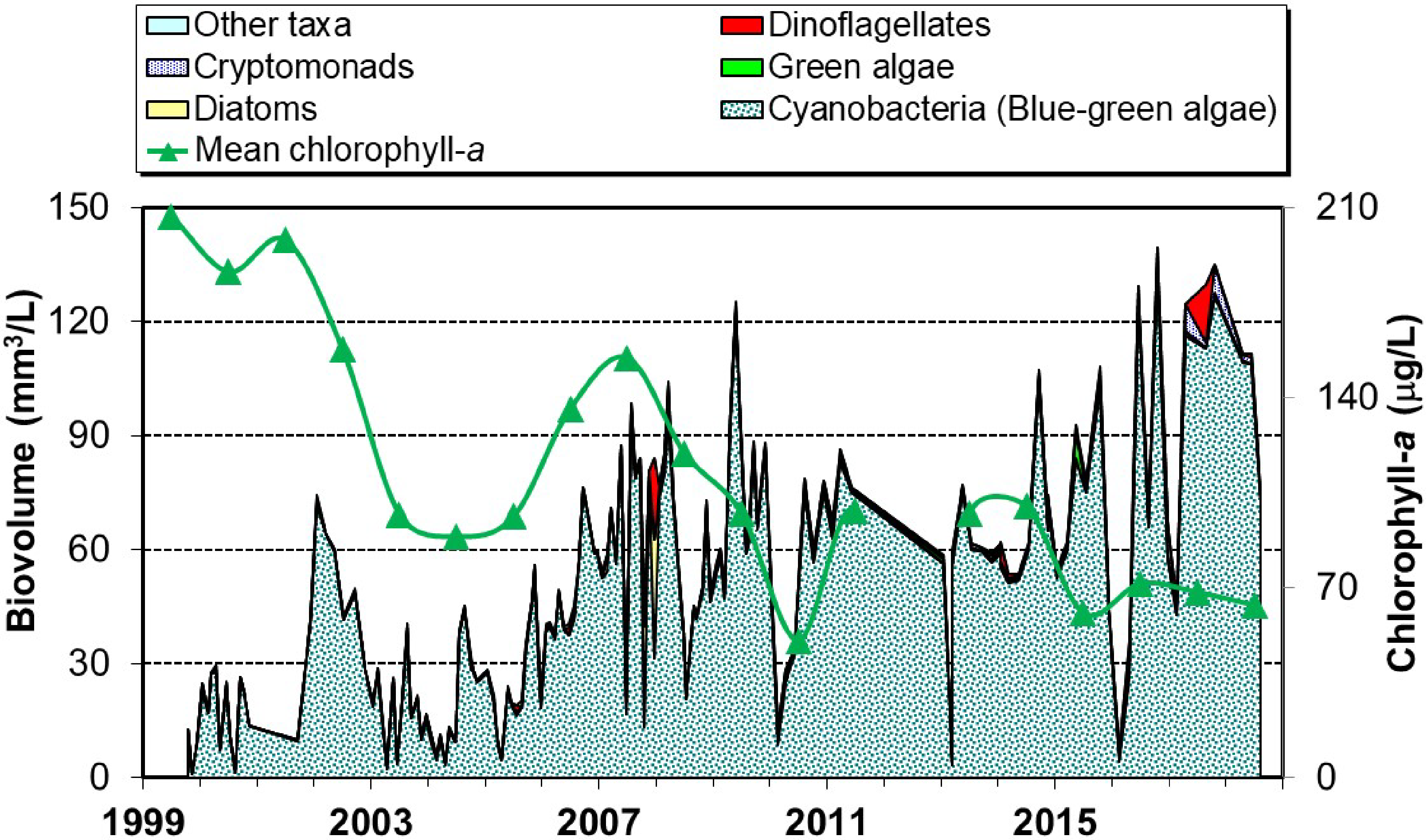


Figure 92. Lake Carlton phytoplankton biovolumes and chlorophyll-a concentrations.

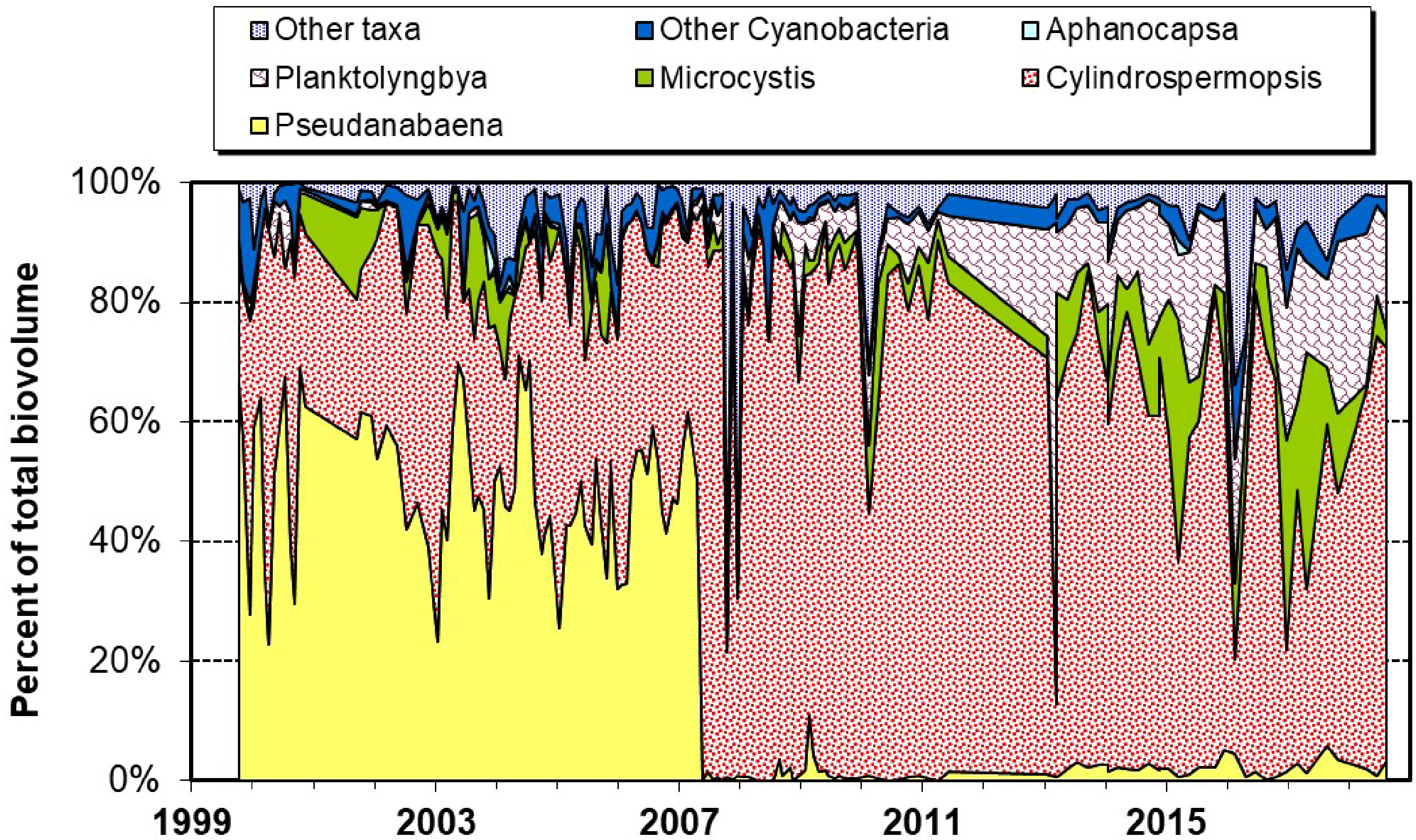


Figure 93. Lake Carlton phytoplankton percent composition.

TROUT LAKE

We do not have annual nutrient budgets for Trout Lake, but FDEP estimated external loading for the years 1995-2000 for the TMDL, using methods similar to those we used for the other basin lakes (Gao and Gilbert 2006). Their estimates of TP discharges from the Pine Meadows Restoration Area were similar to ours and accounted for 49% of the total TP load to Trout Lake. Their estimates of TP discharges from an adjacent muck farm were about half of ours and accounted for 9% of the total TP load to Trout Lake. The remainder of the TP load was primarily watershed runoff from residential and agricultural land uses. Estimated TP discharges from the Pine Meadows Restoration Area substantially decreased during the 1990s, but there has not been much further change since 2000. Our water quality data for Trout Lake don't begin until 2000, and regular sampling did not begin until 2004.

A more recent study (ERD 2017) concluded that surface runoff accounted for 40% of the TP loading to Trout Lake. This study also included estimates of groundwater seepage and internal recycling, which they concluded accounted for 54% of TP loading to the lake, but they did not measure sedimentation losses. In my view, a balanced assessment of internal recycling should consider both nutrient releases from the sediments and sedimentation losses.

In the last 5 years, average TP concentration for Trout Lake has been the highest of all the basin lakes (202 μg TP/L, Figure 10). Chlorophyll-*a* has averaged 70 μg /L. These concentrations exceed the TMDL target TP concentration of 28 μg/L and the TMDL target chlorophyll-*a* concentration of 9.9 μg/L. Average TN concentration in Trout Lake over the last 5 years has been 1.7 mg/L, which also exceeds the TMDL target concentration of 0.78 mg/L. Trout Lake has differed from the other basin lakes in that TP concentrations and masses tended to increase during high-water periods over the period 2005-2015 (Figure 94), suggesting watershed runoff was a key process affecting in-lake TP levels. However, in more recent years with relatively high water levels TP concentrations and masses have tended to decrease. Other parameters tend to show increases in concentrations during low-water periods, particularly chlorophyll-*a* and TSS (Figure 95). Comparison of trends in concentrations and masses show some evidence of greater increases in chlorophyll-*a* concentrations during low-water periods, but not for other parameters

Water color has had a major influence on concentrations of chlorophyll-*a* and total suspended solids concentrations in Trout Lake. During periods of high rainfall water color substantially increased in the lake, particularly after Tropical Storm Faye in 2008, and chlorophyll-*a* and TSS concentrations decreased to low levels during these high color periods (Figure 96). There were no relationships between water color and concentrations of TP or TN. Water color produced by dissolved organic matter, particularly humic substances, reduces light availability, limiting phytoplankton growth. Also, absorption of solar radiation by organic matter can produce hydrogen peroxide and reactive oxygen species that can damage phytoplankton (Leunert et al. 2014). Chlorophyll-*a* and TSS are strongly correlated in Trout Lake, indicating that algal biomass makes up a substantial portion of the TSS.

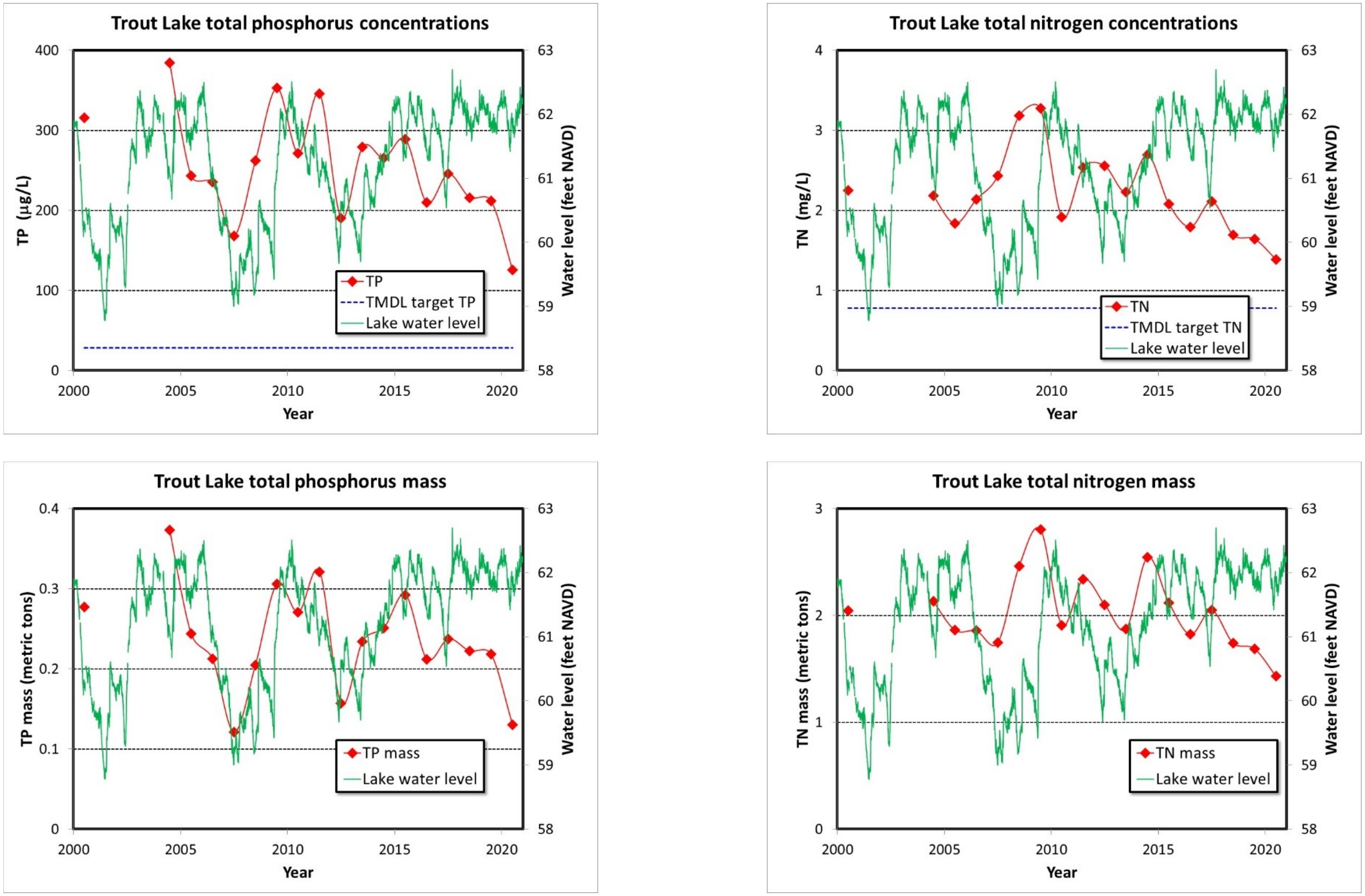


Figure 94. Trout Lake, mean annual total phosphorus and total nitrogen concentrations and masses, and water elevations.

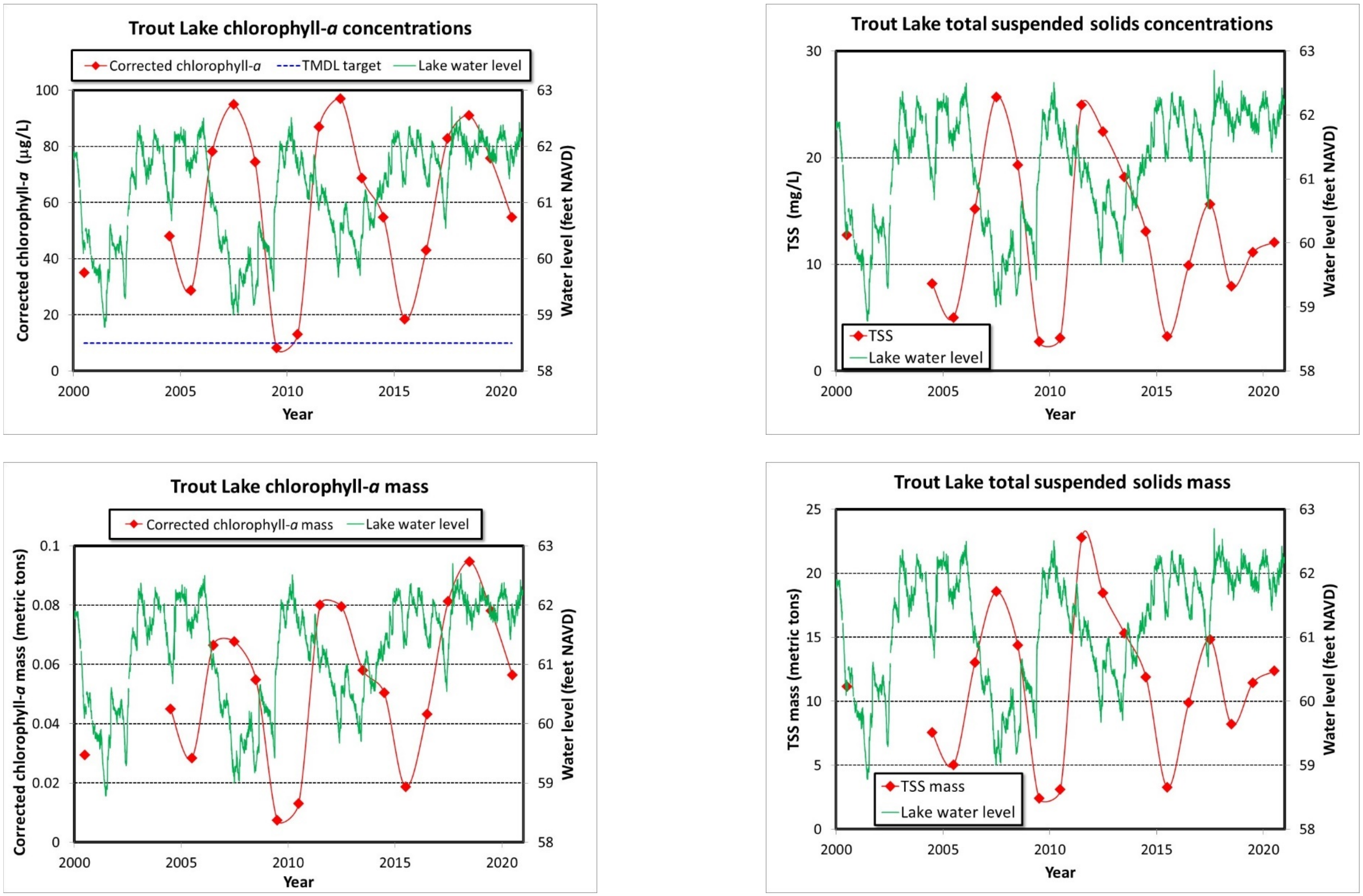


Figure 95. Trout Lake, mean annual chlorophyll-*a* and total suspended solids concentrations and masses, and water elevations.

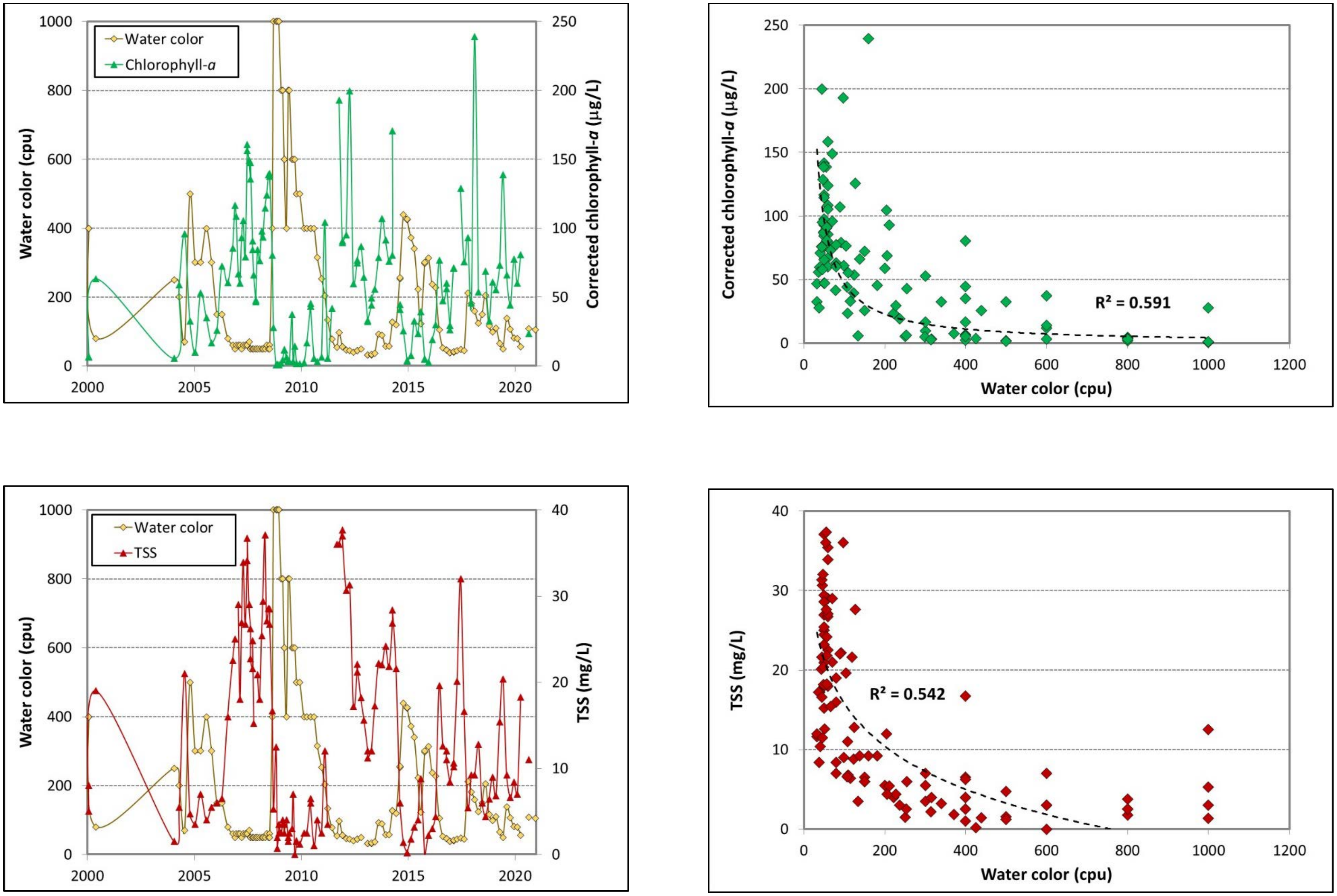


Figure 96. Relationships of water color with chlorophyll-*a* and TSS in Trout Lake.

Results of water quality trend analyses for Trout Lake are summarized in Table 14, with more detailed results including Kendall's tau nonparametric correlation coefficient and statistical significance level of the nonparametric Theil-Sen trend lines in Appendix Table B-11. Water quality trends for Trout Lake. Time series plots are shown for several parameters in Figure 97.

Table 14. Water quality trends for Trout Lake.

Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality, $p<0.05$

(0) No trend, $p\geq0.05$

(–) Significant decreasing trend, $p<0.05$

(+) Significant increasing trend, $p<0.05$

	2004 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, 0	S, 0
TN	0	0
Chlorophyll-*a*	0	0
TSS [a]	0	+
Secchi	+	0
NH_4	–	0
NH_4 [a]	–	–
NO_x [a]	0	0
PO_4 [a]	0	–
TP mass	S, 0	
TN mass	0	
Chlorophyll-*a* mass	0	
TSS mass	0	

[a] Non-detect methods

There were relatively few significant trends in the Trout Lake water quality data. There was a significant increasing trend in Secchi transparency, but there was not a significant trend in water elevation-adjusted residuals, suggesting the apparent trend was due to changing water levels. TSS showed an increasing trend in the water elevation-adjusted residuals, but not with the raw measurements or with TSS mass, suggesting that water level fluctuations masked an increasing trend, However, the significant trend in TSS residuals may have been because there was no correction for serial correlation using the non-detect methods. Although not shown because >15% of the TSS measurements were below MDL, I also did a trend analysis using the standard methods, and that showed a nearly identical Kendall's tau, but was not statistically significant with the correction for serial correlation. NH_4-D showed a decreasing trend using either standard or non-detect methods, but only using non-detect method with the water elevation-adjusted residuals. Again this significant trend in the residuals may have been because of the absence of correction for serial correlation with the non-detect method; Kendall's tau were nearly identical in the two analyses (Appendix Table B- 11). There were no exceedances of the Florida ammonia standard in the subset of Trout Lake data used for trend analyses, although one exceedance did occur in 2008. PO_4-D showed a decreasing trend in the water elevation-adjusted residuals, but not with the raw measurements, suggesting that water level fluctuations masked a decreasing trend.

Recently a hybrid wetland-alum treatment system has been operated at Trout Lake. I don't have information on its operations, but initial plans were to treat water flowing to Trout Lake through its main tributary, and later to also treat water within the lake. Although no significant trends in TP were evident in Trout Lake, TP concentrations have been relatively low since August 2019, all below 200 μg/L, with a minimum of 81 μg/L, perhaps reflecting effects of the treatment system.

Phytoplankton sampling ended in Trout Lake after August 2018. Phytoplankton biovolumes decreased substantially during high water color periods, as did chlorophyll-*a* (Figure 98). During the low biovolume periods, the phytoplankton tended to be dominated by taxa other than cyanobacteria, particularly diatoms (Figure 99). However, cyanobacteria tended to dominate during periods of high biovolume, and *Microcystis aeruginosa* tended to be the dominant cyanobacterium in this lake.

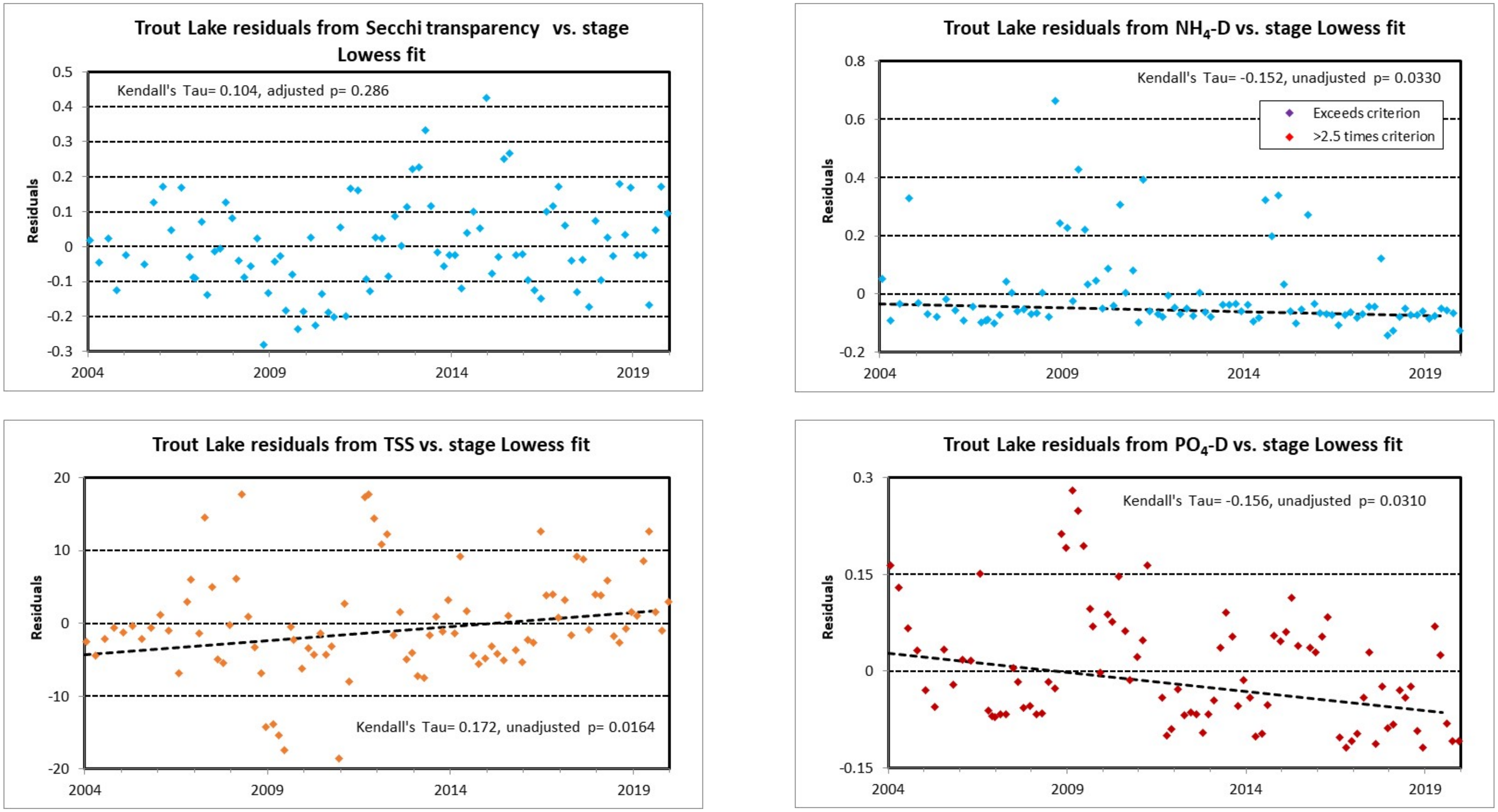


Figure 97. Water quality trends in Trout Lake, 2004-2019.

Daily means and Theil-Sen or ATS trend lines.

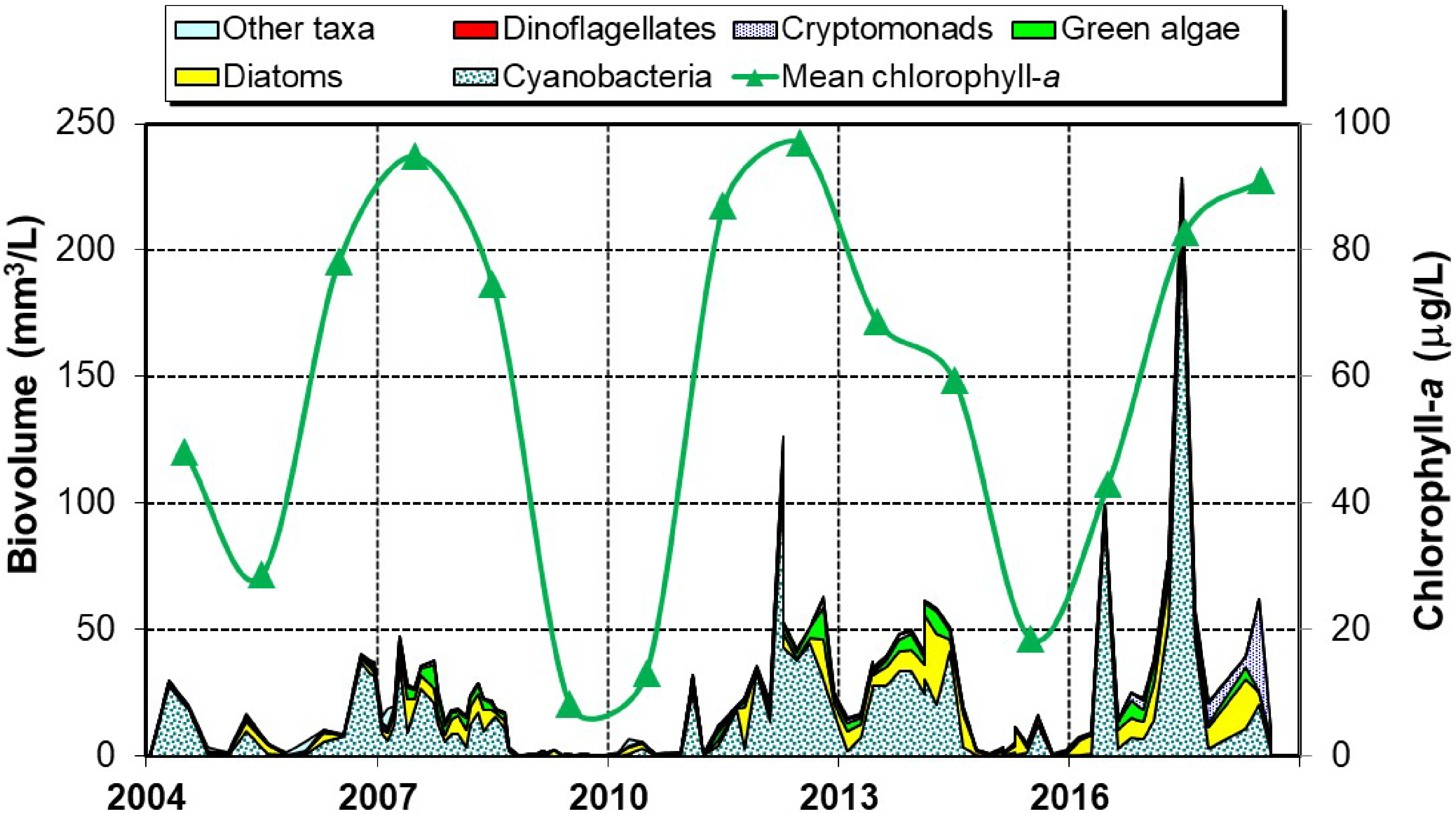


Figure 98. Trout Lake phytoplankton biovolumes and chlorophyll-*a* concentrations.

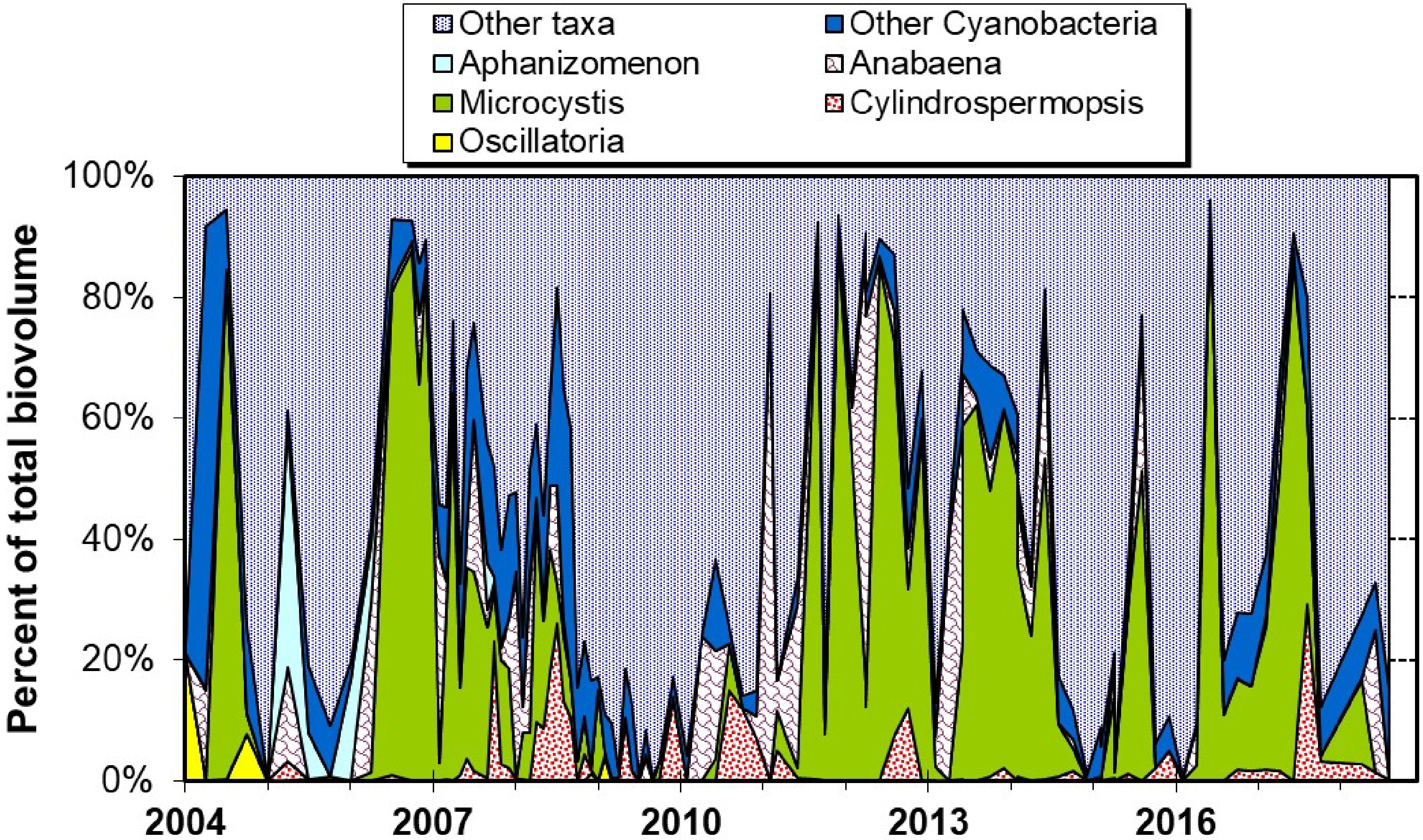


Figure 99. Trout Lake phytoplankton percent composition.

GENERAL DISCUSSION

Six of the basin lakes, Apopka, Beauclair, Dora, Harris, Eustis, and Griffin, have had decreases in total phosphorus loading and water quality improvements. The most significant water quality improvements have been in lakes Griffin, Beauclair, and Dora. Major restoration projects in Lake Apopka basin have reduced phosphorus loads to Lake Apopka (Figure 8). Projects upstream of Lake Beauclair have also reduced concentrations in tributary flows to lakes Beauclair and Dora, and the EMCA restoration has reduced phosphorus discharges to Lake Griffin. Lake Eustis is also affected by the projects upstream of Lake Beauclair, as well as by the Pine Meadows Restoration Area and the Lake Harris Conservation Area (Figure 9). Another factor contributing to reduced tributary loading to lakes in the chain was an extended period of low discharge volumes from upstream lakes, due to low rainfall in recent years. For example, during the 2006-2013 period, there was an estimated cumulative rainfall departure of –36.9 inches from normal for the area contributing to the UORB lakes (M. Daly, personal communication). Figure 100 shows a general decreasing trend in downstream discharges from Lake Eustis at the Burrell Lock and Dam on Haynes Creek; discharges at other basin structures show parallel trends. Tributary flows have increased over the last three years (and some of the discharges from upstream of the Burrell Dan were routed through Harris Bayou beginning in 2008, Figure 100), but nutrient concentrations in those flows have decreased due to the upstream restoration projects. Also, high inflow volumes result in shorter water residence time in the lakes and higher downstream discharges, so a substantial part of the tributary load during those periods is discharged downstream, rather than accumulating in the lakes.

The reasons for improvements in Lake Harris are less clear. The only major restoration project in the immediate watershed for Lake Harris is the LHCA, which reduced estimated TP loading by about 10% (Figure 9). There have been several stormwater projects implemented in the Lake Harris basin, but their estimated TP removal is much smaller than that by the LHCA. Lakes Harris and Eustis have shown parallel trends in water quality, with the primary improvements in water quality in both lakes occurring since 2007 (Figure 40, Figure 48). It is likely that backflows from Lake Eustis to Lake Harris are more significant than assumed, so Lake Harris also benefits from the projects upstream of Lake Eustis. My estimated backflows were based on net flows from monthly water budgets, which were very small. However, short-term reverse flows in the Dead River connecting the lakes are substantial. USGS measured daily flows in the Dead River from 1993-1996. USGS remarks about the measurements were that records were poor, and reported daily discharge represents the net of much larger upstream and downstream discharges. Nevertheless, daily discharge estimates ranged from –503 to +1180 cfs (with negative discharges representing net flow from Lake Eustis to Lake Harris). There were net negative discharges on 28% of the days of measurement.

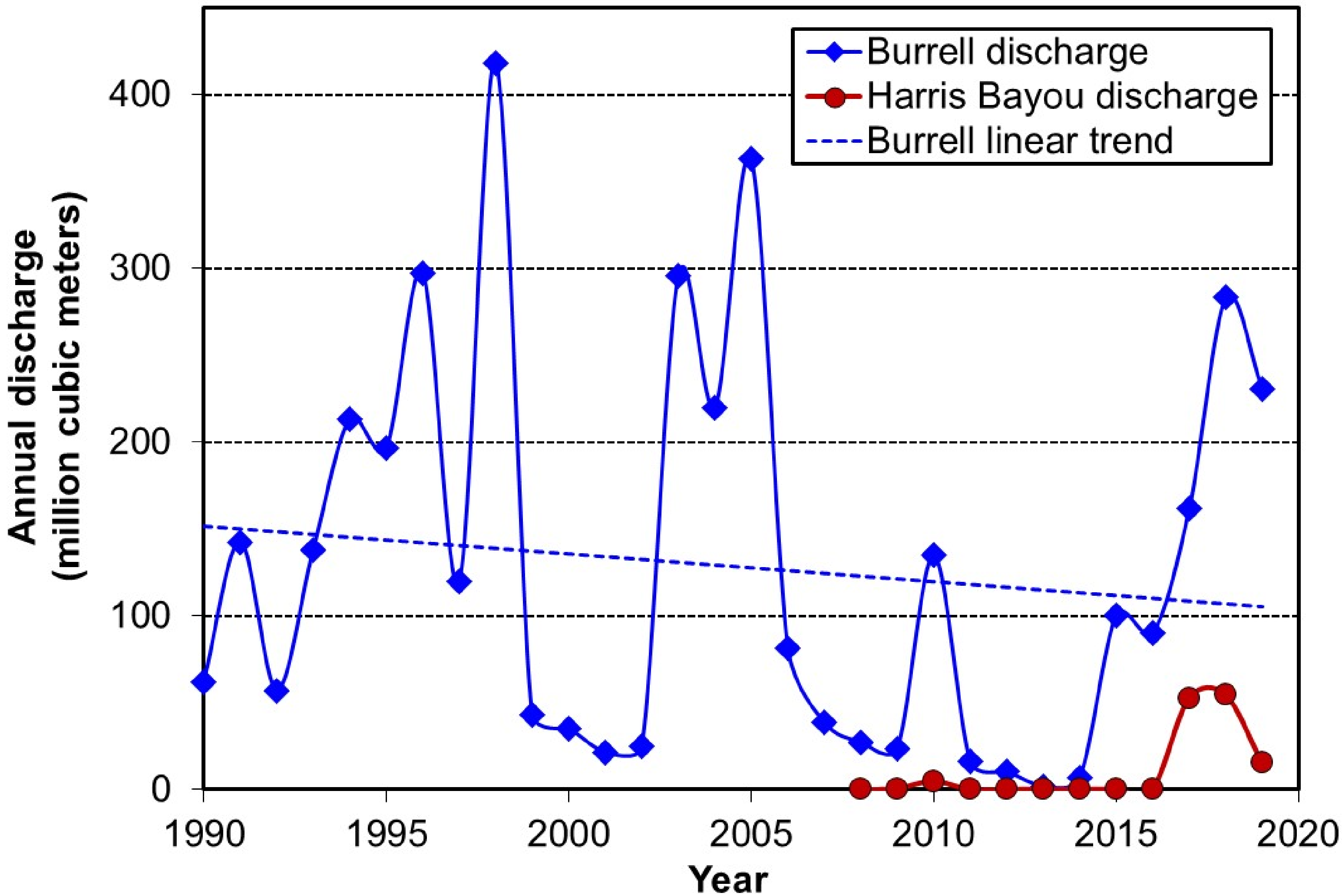


Figure 100. Annual downstream discharge volumes through Burrell Lock and Dam and Harris Bayou, 1990-2019.

Water quality has also improved in Lake Carlton, and to a smaller extent, in Lake Denham. There have been no restoration projects in the Lake Carlton watershed, but the lake is connected to Lake Beauclair through a very short canal, through which there are known to be reverse flows, so water quality in Lake Carlton likely benefits from the restoration projects upstream of Lake Beauclair. The only restoration project affecting Lake Denham was the experimental gizzard shad removal project in from 1990 to 1992. Over that period there appeared to be a substantial decrease in TP and chlorophyll-*a* concentrations in the lake (Figure 84, Figure 85). Since that time there have been some further improvements in TP concentrations. We have little information on operations of the muck farm in the Lake Denham watershed, although in a 2015 conference call with the Florida Department of Agricultural and Consumer Services it was stated the farm was farmed commercially until 2004-2005 and since that time has been a hobby farm. The sod is regularly mowed but is not fertilized. As noted previously, Lake County Water Authority has purchased the farm and plans to restore the property.

TP loading has substantially decreased and has been below or close to the TMDL targets for the last 5 years for lakes Apopka, Beauclair, Dora, Harris, Eustis, and Griffin (Figure 6). However, only lakes Dora, Harris, Eustis, and Griffin have had average TP concentrations close to their TMDL targets over the last 5 years (Figure 10). While average TP concentrations in lakes Apopka and Beauclair have decreased, they remain above their TMDL targets. There may not yet have been a long enough period of reduced external loadings to determine if the concentration targets will be met in all the lakes. In a survey of external load reduction case studies, internal recycling delayed reductions in nutrient concentrations, but generally new equilibria were reached for TP within 10 – 15 years (Jeppesen et al. 2005). There is evidence that TP concentrations are continuing to decrease. All these lakes except Apopka showed significant decreasing trends in TP concentrations over the 2001-2019 period. Although not yet reaching their TMDL target concentrations, lakes Apopka (Figure 15) and Beauclair (Figure 23) have approached those targets over the last 3 – 5 years.

The period covered by this report included dry periods, in which external nutrient loading was low, but also increased water residence times and lowered average water elevations, which could cause deterioration in water quality due to concentration of nutrients in a smaller volume of water and increased exchanges with the lake sediments. The basin lakes often show a trend toward poorer water quality, including higher TP concentrations, as water elevations decrease (Coveney 2016; Havens et al. 2019; Havens and Ji 2018). This is also evident from the comparison of trends in concentrations and masses, particularly for Lake Apopka, which showed substantial increases in concentrations during drought periods (Figure 17). There were some exceptions to the trend of poorer water quality during drought periods. During the drought in the early 2000s there was improved water quality in Lake Griffin (Figure 59, Figure 59) and lower TP concentrations (but not other parameters) in Lake Dora (Figure 34, Figure 34). In these cases, TP loading changes from restoration actions may have overridden the drought effects. Another exception was in Trout Lake, in which TP concentrations and masses tended to increase during high-water periods over the period 2005 – 2015 (Figure 94), suggesting watershed runoff was a key process affecting in-lake TP levels.

Water elevation changes appear to have had the most significant effects on water quality in Lake Apopka, where repeated multi-year drought periods since around 2000 led to cyclic changes in water quality, and in lakes Weir and Yale, where part of the deterioration in water quality appears to be due to a substantial decrease in lake water elevations. In Lake Apopka, water quality deteriorated during the severe droughts, but substantially improved during periods of more normal water levels. The shallow depth and high dynamic ratio of Lake Apopka (Table 1) make it susceptible to strong effects of water level fluctuations on water quality. To compare the water fluctuations among the lakes, I calculated annual averages for lake volumes and average depths from water elevations and morphometric data from the sources shown in Table 1. Over the period covered by this report, Lake Apopka has had the most substantial changes in absolute (maximum minus minimum annual averages) and relative (minimum divided by the maximum annual averages) water volumes and in relative mean depths, while lakes Weir and Yale have the largest changes in absolute mean depths (Figure 101). This is consistent with the conclusion that water level changes have had the most significant effects on these three lakes.

Although water quality has generally improved in Lake Apopka, the improvements in TSS and Secchi transparency have been relatively small. Recent levels for those parameters remain the worst among the basin lakes (Figure 12, Figure 14), and the trend analyses showed deterioration in water-level adjusted residuals during the 2001-2019 period (Table 4, Figure 20). Lake Apopka has the highest dynamic ratio of the basin lakes (Table 1), making it particularly susceptible to sediment resuspension. Lake Apopka is underlain by a layer of flocculent, easily resuspended sediments (Reddy and Graetz 1991; Schelske 1997; Mehta et al. 2009). From radiotracer analyses, Mehta et al. (2009) estimated that the upper 4 – 8 cm of sediments had been recently resuspended. Mehta et al. (2009) described two types of sediment resuspension in the lake. At high wind speeds there is transient resuspension of larger sediment particles. Under lower wind speeds there is persistent suspension of small or buoyant particles, consisting of both living phyto-bacterio-plankton and nonliving organic detritus, which they termed the ‘wash load’. This wash load would be what is measured as TSS, and is the primary factor limiting transparency and light availability for submersed aquatic vegetation. Amec (2018) concluded that non-algal suspended solids (NATSS) was the primary component of the wash load limiting light availability in the lake.

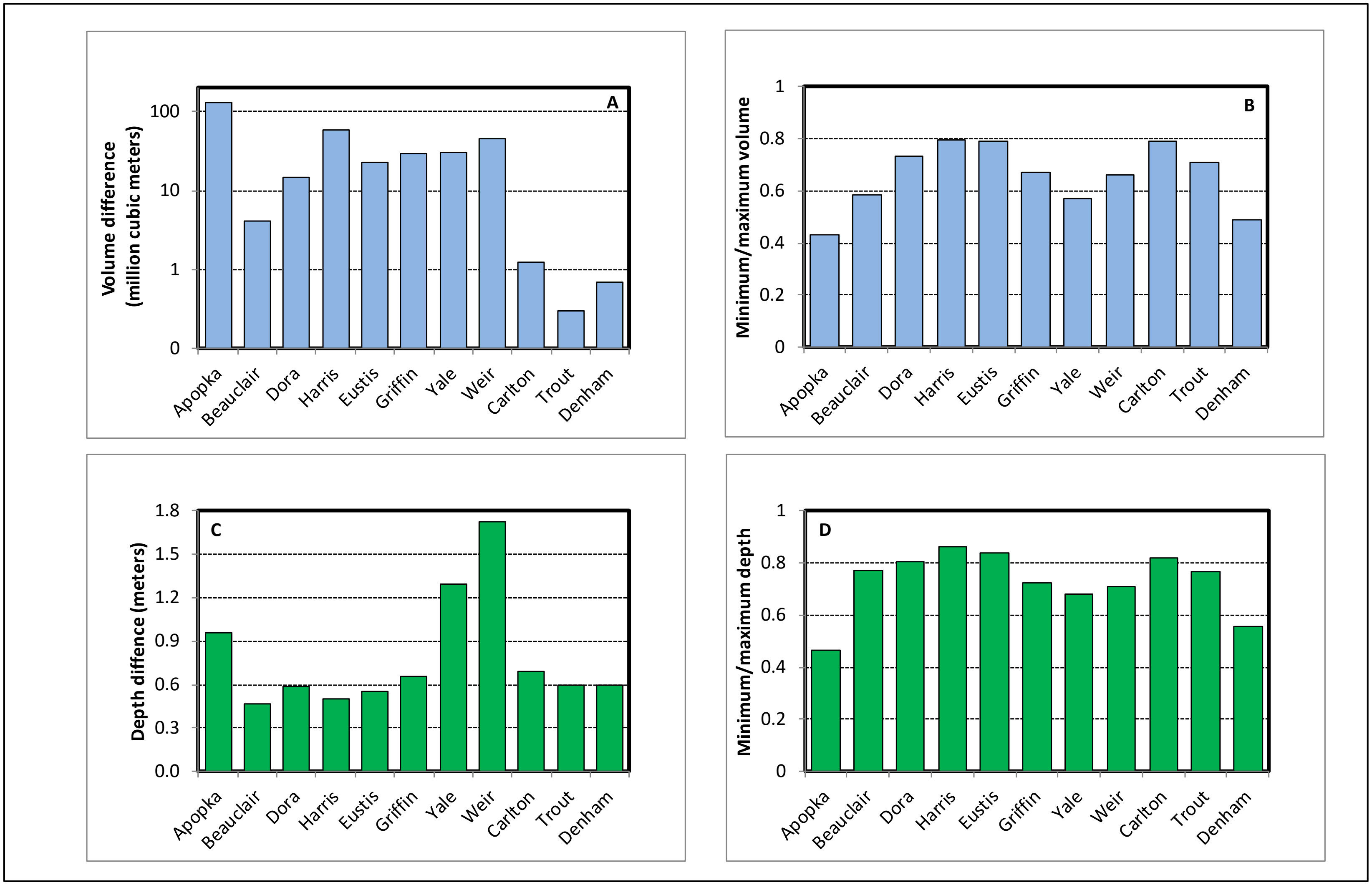


Figure 101. Changes in annual average lake volumes and mean depths.

A. Volume changes (maximum minus minimum annual averages). B. Relative volume changes (minimum divided by the maximum annual averages). C. Mean depth changes. D. Relative mean depth changes.

The deteriorating trends in the residuals from the Lowess smoothed relationships of lake water elevations with TSS and Secchi depth suggest that improvements in these parameters may require further intervention. Several projects have been in progress or proposed to try to reduce resuspended sediment concentrations. Rough fish harvesting, in addition to direct removal of nutrients, is expected to reduce sediment resuspension caused by feeding activities of the fish, although that cannot be quantified. Aquatic vegetation re-establishment, through both plantings and natural recruitment should reduce sediment resuspension, but again that cannot be quantified. The Lake Apopka marsh flow-way removes suspended sediments from water circulated through the system. A project pumping unconsolidated sediments from the lake into the LANS (sediment vacuuming) was operated from 2016-2019. A pilot project dredging a ‘sump’ near the north end of the lake, with the dredge spoils deposited in the LANS was conducted from 2017-2019. Future monitoring will determine if flocculent sediments migrate into the sump; if so they could be repeatedly removed. Plans have also been developed for targeted dredging of easily resuspended sediments (Wood Environment & Infrastructure Solutions, Inc. 2019). Table 15 summarizes masses of resuspended and potentially resuspended sediments in the lake, and quantified removals. Quantified removals by the Lake Apopka Marsh Flow-way and sediment vacuuming projects are substantial compared to TSS masses in the water column but are small compared to estimated annual and total accumulations of flocculent sediments. Sediment accumulation and masses may now be lower than estimated by Schelske (1997). A model developed by Pollman (2016) predicts that if external TP loadings remain low, the sediment layer that is the primary source of resuspension could become effectively depleted within approximately 40 years.

Table 15. Estimated sediment masses and quantified removals for Lake Apopka.

Sediment Component	Metric Tons
Water column TSS (annual means 1987-2019)	6,700 – 21,800
Average annual accumulation of flocculent sediments (Schelske 1997)	44,200
Mass of top 5 cm of lake flocculent sediments (calculated from Schelske 1997)	62,800
Total mass of lake flocculent sediments (Schelske 1997)	2,210,000
TSS removal by Marsh Flow-way (annual means 2004-2019)	3,700
TSS removal by sediment vacuuming (annual means 2016-2019)	260

Water quality has deteriorated in lakes Yale and Weir, which are unaffected by the major basin restoration projects. However, it should be noted that these lakes still have among the best water quality of the basin lakes (Figure 10 – Figure 14). Water quality in Lake Yale deteriorated substantially in the mid-1990s following grass carp stocking and a loss of aquatic vegetation in the lake. There have been efforts to remove the grass carp and in recent years there has been some recovery of aquatic vegetation, and some indications of improvements in water quality. Lake Yale is not projected to meet its TMDL loading target in the UORB BMAP (Upper Ocklawaha Basin Working Group 2007). A recently completed study has further examined nutrient sources and developed restoration alternatives for Lake

Yale (ERD 2017). Lake County Water Authority is considering a whole-lake alum treatment of Lake Yale, one of the recommendations of ERD (2017).

Although there have been increases in estimated TP loading to Lake Weir, analyses conducted in this report (Table 11) suggest that the deterioration in water quality in the lake is in part due to decreases in water elevations. An extended return to higher rainfall may result in higher lake water elevations and improved water quality in Lake Weir. Marion County has been working to develop a management plan for the lake.

The potential nutrient impacts of waterfowl excretion depend on the locations of feeding and roosting. If birds both feed and roost on a lake they recycle nutrients already in lake, but don’t change the nutrient load. If the birds feed on a lake, but roost elsewhere, they export nutrients from the lake. Of most concern is the situation where birds roost on a lake but feed elsewhere, then they import nutrients to the lake. Area residents believe that is the situation for Lake Weir, with the gull populations feeding at Marion County’s Baseline landfill and roosting on the lake. The landfill is about 14 km from Lake Weir, which is well within the flying range of foraging gulls (Patenaude-Monette et al. 2014).

The most geographically relevant study of effects of waterfowl excretion on lake water quality was by Hoyer and Canfield (1994), which included 14 Florida lakes, although none of these had large gull populations. They estimated that the TP load to the lakes from bird populations ranged from 0.1% to 9% of external loads and felt that these were probably overestimates because most birds were feeding on the lakes, and therefore primarily recycling nutrients already in the lakes. They concluded bird populations generally do not significantly affect nutrient loading to Florida lakes under “natural conditions”, although they did allow for the potential of significant impacts if large populations feed outside a lake and roost on the lake.

Another study of a situation perhaps more similar to Lake Weir was done by Portnoy (1990), who studied effects of a gull population on a small Cape Cod lake. He noted gull populations have increased dramatically as result of increased winter survival around urban landfills. The study determined that the gulls were not feeding on the lake. This lake was much smaller than Lake Weir (about 109 acres, whereas Lake Weir is >5,000 acres). Gull populations exceeded 2,000 birds on the Cape Cod lake, although they were not present in such numbers all year long. Portnoy (1990) estimated that gulls contributed 42% of total phosphorus inputs to lake and concluded that large gull concentrations can be important source in lake eutrophication.

The estimates of potential nutrient loading to Lake Weir from gull populations (Figure 76) indicate waterfowl excretion could be a significant source of phosphorus to the lake. However, these estimates are dependent on the assumptions that the birds feed elsewhere and roost on the lake, and that thousands of gulls are present for much of the year. Year-round surveys of the bird populations and their feeding and roosting activities are necessary to verify these assumptions.

There are few clear trends in phytoplankton biovolumes, unlike the decreasing trends in chlorophyll-*a* in many of the lakes. Only Lake Griffin seems to show a clear decrease in peak phytoplankton biovolumes. Generally, there were poor relationships between chlorophyll-*a* and phytoplankton biovolume (Figure 102). Chlorophyll probably is a better estimate of algal biomass than biovolumes. For one reason, chlorophyll measurements are based on a much larger sample volume, often several hundred milliliters, while biovolume estimates are based on counting cells in a fraction of a milliliter. Biovolume estimates are also approximations based on similarity of the cells to standard geometrical shapes. Biovolume estimates may be particularly difficult for colonial or filamentous species, which are among the dominant taxa in basin lakes. Also, there appear to be substantial differences in measurements by different phytoplankton analysts. Initial comparisons of replicated analyses by Phycotech and BSA showed substantial differences in both composition and biovolume estimates. We did two sets of comparisons. In the first round, we had BSA re-analyze 10 samples previously analyzed by Phycotech. In the second round, both firms analyzed 20 new samples. In the first round, Phycotech reported consistently greater cyanobacteria biovolume, averaging about twice that for BSA. The second-round counts were nearly the opposite, BSA averaging twice the cyanobacterial biovolumes reported by Phycotech. For several individual samples in both rounds, differences in cyanobacterial biovolumes between the two firms were 3- to nearly 6-fold. I conducted Kruskal-Wallis tests comparing medians for cyanobacterial biovolumes from the two analyst firms for each lake (these were not for the replicate analyses, but for the period of analyses by each firm). These analyses showed BSA found significantly higher biovolumes for lakes Beauclair, Dora, Harris, Denham, and Carlton, and no significant differences between analysts for the other lakes. These Kruskal-Wallis analyses confound analyst firm and temporal differences, but the apparent higher biovolumes in the later BSA analyses for lakes Beauclair, Dora, Harris, and Carlton conflict with the significant decreasing trends for chlorophyll-*a* in these lakes.

Although some of the lakes have seen decreases in phytoplankton biovolumes, the phytoplankton have still been dominated by cyanobacteria during the warm season. Even Lake Weir, which has TP concentrations that are lower than the TMDL targets for the other lakes, still is typically dominated by cyanobacteria (Figure 82). Cyanobacterial dominance is likely the natural condition for these lakes, so even if the TMDL targets are met, we can still expect that the phytoplankton of these lakes will continue to be dominated by cyanobacteria, at least during the warm season.

There were some substantial differences in cyanobacterial composition among lakes. Most of the lakes tended to be dominated by *Cylindrospermopsis*, or co-dominated by *Cylindrospermopsis* and *Pseudanabaena*, depending on the phytoplankton analyst. However, Lake Apopka tended to be dominated by *Planktolyngbya*, and in the BSA analyses with substantial representation by *Raphidiopsis*. These are all filamentous cyanobacteria that are tolerant to low light levels (Havens et al. 1998; Burford et al. 2016). As noted previously, recent genetic studies indicate *Raphidiopsis* and *Cylindrospermopsis* should be considered the same genus (Aguilera et al. 2018), but it is not certain whether these two taxa in the Ocklawaha lakes are different species or different morphological forms of the same species. In the past *Raphidiopsis* was sometimes distinguished by the absence of heterocysts (specialized cells for nitrogen fixation), although that is not how BSA distinguished these taxa, and only some of the *Cylindrospermopsis* identified by BSA had heterocysts. One possible explanation for the greater prominence of *Raphidiopsis* in Lake Apopka is higher nitrogen availability in that lake. Baseline and recent TN concentrations have been highest in Lake Apopka (Figure 13), although baseline concentrations in some of the lakes exceed recent TN concentrations in Lake Apopka.

Another difference among lakes has been the dominance by *Microcystis* in Trout Lake. *Microcystis* is adapted to high light levels and is favored by a stratified water column that allows the buoyant cells to aggregate near the water surface (Havens et al. 1998, Briand et al. 2004, Harke et al. 2016). Trout Lake has the smallest surface area and dynamic ratio of the basin lakes (Table 1), which may indicate a greater likelihood of water column stratification. *Microcystis* may also be favored by the very high TP concentrations in Trout Lake (Figure 10), although it also requires nitrogen, which is not particularly high in Trout Lake (Figure 13).

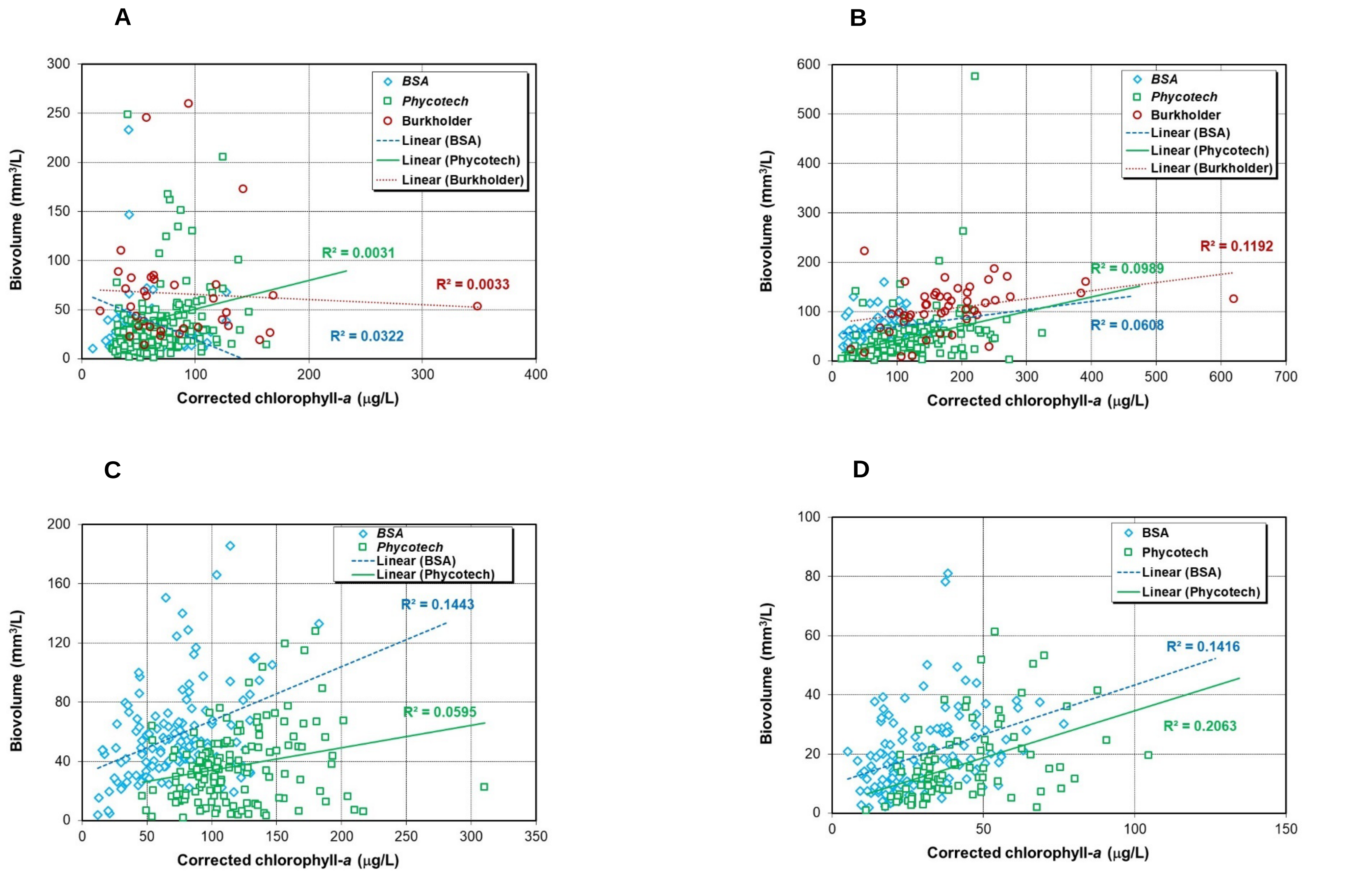


Figure 102. Relationships between chlorophyll-*a* and phytoplankton biovolume measurements.

A. Lake Apopka. B. Lake Beauclair. C. Lake Dora. D. Lake Harris.

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APPENDIX A. FDEP AND SJRWMD DATA QUALIFIER CODES

The following list and definitions of FDEP Data Qualifier Codes come from this website:

https://www.flrules.org/Gateway/View_notice.asp?id=14839077

The following codes shall be used by laboratories and/or field organizations when reporting sample data values that either meet the specified descriptions outlined below or do not meet the applicable quality control criteria specified for the laboratory or field result. Data qualifier codes listed in summary reports or other presentations comprising information that has been reformatted from original reports generated by field or laboratory organizations or individuals shall meet the requirements of subsections 62-160.240(4) and 62-160.340(7), F.A.C. Data qualifier codes added to sample results during data review procedures conducted by organizations or individuals other than the generators of original reports shall meet the requirements of subsections 62-160.240(5) and 62-160.340(8), F.A.C.

CODE	DEFINITION
A	Value reported is the arithmetic mean (average) of two or more determinations. This code shall be used if the reported value is the average of results for two or more discrete and separate samples. These samples shall have been processed and analyzed independently. Do not use this code if the data are the result of replicate analysis on the same sample aliquot, extract or digestate.
B	Results based upon colony counts outside the acceptable range. This code applies to microbiological tests and specifically to membrane filter colony counts. The code is to be used if the colony count is generated from a plate in which the total number of coliform colonies is outside the method indicated ideal range. This code is not to be used if a 100 mL sample has been filtered and the colony count is less than the lower value of the ideal range.
F	When reporting species: F indicates the female sex.
H	Value based on field kit determination; results may not be accurate. This code shall be used if a field screening test (e.g., field gas chromatograph data, immunoassay, or vendor-supplied field kit) was used to generate the value and the field kit or method has not been recognized by the Department as equivalent to laboratory methods.
I	The reported value is greater than or equal to the laboratory method detection limit but less than the laboratory practical quantitation limit.

CODE	DEFINITION
J	Estimated value. A "J" – qualified sample value shall be accompanied by a detailed explanation to justify the reason(s) for designating the value as estimated. Where possible, the organization shall report whether the actual sample value is estimated to be less than or greater than the reported value, to assist data users in any evaluation of the usability of the sample value. A "J" data qualifier code shall not be used as a substitute for G, K, L, M, S, T, V, or Y, however, if additional reasons exist for identifying the value as an estimate (e.g., laboratory control spike or matrix spiked failed to meet acceptance criteria), the "J" code may be added to a G, K, L, M, T, U, V, or Y qualifier. Examples of situations in which a "J" code must be reported include: instances where a quality control item associated with the reported value failed to meet the established quality control criteria (the specific failure must be identified); instances when the sample matrix interfered with the ability to make any accurate determination; instances when data are questionable because of improper laboratory or field protocols (e.g., composite sample was collected instead of a grab sample); instances when the analyte was detected at or above the method detection limit in an analytical laboratory blank other than the method blank (such as a calibration blank) and, the blank value is greater than 10% of the associated sample value; or, instances when the field or laboratory calibrations or calibration verifications did not meet calibration acceptance criteria, including quantitative or chronological bracketing requirements for field testing data.
K	Off-scale low. Actual value is known to be less than the value given. This code shall not be used for microbiological tests or for biochemical oxygen demand. This code shall not be used for field-testing measurements where quantitative bracketing is required. This code shall be used if:
	1. The value is less than the lowest calibration standard and the calibration curve is known to be non-linear; or
	2. The value is known to be less than the reported value based on sample size, dilution.
	This code shall not be used to report values that are less than the laboratory practical quantitation limit or laboratory method detection limit.
L	Off-scale high. Actual value is known to be greater than value given. This code shall not be used for microbiological tests or biochemical oxygen demand. This code shall not be used for field-testing measurements where quantitative bracketing is required. To be used when the concentration of the analyte is above the acceptable level for quantitation (exceeds the linear range or highest calibration standard) and the calibration curve is known to exhibit a negative deflection.
M	When reporting chemical analyses: presence of material is verified but not quantified; the actual value is less than the value given. The reported value shall be the laboratory practical quantitation limit. This code shall be used if the level is too low to permit accurate quantification, but the estimated concentration is greater than or equal to the method detection limit. If the value is less than the method detection limit use "T" below.
N	Presumptive evidence of presence of material. This qualifier shall be used if:
	1. The component has been tentatively identified based on mass spectral library search; or
	2. There is an indication that the analyte is present, but quality control requirements for confirmation were not met (i.e., presence of analyte was not confirmed by alternative procedures).
O	Sampled, but analysis lost or not performed.
Q	Sample held beyond the accepted holding time. This code shall be used if the value is derived from a sample that was prepared or analyzed after the approved holding time restrictions for sample preparation or analysis.

CODE	DEFINITION
T	Value reported is less than the laboratory method detection limit. The value is reported for informational purposes only and shall not be used in statistical analysis.
U	Indicates that the compound was analyzed for but not detected. This symbol shall be used to indicate that the specified component **was not** detected. The value associated with the qualifier shall be the laboratory method detection limit. Unless requested by the client, less than the method detection limit values shall not be reported (see “T” above).
V	A “V” – qualified sample value indicates that the analyte was detected at or above the method detection limit in both the sample and the associated method blank and the blank value was greater than 10% of the associated sample value. The 10% criterion shall not apply to blank results for biochemical oxygen demand (BOD) or microbiological tests. For BOD tests, the “V” code shall be used for all sample results where the associated method blank result exceeds the maximum blank DO depletion specified in the analytical method. For microbiological tests, the “V” code shall be used for all samples where the associated method blank indicates growth of the target organism. Note: unless specified by the method, the value in the blank shall not be subtracted from associated samples.
X	Indicates, when reporting results from a Stream Condition Index Analysis (SCI 1000), that insufficient individuals were present in the sample to achieve a minimum of 280 organisms for identification (the method calls for two aliquots of 140-160 organisms), suggesting either extreme environmental stress or a sampling error.
Y	The laboratory analysis was from an improperly preserved sample. The data may not be accurate.
Z	Too many colonies were present for accurate counting. Historically, this condition has been reported as “too numerous to count” (TNTC). The “Z” qualifier code shall be reported when the total number of colonies of all types is more than 200 in all dilutions of the sample. When applicable to the observed test results, a numeric value for the colony count for the microorganism tested shall be estimated from the highest dilution factor (smallest sample volume) used for the test and reported with the qualifier code. Atypical, non-target, spreading colonies or other interferences may prevent estimation of typical target organism counts.
?	Data are rejected and should not be used. Some or all of the quality control data for the analyte were outside criteria, and the presence or absence of the analyte cannot be determined from the data.
*	Not reported due to interference.

The following codes deal with certain aspects of field activities. The codes shall be used if the laboratory has knowledge of the specific sampling event. The codes shall be added by the organization collecting samples if they apply:

CODE	DEFINITION
D	Measurement was made in the field (i.e., in situ). This code applies to any value (**except** field measurements of pH, specific conductance, dissolved oxygen, temperature, total residual chlorine, transparency, turbidity or salinity) that was obtained under field conditions using approved analytical methods. If the parameter code specifies a field measurement (e.g., "Field pH"), this code is not required.
E	Indicates that extra samples were taken at composite stations.
G	A "G" – qualified sample value indicates that the analyte was detected at or above the method detection limit in both the sample and the associated field blank, equipment blank, or trip blank, and the blank value was greater than 10% of the associated sample value. The value in the blank shall not be subtracted from associated samples.
R	Significant rain in the past 48 hours. (Significant rain typically involves rain in excess of 1/2 inch within the past 48 hours.) This code shall be used when the rainfall might contribute to a lower or higher than normal value.
S	Secchi disk visible to bottom of waterbody. The value reported is the depth of the waterbody at the location of the Secchi disk measurement.
!	Data deviate from historically established concentration ranges.

Rulemaking Authority 403.061, 403.0623 FS. Law Implemented 373.026, 373.309, 373.409, 373.413, 373.414, 373.416, 373.4592, 376.303, 376.305, 376.3071, 403.0623, 403.0625, 403.087, 403.088, 403.0881, 403.504, 403.704, 403.707, 403.722, 403.853 FS. History–New 1-1-91, Amended 2-4-93, 2-27-94, Formerly 17-160.700, Amended 3-24-96, 4-9-02, 6-8-04, 12-3-08, 7-30-14.

The following are SJRWMD Internal Data Qualifier Codes:

CODE	DEFINITION
W	Value observed is less than lowest value reportable under T code. This code is used when a positive value is not observed or calculated for a result, i.e. the test instrument or calculation is not capable of producing negative values. In these cases, the lowest reportable value, which is the lowest positive value that is observable, is reported with the W.
X	Value is for a quasi vertically-integrated sample.
>	Field Blank analyte value is high (> 2 x MDL) and has been confirmed by rerun. The same analyte for all environmental samples associated with the Field Blank (those samples collected by the same sample collection team using the same collection device on the same date of collection) shall be assigned the > code.
#	See the accompanying narrative explanation for important information from the Project/Data Manager.

APPENDIX B. SUMMARY OF WATER QUALITY TREND ANALYSES.

Table B- 1. Water quality trends for Lake Apopka.

Kendall's Tau and significance levels. Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality at $p<0.05$

	1987 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, –0.420, p= 0.0000267	S, –0.417, p= 0.0000202	–0.185, p= 0.127	S, +0.0189, p= 0.817
TN	S, –0.168, p= 0.0773	S, –0.267, p= 0.00299	–0.181, p= 0.139	S, +0.0264, p= 0.704
Chlorophyll-*a*	S, –0.253, p= 0.00292	S, –0.271, p= 0.00133	–0.105, p= 0.406	+0.025, p= 0.789
TSS	S, –0.0710, p= 0.417	S, –0.111, p= 0.200	+0.0099, p= 0.936	+0.199, p= 0.0173
Secchi	S, +0.312, p= 0. 000762	S, +0.325, p= 0. 000422	S, +0.0439, p= 0.730	S, –0.202, p= 0.0327
NH_4 [a]	+0.0480, p= 0.253	–0.0347, p= 0.403	–0.284, p= 0.000063	–0.0375, p= 0.583
NH_4 [a, b, c]	–0.0439, p= 0.217	–0.0236, p= 0.530		
NO_x [a, b]	–0.226, p= 0.0000000	–0.163, p= 0.0000166	–0.0760, p= 0.0328	–0.0569, p= 0.262
PO_4 [a, b]	–0.0787, p= 0.000891	–0.0795, p= 0.0352	–0.0351, p= 0.115	–0.0154, p= 0.750
TP mass	–0.432, p= 0.0000000		+0.0154, p= 0.868	
TN mass	–0.306, p= 0.0000008		+0.0687, p= 0.382	
Chlorophyll-*a* mass	–0.299, p= 0.0000077		+0.0933, p= 0.343	
TSS mass	–0.129, p= 0.143		+0.190, p= 0.0837	

[a] Dissolved fraction for both time periods

[b] Non-detect methods

[c] NH4-D not analyzed using non-detect methods for 2001-2019 because <5% below MDL

Table B- 2. Water quality trends for Lake Beauclair.

Kendall's Tau and significance levels. Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality at $p<0.05$

	1989 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	–0.647, p= 0.0000000	S, –0.569, p= 0.0000001	S, –0.597, p= 0.000018	S, –0.528, p= 0.0000535
TN	–0.456, p= 0.0000017	–0.550, p= 0.0000000	–0.489, p= 0.000972	–0.469 p= 0.00000173
Chlorophyll-*a*	–0.439, p= 0.0000000	S, –0.450, p= 0. 0000017	–0462, p= 0.0000015	–0.361, p= 0.0000014
TSS	–0.405, p= 0.0000000	–0.441, p= 0.0000000	–0.334, p= 0.00000256	–0.220, p= 0.0000976
Secchi	+0.463, p= 0.0000001	+0.509, p= 0.0000000	+0.422, p= 0.000129	+0.330, p= 0.0000148
NH_4 [a]	–0.0822, p= 0.0683	S, –0.141, p= 0.0947	–0.231, p= 0.0000713	S, –0.208, p= 0.0901
NH_4 [a, b]	–0.0671, p= 0.0792	S, –0.114, p= 0.00274	–0.226, p= 0.0000059	S, –0.181, p= 0.000279
NO_x [a, b]	0.0551, p= 0.132	–0.0728, p= 0.0588	S, –0.0621, p= 0.114	–0.104, p= 0.0392
PO_4 [a, b]	–0.0664, p= 0.0104	–0.0503, p= 0.189	–0.120, p= 0.0000065	–0.102, p= 0.0392
TP mass	S, –0.676, p=0.0000000		S, –0.533, p=0.0000617	
TN mass	–0.605, p=0.0000000		–0.535, p=0.00000011	
Chlorophyll-*a* mass	–0.495, p=0.0000000		–0.444, p=0.00000002	
TSS mass	–0.484, p=0.0000000		–0.332, p=0.00000003	

[a] Dissolved fraction for both time periods.

[b] Non-detect methods

Table B- 3. Water quality trends for Lake Dora.

Kendall's Tau and significance levels. Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality at p<0.05

	1986 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, –0.589, p=0.00000017	S, –0.438, p=0.0000198	S, –0.408, p=0.00131	S, –0.344, p=0.00105
TN	–0.531, p=0.00000008	–0.542, p=0.00000000	–0.567, p=0.0000792	–0.534, p=0.00000018
Chlorophyll-*a*	–0.368, p=0.000363	–0.357, p=0.0000981	–0.571, p=0.00000005	–0.494, p=0.00000002
TSS	–0.474, p=0.00000000	–0.441, p=0.00000000	–0.374, p=0.00000002	S, –0.357, p=0.000743
Secchi	+0.322, p=0.0113	+0.320, p=0.00140	+0.564, p=0.0000289	+0.455, p=0.0000399
NH_4 [a]	S, –0.0807, p=0.269	–0.0897, p=0.159	–0.100, p=0.351	–0.0142, p=0.892
NH_4 [a, b]	S, –0.0862, p=0.0758	–0.114, p=0.0189	–0.0899, p=0.0610	+0.0101, p=0.833
NO_x [a, b]	+0.0253, p=0.603	+0.0211, p=0.668	+0.0201, p=0.504	+0.0105, p=0.823
PO_4 [b]	ND [c]	ND	–0.0452, p=0.0188	–0.0449, p=0.332
TP mass	S, –0.577, p=0.00000013		S, –0.388, p=0.000197	
TN mass	–0.616, p=0.0000000		–0.619, p=0.00000000	
Chlorophyll-*a* mass	–0.389, p=0.0000320		–0.576, p=0.00000000	
TSS mass	–0.519, p=0.00000000		–0.379, p=0. 00000000	

[a] Synthetic total fraction for 1986-2019; dissolved fraction for 2001-2019

[b] Non-detect methods

[c] ND – No data

Table B- 4. Water quality trends for Lake Harris-Little Harris.

Kendall's Tau and significance levels. Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality at $p<0.05$

	1990 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, –0.216, p=0.0229	S, –0.236, p=0.00956	–0.450, p=0.00000001	–0.377, p=0.00000005
TN	S, –0.429, p=0.000130	S, –0.432, p=0.0000198	–0.356, p=0.0182	–0.269, p=0.0293
Chlorophyll-*a*	S, –0.480, p=0.00000740	S, –0.468, p=0.00000148	–0.401, p=0.00365	S, –0.371, p=0.00166
TSS	S, –0.223, p=0.0314	S, –0.221, p=0.0158	–0.384, p=0.0000107	–0.243, p=0.00106
Secchi	S, +0.329, p=0.00209	S, +0.366, p=0.0000911	+0.327, p=0.00446	S, +0.236, p=0.0334
NH_4 [a, b]	–0.104, p= 0.0411	–0.0381, p=0.454	S, +0.00677, p=0.885	S, +0.0760, p=0.104
NO_x [a, b]	S, –0.0425, p=0.411	S, –0.0848, p=0.113	S, – 0.0474, p=0.231	S, –0.103, p= 0.0286
PO_4 [b]	ND [c]	ND	– 0.0288, p=0.0421	– 0.0274, p=0.554
TP mass	S, –0.222, p=0.0155		–0.420, p=0.00000000	
TN mass	S, –0.441, p=0.0000302		–0.300, p=0.0229	
Chlorophyll-*a* mass	S, –0.515, p=0.00000118		–0.388, p=0.00112	
TSS mass	S, –0.252, p=0.00904		–0.362, p=0.00000285	

[a] Synthetic total fraction for 1990-2019; dissolved fraction for 2001-2019

[b] Non-detect methods

[c] ND – No data

Table B- 5. Water quality trends for Lake Eustis.

Kendall's Tau and significance levels. Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality at $p<0.05$

	1990 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, –0.310, p=0.00306	S, –0.306, p=0.00244	–0.431, p=0.0000315	–0.352, p=0.000077
TN	–0.442, p=0.000085	–0.456, p=0.00000207	–0.462, p=0.0149	–0.409, p=0.00076
Chlorophyll-*a*	–0.323, p=0.000485	–0.335, p=0.000141	S, –0.416, p=0.00196	S, –0.325, p=0.00296
TSS	–0.287, p=0.00711	–0.263, p=0.00320	–0.367, p=0.0208	–0.214, p=0.0886
Secchi	+0.344, p=0.0000312	+0.336, p=0.0000131	S, +0.356, p=0.00453	S, +0.258, p=0.0149
NH_4 [a, d]	S, –0.0836, p=0.276	–0.128, p=0.0437		
NH_4 [a, b]	S, –0.113, p=0.0280	S, –0.129, p=0.0125	S, –0.0849, p=0.0712	S, –0.0748, p=0.112
NO_x [a, b]	–0.0694, p=0.171	–0.105, p=0.0438	S, –0.0154, p=0.679	S, –0.0470, p=0.320
PO_4 [b]	ND [c]	ND	–0.0666, p=0.00190	–0.0498, p=0.283
TP mass	S, –0.298, p=0.00447		–0.384, p=0.0000537	
TN mass	–0.459, p=0.0000105		–0.426, p=0.00660	
Chlorophyll-*a* mass	–0.333, p=0.000227		S, –0.416, p=0.00162	
TSS mass	–0.285, p=0.00283		–0.332, p=0.0207	

[a] Synthetic total fraction for 1990-2019; dissolved fraction for 2001-2019

[b] Non-detect methods

[c] ND – No data

[d] NH_4 not analyzed by standard methods for 2001-2019 because >15% below MDL

Table B- 6. Water quality trends for Lake Griffin.

Kendall's Tau and significance levels. Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality at $p<0.05$

	1983 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	–0.526, p=0.0000000	–0.496, p=0.0000000	S, –0.458, p=0.000175	S, –0.389, p=0.000555
TN	–0.493, p= 0.0000886	–0.503, p= 0.0000031	–0.483, p=0.0000936	S, –0.502, p=0.000159
Chlorophyll-*a*	–0.367, p= 0.00423	–0.325, p= 0.00775	–0.340, p=0.00113	S, –0.322, p=0.00702
TSS	–0.401, p= 0.0000091	–0.404, p= 0.00000120	–0.346, p=0.0000820	–0.282, p=0.000316
Secchi	+0.331, p= 0.0000185	+0.351, p= 0.00000059	+0.344, p=0.00395	+0.258, p=0.0196
NH_4 [a, d]	S, –0.132, p=0.0685	–0.126, p=0.0821		
NH_4 [a, b]	S, –0.134, p=0.00316	S, –0.112, p=0.0140	S, –0.0961, p=0.0344	S, –0.0931, p=0.0405
NO_x [a, b]	S, –0.0679, p=0.134	S, –0.0531, p=0.255	S, +0.0232, p=0.532	S, +0.0111, p=0.807
PO_4 [b]	ND [c]	ND	–0.0680, p=0.000649	–0.0585, p=0.191
TP mass	S, –0.541, p=0.00000031		S, –0.366, p=0.000968	
TN mass	–0.544, p=0.0000021		S, –0.534, p=0.000928	
Chlorophyll-*a* mass	–0.345, p=0.00604		–0.306, p=0.00682	
TSS mass	–0.416, p=0.0000013		–0.338, p=0.0000645	

[a] Synthetic total fraction for 1983-2019; dissolved fraction for 2001-2019

[b] Non-detect methods

[c] ND – No data

[d] NH_4 not analyzed by standard methods for 2001-2019 because >15% below MDL

Table B- 7. Water quality trends for Lake Yale.

Kendall's Tau and significance levels. Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality at $p<0.05$

	1986 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	+0.336, p= 0.00000009	S, +0.134, p= 0.0875	+0.0115, p=0.891	–0.0493, p=0.486
TP [b, d]	+0.327, p=0.00000000	S, +0.135, p=0.00516		
TN	+0.430, p= 0.000321	S, +0.141, p= 0.103	–0.118, p= 0.403	–0.193, p=0.0989
Chlorophyll-*a*	+0.316, p= 0.00351	S, +0.0521, p= 0.525	–0.163, p= 0.186	–0.0230, p= 0.0319
TSS	+0.253, p= 0.000547	+0.0247, p= 0.710	–0.118, p=0.351	–0.205, p=0.0572
TSS [b, d]	+0.249, p=0.00000030	+0.0163, p=0.739		
Secchi	–0.278, p= 0.00548	S, –0.0233, p= 0.787	+0.105, p= 0.396	+0.173, p= 0.0745
NH_4 [a, b]	S, +0.0625, p=0.193	+0.0210, p=0.662	–0.0749, p=0.248	+0.0398, p=0.541
NO_x [a, b]	S, +0.0109, p=0.819	S, +0.0156, p=0.752	–0.0527, p=0.327	S, –0.0513, p=0.429
PO_4 [b]	ND [c]	ND	–0.0486, p=0.0314	–0.0488, p=0.444
TP mass	S, +0.200, p= 0.0136		–0.122, p=0.110	
TN mass	S, +0.299 p= 0.00273		–0.240, p=0.0484	
Chlorophyll-*a* mass	S, +0.293, p= 0.00360		–0.195, p=0.0971	
TSS mass	S, +0.0895, p= 0.305		–0.203, p=0.0605	

[a] Synthetic total fraction for 1986-2019; dissolved fraction for 2001-2019

[b] Non-detect methods

[c] ND – No data

[d] TP and TSS not analyzed by non-detect methods for 2001-2019 because <5% below MDL

Table B- 8. Water quality trends for Lake Weir.

Kendall's Tau and significance levels. Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality at $p<0.05$

	1984 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, +0.173, p=0.0133	S, +0.106, p=0.107	+0.206, p=0.00710	+0.0755, p=0.296
TP [b]	+0.126, p=0.00813	–0.0586, p=0.218	+0.188, p=0.00329	+0.0701, p=0.274
TN	+0.298, p=0.000148	+0.0646, p=0.364	+0.315, p=0.00765	+0.212, p=0.0459
Chlorophyll-*a*	+0.381, p=0.000524	S, +0.116, p=0.140	+0.259, p=0.199	+0.205, p=0.244
TSS	+0.164, p=0.00419	+0.139, p=0.0101	+0.275, p=0.000560	+0.229, p=0.00343
TSS [b]	+0.160, p=0.000585	+0.139, p=0.00302	+0.261, p=0.000036	+0.235, p=0.000221
Secchi	–0.437, p=0.00000000	–0.184, p=0.00753	–0.235, p=0.00454	–0.124, p=0.0914
NH_4 [a, b]	–0.0981, p=0.0312	–0.0601, p=0.198	S, –0.0286, p=0.694	–0.0313, p=0.675
NO_x [b]	ND [c]	ND	S, –0.0212, p=0.751	–0.0465, p=0.547
PO_4 [b]	ND	ND	S, –0.0373, p=0.451	–0.0221, p=0.738
TP mass	S, +0.0590, p=0.358		+0.125, p=0.0901	
TN mass	+0.0263, p=0.728		+0.152, p=0.172	
Chlorophyll-*a* mass	+0.246, p=0.0133		+0.155, p=0.464	
TSS mass	–0.0190, p=0.744		+0.185, p=0.0193	

[a] Synthetic total fraction for 1984-2019; dissolved fraction for 2001-2019

[b] Non-detect methods

[c] ND – No data

Table B- 9. Water quality trends for Lake Denham.

Kendall's Tau and significance levels. Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality at $p<0.05$

	1989 - 2019		2001 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, –0.288, p=0.00263	–0.222, p=0.000557	–0.293, p=0.000333	–0.243, p=0.00251
TN	–0.0882, p=0.348	–0.121, p=0.0863	–0.172, p=0.156	–0.0646, p=0.469
Chlorophyll-*a*	S, –0.092, p= 0.255	S, –0.0969, p= 0.214	–0.0808, p=0.353	–0.0544, p=0.513
TSS	–0.224, p=0.0466	–0.203, p=0.0457	–0.102, p=0.509	+0.0706, p=0.558
Secchi	+0.115, p= 0.208	+0.065, p= 0.461	+0.0536, p= 0.637	–0.0219, p= 0.840
NH_4 [a, b]	–0.00837, p=0.870	–0.0730, p=0.153	–0.162, p=0.0124	–0.0178, p=0.784
NO_x [a, b]	–0.0425, p=0.365	–0.0471, p=0.354	–0.0153, p=0.733	S, –0.0215, p=0.737
PO_4 [b]	–0.0140, p=0.653	–0.0240, p=0.634	–0.0390, p=0.103	–0.0204, p=0.746
TP mass	–0.222, p= 0.00485		–0.0720, p=0.492	
TN mass	–0.152, p=0.0487		–0.0198, p=0.839	
Chlorophyll-*a* mass	S, –0.0846, p= 0.302		+0.0235, p=0.784	
TSS mass	–0.246, p=0.0348		+0.0285, p=0.830	

[a] Dissolved fraction for both time periods.

[b] Non-detect methods

Table B- 10. Water quality trends for Lake Carlton.

Kendall's Tau and significance levels. Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality at $p<0.05$

	2002 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, –0.263, p= 0.0361	S, –0.290, p= 0.0251
TN	S, –0.280, p= 0.100	–0.390, p= 0.00202
Chlorophyll-*a*	S, –0.290, p= 0.0518	–0.245, p= 0.0239
TSS	–0.129, p= 0.367	–0.0043, p= 0.963
Secchi	+0.300, p= 0.0118	+0.174, p= 0.0604
NH_4	S, –0.234, p=0.0179	–0.0882, p=0.256
NH_4 [a]	S, –0.197, p=0.00430	S, –0.139, p=0.0433
NO_x [a]	S, +0.0163, p=0.718	S, +0.000224, p=1.0
PO_4 [a]	–0.0103, p=0.687	–0.0187, p=0.783
TP mass	S, –0.218, p=0.0333	
TN mass	S, –0.364, p=0.0156	
Chlorophyll-*a* mass	S, –0.296, p=0.0333	
TSS mass	–0.104, p=0.373	

[a] Non-detect methods

Table B- 11. Water quality trends for Trout Lake.

Kendall's Tau and significance levels. Significance levels are adjusted for serial correlation except for analyses using non-detect methods (see discussion in the Methods, Trend Analyses section).

(S) Significant seasonality at $p<0.05$

	2004 - 2019	
Parameter	**Unadjusted Trend**	**Water elevation adjusted trend**
TP	S, –0.122, p= 0.332	S, –0.122, p= 0.257
TN	–0.207, p= 0.182	–0.0592, p= 0.646
Chlorophyll-*a*	+0.0538, p= 0.662	+0.179, p= 0.0780
TSS [a]	–0.0165, p= 0.820	+0.172, p= 0.0164
Secchi	+0.216, p= 0.0442	+0.104, p= 0.286
NH_4	–0.236, p= 0.00866	–0.152, p= 0.0914
NH_4 [a]	–0.243, p= 0.000647	–0.152, p= 0.0330
NO_x [a]	–0.0710, p=0.277	–0.0970, p=0.175
PO_4 [a]	–0.104, p=0.147	–0.156, p=0.0310
TP mass	S, +0.0196, p= 0.887	
TN mass	–0.101, p= 0.455	
Chlorophyll-*a* mass	+0.104, p= 0.348	
TSS mass	+0.00774, p= 0.968	

[a] Non-detect methods

APPENDIX C. UPPER OCKLAWAHA BASIN LAKES WATER QUALITY AVERAGES AND TARGETS

Table C- 1. Average total phosphorus concentrations and TMDL targets.

TMDL Baseline Period is 1989–1994 for Lake Apopka, 1991–2000 for the other lakes.

Lake	TMDL Baseline Period (μg/L)	2016–2020 (μg/L)	TMDL Target (μg/L)
Apopka	211	74	55
Beauclair	207	48	32
Dora	95	36	31
Harris	39	27	26
Eustis	44	26	25
Griffin	102	34	32
Yale	23	31	20
Weir	14	20	10
Carlton	80	47	32
Trout	185	202	28
Denham	97	91	40

Table C- 2. Average corrected chlorophyll-*a* concentrations and TMDL expected values or targets.

TMDL Baseline Period is 1989–1994 for Lake Apopka, 1991–2000 for the other lakes.

TMDL targets for lakes Weir, Trout, and Denham; for other lakes expected concentrations at the TMDL target TP concentration.

Lake	TMDL Baseline Period (μg/L)	2016–2020 (μg/L)	TMDL Target (μg/L)
Apopka	95	40	ND[1]
Beauclair	148	35	30
Dora	135	32	29
Harris	59	18	22
Eustis	52	19	20
Griffin	147	30	30
Yale	16	16	14
Weir	8	15	6
Carlton	175	54	30
Trout	72	70	9.8
Denham	68	64	26.8

[1] ND – Not determined

Table C- 3. Average Secchi transparency and TMDL expected values.

TMDL Baseline Period is 1989–1994 for Lake Apopka, 1991–2000 for the other lakes.

Expected Secchi transparency at the TMDL target TP concentration.

Lake	TMDL Baseline Period (m)	2016–2020 (m)	TMDL Target (m)
Apopka	0.21	0.33	ND[1]
Beauclair	0.31	0.68	0.71
Dora	0.36	0.72	0.73
Harris	0.55	0.91	0.85
Eustis	0.54	0.82	0.87
Griffin	0.28	0.71	0.71
Yale	1.28	0.93	1.06
Weir	1.63	1.29	ND
Carlton	0.34	0.54	0.71
Trout	0.53	0.54	ND
Denham	0.40	0.49	ND

[1] ND – Not determined

Table C- 4. Average total nitrogen concentrations and TMDL targets.

TMDL Baseline Period is 1989–1994 for Lake Apopka, 1991–2000 for the other lakes.

Lake	TMDL Baseline Period (mg/L)	2016–2020 (mg/L)	TMDL Target (mg/L)
Apopka	5.29	2.94	ND[1]
Beauclair	4.34	1.76	ND
Dora	4.19	1.63	ND
Harris	2.17	1.23	ND
Eustis	3.02	1.39	ND
Griffin	4.40	1.63	ND
Yale	1.39	1.58	ND
Weir	0.86	0.95	0.68
Carlton	3.66	1.96	ND
Trout	1.68	1.72	0.78
Denham	2.68	2.07	1.10

[1] ND – Not determined

Table C- 5. Average total suspended solids concentrations.

TMDL Baseline Period is 1989–1994 for Lake Apopka, 1991–2000 for the other lakes.

TMDL targets or expected values not determined for TSS.

Lake	TMDL Baseline Period (mg/L)	2016–2020 (mg/L)
Apopka	81	51
Beauclair	39	13
Dora	31	11
Harris	18	8
Eustis	18	9
Griffin	32	10
Yale	9	8
Weir	6	7
Carlton	25	13
Trout	NA[2]	11
Denham	27	16

[2] NA – Not available

APPENDIX D. ANNUAL TP AND TN EXTERNAL LOADING TO THE UPPER OCKLAWAHA BASIN LAKES

Table D- 1. Annual external total phosphorus load estimates for Lake Apopka (kg/year).

Year	LANS	Atmospheric	Basin Runoff	Johns Lake	Septic[1]/ Point Sources	Apopka Spring/SeepIn	Total TP load TMDL=15,900
1989	113,320	9,772	1,571	0	689	1,030	126,381
1990	63,574	5,940	1,364	0	390	1,003	72,271
1991	43,990	5,959	1,698	0	448	1,001	53,095
1992	37,596	3,930	1,945	0	545	971	44,987
1993	17,262	2,881	1,660	0	755	1,000	23,558
1994	43,678	841	2,256	0	1,039	1,174	48,988
1995	29,139	1,204	1,402	0	1,490	1,071	34,306
1996	47,874	1,967	1,147	0	717	1,017	52,721
1997	37,688	3,177	1,482	0	588	1,044	43,980
1998	50,928	3,430	1,120	104	267	956	56,804
1999	19,765	3,760	1,298	0	239	890	25,952
2000	590	4,545	616	0	183	812	6,746
2001	571	2,647	964	0	227	646	5,056
2002	19,418	2,880	1,385	0	276	674	24,633
2003	35,541	3,406	1,227	51	558	817	41,600
2004	24,345	4,131	1,406	564	483	981	31,911
2005	23,524	3,731	1,355	491	423	1,127	30,650
2006	1,769	2,825	871	40	412	756	6,673
2007	420	5,351	1,043	0	370	704	7,887
2008	17,712	5,928	1,401	0	531	872	26,443
2009	7,189	5,835	1,484	547	484	902	16,441
2010	7,842	6,099	1,353	81	367	851	16,593
2011	945	4,313	1,810	0	238	727	8,033
2012	0	9,302	2,012	0	206	731	12,250
2013	0	3,909	2,141	0	762	664	7,476

Year	LANS	Atmospheric	Basin Runoff	Johns Lake	Septic[1]/ Point Sources	Apopka Spring/SeepIn	Total TP load TMDL=15,900
2014	353	4,458	2,550	29	824	727	8,941
2015	560	6,497	2,096	162	990	700	11,005
2016	2,258	5,134	2,389	109	884	821	11,595
2017	5,466	6,030	2,588	294	1,150	702	16,229
2018	3,942	4,026	2,861	979	1,137	1,087	14,031
2019	2,507	4,025	2,495	662	1,089	906	11,683

[1] NA – Septic tank load estimates began in 2013.

Table D- 2. Annual external total nitrogen load estimates for Lake Apopka (kg/year).

Year	LANS	Atmospheric	Basin Runoff	Johns Lake	Septic[1]/ Point Sources	Apopka Spring/SeepIn	Total TN load
2012	0	133,101	27,439	0	1,717	77,103	239,360
2013	0	82,446	28,789	0	11,102	77,503	199,839
2014	1,324	88,762	34,367	1,003	12,214	73,402	211,072
2015	10,327	85,187	27,979	5,770	13,373	73,037	215,674
2016	14,510	87,286	31,888	4,393	12,379	66,379	216,835
2017	78,358	90,926	34,758	7,623	13,722	65,419	290,805
2018	52,910	82,611	37,862	19,693	13,224	97,072	303,371
2019	56,386	74,724	33,341	15,900	11,588	86,416	278,355

[1] NA – Septic tank load estimates began in 2013.

Table D- 3. Annual external total phosphorus load estimates for Lake Beauclair (kg/year).

Year	Urban-residential runoff	Agriculture runoff	Natural area runoff	Atmospheric	Tributary discharge	Septic/ Point Sources	Total TP load TMDL= 3,200
1991	144	1,323	183	251	15,920	88	17,907
1992	120	940	152	159	6,658	88	8,118
1993	104	738	132	108	17,256	88	18,427
1994	150	1,247	189	67	32,128	88	33,869
1995	127	1,002	160	43	30,515	88	31,934
1996	121	832	153	160	29,156	88	30,509
1997	127	1,003	160	179	16,386	88	17,943
1998	113	863	144	133	39,954	88	41,295
1999	130	1,101	165	148	7,114	88	8,746
2000	74	395	94	163	2,422	88	3,236
2001	67	489	107	82	3,115	98	3,958
2002	107	785	163	98	2,746	98	3,996
2003	78	562	125	111	12,910	98	13,883
2004	97	748	148	133	18,426	98	19,650
2005	94	688	153	130	10,209	98	11,372
2006	63	227	99	92	1,045	98	1,624
2007	63	403	117	163	924	118	1,788
2008	129	574	183	173	1,693	118	2,870
2009	134	627	204	203	1,294	118	2,581
2010	71	275	126	206	1,517	118	2,314
2011	86	315	138	148	266	118	1,071
2012	68	261	121	293	253	118	1,114
2013	83	271	123	121	207	111	916
2014	110	422	152	154	79	111	1,027
2015	82	260	118	254	264	111	1,089
2016	106	339	137	188	349	111	1,230
2017	139	533	178	233	5,245	111	6,440
2018	108	395	158	165	6,389	111	7,328
2019	108	376	152	171	5,350	111	6,269

Table D- 4. Annual external total nitrogen load estimates for Lake Beauclair (kg/year).

Year	Urban-residential runoff	Agriculture runoff	Natural area runoff	Atmospheric	Tributary discharge	Septic/ Point Sources	Total TN load
1991	1,629	8,275	5,774	3,791	196,636	1,545	217,651
1992	1,359	6,019	4,817	2,787	86,201	1,545	102,728
1993	1,179	4,802	4,177	2,826	310,388	1,545	324,916
1994	1,691	8,540	5,991	2,289	714,441	1,545	734,497
1995	1,429	6,697	5,066	2,646	499,172	1,545	516,556
1996	1,368	5,401	4,848	1,811	383,346	1,545	398,319
1997	1,431	6,565	5,073	2,790	124,688	1,545	142,092
1998	1,280	7,208	4,537	2,256	658,853	1,545	675,680
1999	1,468	6,726	5,202	2,976	68,080	1,545	85,997
2000	838	2,365	2,970	3,025	68,818	1,545	79,560
2001	704	4,993	3,407	3,242	65,178	1,725	79,249
2002	1,156	7,588	5,149	3,477	58,035	1,725	77,130
2003	853	4,868	3,952	3,448	307,161	1,725	322,007
2004	1,092	32,139	4,664	3,727	446,816	1,725	490,162
2005	1,039	30,775	4,811	3,624	281,164	1,725	323,138
2006	697	9,425	3,120	2,709	42,164	1,725	59,840
2007	799	18,128	3,699	3,420	34,657	2,091	62,795
2008	1,729	8,700	5,786	3,534	57,423	2,091	79,263
2009	1,799	17,823	6,426	3,763	56,658	2,091	88,560
2010	900	9,796	4,003	3,106	77,198	2,091	97,094
2011	1,128	9,145	4,362	2,530	20,406	2,091	39,661
2012	868	8,680	3,820	4,092	20,757	2,091	40,309
2013	943	9,623	3,872	2,474	10,717	1,966	29,595
2014	1,273	14,178	4,775	2,995	4,557	1,966	29,744
2015	927	9,417	3,745	3,298	11,554	1,966	30,906
2016	1,241	10,083	4,329	3,050	14,325	1,966	34,993
2017	1,649	16,039	5,592	3,562	134,931	1,966	163,740
2018	1,242	13,858	4,971	3,339	227,947	1,966	253,323
2019	1,239	12,893	4,809	3,221	200,521	1,966	224,649

Table D- 5. Annual external total phosphorus load estimates for Lake Dora (kg/year).

Year	Urban-residential runoff	Agriculture runoff	Natural area runoff	Atmospheric	Tributary discharge	Septic/ Point Sources	Total TP load TMDL= 6,000
1991	1,075	15	172	1,013	10,936	187	13,398
1992	922	13	143	647	4,663	187	6,576
1993	812	11	124	442	10,725	187	12,302
1994	1,132	16	178	272	30,962	187	32,747
1995	981	14	151	173	17,402	187	18,907
1996	951	13	144	638	25,579	187	27,512
1997	982	14	151	736	9,920	188	11,990
1998	852	12	135	531	46,030	187	47,746
1999	930	14	155	607	4,700	187	6,592
2000	550	8	88	686	2,477	187	3,996
2001	621	8	100	355	1,694	212	2,989
2002	952	27	159	405	2,141	212	3,895
2003	695	11	129	438	10,510	212	11,994
2004	846	26	155	545	10,855	212	12,638
2005	933	23	165	529	8,810	212	10,672
2006	493	8	91	379	1,738	214	2,922
2007	653	20	123	711	945	203	2,655
2008	1,001	84	190	738	2,082	204	4,299
2009	1,069	89	209	845	1,872	202	4,287
2010	782	20	143	855	2,639	202	4,641
2011	791	47	154	625	533	203	2,352
2012	637	21	120	1,227	537	203	2,745
2013	752	28	131	520	207	300	1,938
2014	955	31	154	611	296	300	2,346
2015	748	17	121	889	522	299	2,596
2016	857	38	144	687	472	299	2,497
2017	1,047	61	175	833	3,996	299	6,410
2018	1,023	40	173	550	5,146	299	7,231
2019	949	28	153	565	4,398	299	6,392

Table D- 6. Annual external total nitrogen load estimates for Lake Dora (kg/year).

Year	Urban-residential runoff	Agriculture runoff	Natural area runoff	Atmospheric	Tributary discharge	Septic/ Point Sources	Total TN load
1991	7,313	181	5,067	15,335	184,901	4,625	217,422
1992	6,100	151	4,226	11,306	110,673	4,714	137,170
1993	5,289	131	3,665	11,574	221,632	4,678	246,969
1994	7,587	188	5,257	9,329	626,090	4,877	653,330
1995	6,415	159	4,445	10,544	375,774	4,924	402,261
1996	6,140	152	4,254	7,231	356,207	4,997	378,981
1997	6,424	159	4,451	11,437	205,364	4,928	232,763
1998	5,746	142	3,981	8,988	534,275	4,431	557,564
1999	6,588	163	4,565	12,220	99,056	4,030	126,623
2000	3,760	93	2,606	12,715	81,973	3,950	105,096
2001	4,666	85	2,998	13,889	73,534	3,782	98,953
2002	7,287	287	4,742	14,219	67,994	3,780	98,309
2003	5,245	124	3,811	13,556	319,676	3,778	346,190
2004	6,498	282	4,600	15,324	317,993	3,790	348,489
2005	7,146	257	4,907	14,875	302,341	3,761	333,287
2006	3,732	85	2,709	10,840	60,151	3,761	98,953
2007	5,306	222	3,660	14,936	40,927	3,577	68,628
2008	8,440	939	5,649	14,867	90,527	3,583	124,006
2009	9,007	995	6,227	15,553	87,520	3,572	122,875
2010	6,334	217	4,267	13,233	98,938	3,574	126,564
2011	6,549	521	4,569	10,765	26,835	3,577	52,816
2012	5,186	231	3,556	16,967	23,592	3,577	53,109
2013	5,860	296	3,903	10,726	10,233	5,282	36,300
2014	7,434	324	4,585	12,022	15,604	5,282	45,252
2015	5,790	176	3,600	11,534	19,711	5,277	46,088
2016	6,710	404	4,291	11,363	17,561	5,277	45,607
2017	8,263	639	5,204	12,655	110,405	5,281	142,446
2018	7,994	424	5,140	11,185	194,683	5,284	224,710
2019	7,381	296	4,573	10,582	142,229	5,278	170,338

Table D- 7. Annual external total phosphorus load estimates for Lake Harris (kg/year).

Year	Harris Bayou Restoration Area	Basin runoff	Spring discharges	Atmospheric	Tributary discharge	Septic/ Point Sources	Total TP load TMDL= 8,300
1991	2,160	5,009	994	4,345	114	620	13,241
1992	3,241	3,388	771	2,781	87	611	10,878
1993	3,241	2,875	826	1,889	79	606	9,516
1994	9,070	4,404	842	1,156	1,141	619	17,233
1995	1,517	3,483	981	737	659	616	7,993
1996	4,451	3,906	1,124	2,726	1,641	625	14,473
1997	3,330	3,643	842	3,158	1,285	612	12,870
1998	2,861	3,225	1,247	2,258	13,226	615	23,432
1999	418	3,809	832	2,607	215	610	8,491
2000	1,038	1,915	823	2,935	32	601	7,345
2001	0	2,641	915	1,669	118	699	6,042
2002	812	3,908	1,000	1,768	132	690	8,310
2003	2,308	2,837	1,074	1,964	4,865	765	13,812
2004	100	3,845	1,029	2,424	3,151	662	11,210
2005	1,751	3,653	1,126	2,277	2,092	662	11,562
2006	607	1,817	979	1,639	104	662	5,808
2007	0	2,560	811	3,154	33	994	7,552
2008	373	3,032	1,018	3,065	248	994	8,729
2009	0	3,790	889	3,509	149	994	9,332
2010	0	2,640	955	3,635	100	995	8,325
2011	0	2,926	737	2,647	57	994	7,361
2012	0	2,706	804	5,436	55	994	9,994
2013	0	2,430	797	2,311	32	1,082	6,652
2014	0	3,570	809	2,695	221	1,082	8,376
2015	0	2,451	865	3,928	1,786	1,082	10,112
2016	0	2,404	770	2,930	1,538	1,082	8,724
2017	0	4,048	1,115	3,591	1,200	1,084	11,038
2018	0	3,307	1,191	2,330	2,516	1,083	10,427
2019	0	2,808	1,220	2,320	2,210	1,083	9,641

Table D- 8. Annual external total nitrogen load estimates for Lake Harris (kg/year).

Year	Harris Bayou Restoration Area	Basin runoff	Spring discharges	Atmospheric	Tributary discharge	Septic/ Point Sources	Total TN load
1991	5,708	65,067	42,788	65,744	1,256	11,381	191,944
1992	8,562	48,346	34,549	48,585	2,174	11,109	153,325
1993	8,562	41,478	38,819	49,443	748	11,018	150,068
1994	19,690	60,893	41,419	39,707	49,363	11,305	222,377
1995	5,326	51,208	50,952	45,052	37,158	11,221	200,917
1996	10,780	49,433	56,697	30,886	50,532	11,326	209,655
1997	4,926	51,384	44,426	49,103	7,851	11,250	168,939
1998	7,725	45,630	56,615	38,255	147,457	11,183	306,864
1999	2,289	53,022	39,748	52,472	1,985	11,671	161,189
2000	9,196	28,386	28,159	54,422	1,046	11,243	132,452
2001	0	42,321	19,967	69,239	7,751	12,676	151,954
2002	3,097	61,271	23,793	62,839	3,828	12,466	167,293
2003	14,376	46,084	37,135	62,951	233,190	13,212	406,949
2004	922	60,888	34,512	70,563	139,271	11,768	317,925
2005	19,812	58,048	35,490	64,347	96,848	11,790	286,335
2006	6,753	31,216	30,409	47,876	2,561	11,795	130,608
2007	0	42,580	21,104	67,473	187	17,646	148,989
2008	6,418	50,355	21,999	59,996	8,205	17,639	164,613
2009	0	61,416	25,993	63,624	5,951	17,653	174,637
2010	0	44,812	29,079	56,302	700	17,659	148,552
2011	0	48,191	21,722	45,375	864	17,650	133,802
2012	0	45,910	19,115	77,917	2,440	17,650	163,033
2013	0	42,093	18,754	49,388	1,829	19,209	131,272
2014	0	59,332	21,511	54,670	3,738	19,209	158,460
2015	0	42,148	24,017	51,904	59,140	19,167	196,376
2016	0	41,869	22,298	48,144	72,667	19,162	204,140
2017	0	65,886	23,829	55,064	63,557	19,219	227,556
2018	0	55,945	28,847	47,109	80,116	19,200	231,216
2019	0	48,006	27,655	42,488	67,327	19,175	204,652

Table D- 9. Annual external total phosphorus load estimates for Lake Eustis (kg/year).

Year	Pine Meadows Restoration Area	Basin runoff	Atmospheric	Lake Dora discharge	Lake Harris discharge	Septic/ Point Sources	Total TP load TMDL= 9,200
1991	0	3,804	1,797	3,571	3,171	692	13,991
1992	174	2,271	1,150	2,034	918	692	7,922
1993	487	1,239	786	8,383	2,347	692	14,434
1994	699	1,758	481	17,154	2,543	692	24,344
1995	591	1,617	304	7,241	2,008	692	13,207
1996	428	1,372	1,126	15,238	6,644	692	26,191
1997	642	1,409	1,313	3,701	1,505	692	10,017
1998	438	1,374	933	26,285	7,660	692	37,985
1999	944	1,550	1,084	1,591	980	692	7,633
2000	383	1,002	1,233	1,393	731	692	5,589
2001	205	1,610	673	1,010	648	960	5,069
2002	178	1,897	721	868	542	960	5,762
2003	238	1,619	786	4,885	7,625	960	16,211
2004	314	1,926	994	6,125	7,306	960	18,148
2005	474	2,422	930	6,324	6,848	960	18,502
2006	241	1,502	678	1,407	1,903	960	6,328
2007	194	1,483	1,310	708	1,327	1,044	6,083
2008	297	2,211	1,349	1,354	908	1,049	8,062
2009	265	2,620	1,446	1,396	325	1,045	7,939
2010	192	2,225	1,514	2,825	2,123	1,055	10,176
2011	365	1,749	1,087	380	310	1,050	5,214
2012	381	1,496	2,236	366	253	1,050	5,744
2013	142	1,636	929	94	266	1,171	4,104
2014	285	2,086	1,174	120	297	1,171	5,659
2015	52	1,883	1,646	520	1,930	1,165	7,118
2016	111	1,673	1,203	515	1,916	1,165	6,485
2017	275	2,073	1,513	2,733	1,891	1,165	10,207
2018	185	1,927	1,011	4,474	2,877	1,172	11,995
2019	92	2,269	1,023	3,546	2,566	1,165	10,871

Table D- 10. Annual external total nitrogen load estimates for Lake Eustis (kg/year).

Year	Pine Meadows Restoration Area	Basin runoff	Atmospheric	Lake Dora discharge	Lake Harris discharge	Septic/ Point Sources	Total TN load
1991	0	44,871	27,199	210,431	86,753	12,222	382,364
1992	1,122	32,972	20,102	113,686	55,710	12,221	234,528
1993	3,134	24,912	20,582	278,690	111,935	12,220	448,737
1994	4,496	32,211	16,526	720,777	156,797	12,222	944,410
1995	3,801	29,273	18,569	388,092	212,376	12,221	663,614
1996	2,843	27,260	12,752	480,601	345,383	12,221	879,848
1997	3,273	27,747	20,411	191,937	76,825	12,221	331,710
1998	3,934	27,125	15,801	521,615	504,812	12,221	1,083,590
1999	4,727	28,741	21,813	91,498	46,771	12,221	205,362
2000	2,387	20,744	22,873	78,396	38,384	12,219	169,528
2001	2,974	32,841	26,907	59,139	28,077	16,961	155,896
2002	3,028	39,144	25,155	47,776	26,944	16,966	153,335
2003	2,536	33,430	24,656	324,741	328,659	16,971	721,471
2004	3,062	38,664	28,822	295,991	230,527	16,972	608,431
2005	2,729	40,339	26,151	390,767	254,844	16,972	726,232
2006	1,908	29,991	19,447	68,698	70,410	16,965	194,854
2007	2,919	30,558	27,428	41,570	58,720	18,457	171,645
2008	4,057	43,360	27,099	74,677	29,365	18,487	197,008
2009	3,547	43,327	25,898	68,831	13,881	18,473	173,322
2010	2,885	35,962	23,365	154,092	106,147	18,523	335,401
2011	2,857	35,138	18,365	22,535	14,630	18,494	106,960
2012	4,497	30,700	30,979	18,603	13,029	18,494	108,394
2013	3,699	32,947	18,847	5,327	14,518	20,626	86,178
2014	7,015	40,904	24,188	7,939	17,955	20,626	114,539
2015	2,576	34,758	21,213	32,573	121,768	20,591	224,176
2016	2,422	33,252	18,745	26,396	111,015	20,589	202,968
2017	3,812	40,943	22,632	158,734	86,889	20,599	329,836
2018	2,950	37,993	20,452	186,690	135,839	20,632	398,849
2019	2,676	38,709	18,907	141,153	132,975	20,594	348,154

Table D- 11. Annual external total phosphorus load estimates for Lake Griffin (kg/year).

Year	Emeralda Marsh Conservation Area	Harris Bayou Restoration Area	Basin runoff	Atmospheric	Tributary discharge	Septic/ Point Sources	Total TP load TMDL= 12,200
1984-1993	785	0	29,286	1,887	7,158	680	39,797
1994	14,679	0	2,542	849	10,823	858	29,749
1995	26,745	0	2,048	505	11,253	858	41,409
1996	22,387	0	2,037	1,965	21,474	858	48,721
1997	12,961	0	2,143	2,282	4,109	857	22,351
1998	8,144	0	1,794	1,555	29,807	859	42,159
1999	6,560	0	2,020	1,824	2,375	858	13,637
2000	9,080	0	1,184	2,116	1,580	858	14,818
2001	2,519	0	2,164	1,030	1,225	947	7,885
2002	4,200	0	2,480	1,226	1,215	944	10,065
2003	3,245	0	1,770	1,413	13,147	944	20,519
2004	904	0	2,692	1,693	15,030	944	21,263
2005	3,302	0	2,374	1,567	16,735	944	24,921
2006	1,035	0	1,383	1,192	4,068	944	8,622
2007	44	0	1,855	2,322	2,296	1,202	7,719
2008	256	485	2,821	2,419	2,066	1,202	9,250
2009	359	1,138	2,753	2,447	1,245	1,202	9,144
2010	141	1,600	1,718	2,468	6,274	1,202	13,403
2011	210	726	2,230	1,842	429	1,202	6,640
2012	93	588	1,936	4,504	433	1,202	8,757
2013	111	320	1,705	1,813	69	1,097	5,116
2014	145	655	2,483	2,360	219	1,097	6,959
2015	158	321	1,701	4,037	3,793	1,097	11,107
2016	80	210	1,333	2,696	2,536	1,097	7,952
2017	506	883	3,453	3,353	6,639	1,098	15,932
2018	534	1,949	2,500	2,398	10,727	1,099	19,208
2019	339	843	1,732	2,328	7,249	1,098	13,591

Table D- 12. Annual external total nitrogen load estimates for Lake Griffin (kg/year).

Year	Emeralda Marsh Conservation Area	Harris Bayou Restoration Area	Basin runoff	Atmospheric	Tributary discharge	Septic/ Point Sources	Total TN load
1984-1993	1,185	0	115,953	31,994	295,823	12,076	457,032
1994	25,950	0	35,502	29,864	618,798	15,280	725,393
1995	12,922	0	28,496	31,621	617,391	15,319	705,749
1996	-14,536	0	28,444	22,378	839,225	15,255	890,766
1997	28,030	0	29,910	36,451	159,748	15,264	269,403
1998	11,967	0	25,044	26,445	1,105,802	15,243	1,184,501
1999	-6,350	0	28,198	36,673	113,372	15,269	187,162
2000	1,955	0	16,459	39,487	68,909	15,253	142,063
2001	18,609	0	30,680	41,683	64,165	16,865	172,002
2002	43,721	0	33,345	42,670	51,993	16,886	188,615
2003	42,261	0	24,886	47,054	653,613	16,833	784,647
2004	13,226	0	38,768	49,811	545,139	16,874	663,818
2005	45,296	0	34,120	44,472	643,764	16,824	784,478
2006	11,439	0	20,041	36,151	158,937	16,816	243,385
2007	2,283	0	26,798	49,928	109,303	21,438	209,751
2008	6,466	7,301	41,001	49,801	72,555	21,594	198,720
2009	8,807	17,327	39,409	43,663	52,710	21,444	183,360
2010	5,998	13,256	25,510	37,375	325,913	21,384	429,437
2011	6,917	14,884	32,619	31,277	20,263	21,474	127,433
2012	4,952	16,704	28,504	66,114	15,858	21,474	153,607
2013	4,349	7,320	25,202	37,777	2,502	19,627	96,777
2014	5,481	13,832	35,393	48,683	10,281	19,627	133,296
2015	4,913	3,913	24,866	53,769	154,203	19,534	261,197
2016	4,800	4,413	19,902	42,299	122,893	19,683	213,990
2017	15,470	21,812	48,222	51,844	364,132	19,601	521,081
2018	14,984	7,780	35,440	49,962	478,973	19,538	606,677
2019	11,177	5,386	25,392	42,956	329,190	19,498	433,598

Table D- 13. Annual external total phosphorus load estimates for Lake Yale (kg/year).

Year	Urban-residential runoff	Agriculture runoff	Natural area runoff	Atmospheric	Tributary discharge	Septic/ Point Sources	Total TP load TMDL= 1,290
1991	325	113	277	1,303	0	397	2,414
1992	329	115	281	768	0	346	1,839
1993	270	94	230	477	0	263	1,335
1994	402	140	343	256	0	182	1,323
1995	323	113	275	218	0	168	1,096
1996	322	112	275	997	0	154	1,860
1997	339	118	289	708	0	151	1,605
1998	283	99	242	666	0	147	1,437
1999	319	112	272	539	0	148	1,390
2000	186	65	159	612	0	143	1,166
2001	241	78	223	359	0	228	1,129
2002	313	126	281	364	0	228	1,311
2003	279	50	220	438	0	228	1,216
2004	375	153	306	651	0	228	1,714
2005	373	98	295	805	0	228	1,800
2006	216	39	173	429	0	228	1,084
2007	257	90	207	655	0	268	1,477
2008	487	567	434	733	2	269	2,491
2009	446	482	394	695	0	269	2,284
2010	328	112	257	775	6	268	1,746
2011	312	252	264	523	0	269	1,620
2012	277	98	226	1,130	0	269	2,000
2013	285	88	195	450	0	256	1,275
2014	473	267	331	563	0	256	1,891
2015	334	152	240	793	0	257	1,777
2016	269	103	184	547	0	257	1,360
2017	473	460	354	715	1	258	2,259
2018	404	203	295	481	0	258	1,643
2019	367	122	251	482	0	258	1,480

Table D- 14. Annual external total nitrogen load estimates for Lake Yale (kg/year).

Year	Urban-residential runoff	Agriculture runoff	Natural area runoff	Atmospheric	Tributary discharge	Septic/ Point Sources	Total TN load
1991	2,337	1,054	8,340	18,950	0	3,816	34,497
1992	2,370	1,069	8,459	14,001	0	3,546	29,445
1993	1,944	877	6,938	12,758	0	3,118	25,635
1994	2,894	1,305	10,328	8,997	0	2,707	26,231
1995	2,323	1,047	8,290	13,660	0	2,663	27,983
1996	2,319	1,046	8,275	11,350	0	2,557	25,545
1997	2,438	1,099	8,701	11,314	0	2,581	26,133
1998	2,041	921	7,286	11,321	0	2,541	24,110
1999	2,299	1,037	8,203	10,842	0	2,553	24,933
2000	1,342	605	4,788	11,427	0	2,485	20,647
2001	1,874	1,368	6,673	14,561	0	4,013	28,489
2002	2,447	2,202	8,381	12,474	0	4,017	29,521
2003	2,247	839	6,581	13,981	0	4,010	27,659
2004	3,094	2,459	9,131	19,264	0	4,016	37,964
2005	3,031	1,599	8,814	23,092	0	4,015	40,550
2006	1,744	654	5,150	12,997	0	4,005	24,550
2007	2,079	1,396	6,188	13,395	0	4,720	27,777
2008	4,115	8,582	12,850	15,023	44	4,727	45,341
2009	3,762	7,336	11,668	12,352	0	4,726	39,844
2010	2,634	1,721	7,684	11,903	564	4,723	29,230
2011	2,606	3,907	7,856	8,617	0	4,725	27,712
2012	2,253	1,524	6,747	16,178	0	4,725	31,426
2013	2,281	857	5,814	9,140	0	4,511	22,604
2014	3,843	2,579	9,854	11,854	0	4,511	32,641
2015	2,703	1,474	7,167	10,497	0	4,514	26,356
2016	2,163	996	5,480	8,247	2	4,511	21,399
2017	3,942	4,433	10,476	10,827	30	4,517	34,225
2018	3,280	1,965	8,782	9,823	14	4,522	28,386
2019	2,939	1,178	7,485	8,876	6	4,518	25,002

Table D- 15. Annual external total phosphorus load estimates for Lake Weir (kg/year).

Year	Urban-residential runoff	Agriculture runoff	Natural area runoff	Atmospheric	Septic	Total TP load PLRG= 1,230 TMDL= 1,667
1991	123	39	91	1,096	234	1,583
1992	170	54	127	831	234	1,415
1993	159	50	118	578	234	1,140
1994	222	70	165	368	234	1,059
1995	183	58	136	214	234	825
1996	136	43	101	728	234	1,242
1997	161	51	120	923	234	1,489
1998	141	45	105	653	234	1,179
1999	132	42	98	699	234	1,205
2000	98	31	73	875	234	1,311
2001	122	19	92	481	310	1,023
2002	164	27	123	501	310	1,124
2003	119	14	116	571	310	1,129
2004	141	23	131	713	310	1,318
2005	183	22	184	686	310	1,384
2006	77	8	75	484	310	953
2007	140	35	126	1,018	397	1,716
2008	161	55	135	949	397	1,696
2009	175	55	149	1,037	397	1,812
2010	112	13	109	1,025	397	1,656
2011	127	27	118	791	397	1,459
2012	131	23	123	1,707	397	2,381
2013	120	14	120	694	330	1,279
2014	158	24	154	807	330	1,473
2015	120	15	119	1,147	330	1,731
2016	105	14	105	831	330	1,385
2017	168	37	156	1,048	330	1,740
2018	203	67	173	725	330	1,498
2019	142	21	139	726	330	1,359

Table D- 16. Annual external total nitrogen load estimates for Lake Weir (kg/year).

Year	Urban-residential runoff	Agriculture runoff	Natural area runoff	Atmospheric	Septic	Total TN load TMDL= 27,432
1991	1,414	372	2,738	13,934	4,135	22,593
1992	1,956	515	3,787	15,163	4,135	25,556
1993	1,829	481	3,541	16,497	4,135	26,483
1994	2,554	672	4,945	13,203	4,135	25,509
1995	2,108	555	4,081	14,493	4,135	25,370
1996	1,567	412	3,034	8,171	4,135	17,319
1997	1,853	488	3,588	14,356	4,135	24,419
1998	1,630	429	3,155	11,007	4,135	20,356
1999	1,518	400	2,940	13,164	4,135	22,156
2000	1,129	297	2,186	16,460	4,135	24,206
2001	1,399	195	2,860	19,444	5,023	28,922
2002	1,885	278	3,831	17,449	5,023	28,467
2003	1,383	151	3,572	18,591	5,023	28,720
2004	1,665	252	4,038	21,057	5,023	32,035
2005	1,361	165	3,280	19,908	5,023	29,737
2006	890	91	2,326	13,861	5,023	22,191
2007	1,662	375	3,890	23,089	7,005	36,021
2008	1,940	585	4,133	18,983	7,005	32,647
2009	2,083	589	4,565	18,971	7,005	33,214
2010	1,287	143	3,380	15,689	7,005	27,504
2011	1,491	285	3,627	13,959	7,005	26,366
2012	1,520	250	3,795	25,996	7,005	38,566
2013	1,383	153	3,644	15,124	5,832	26,136
2014	1,828	254	4,685	16,947	5,832	29,546
2015	1,377	157	3,612	15,415	5,832	26,393
2016	1,211	146	3,171	13,651	5,832	24,011
2017	1,973	396	4,705	16,543	5,832	29,449
2018	2,438	705	5,216	15,408	5,832	29,599
2019	1,642	226	4,203	13,744	5,832	25,647

APPENDIX E. COMPARISON OF SJRWMD RUNOFF MODELING FOR THE UPPER OCKLAWAHA BASIN LAKES WITH BASIN RUNOFF STUDIES

SUMMARY

St. Johns River Water Management District (SJRWMD) modeled estimates of stormwater runoff used in PLRG/TMDL development and nutrient budgets for the lakes in the Upper Ocklawaha River basin were compared with runoff measurements or estimates from three recent runoff studies conducted in the basin. For two Lake Apopka tributaries, SJRWMD modeled TP runoff tended to be lower than measured in storm event monitoring by ECT (2018), primarily due to very high TP concentrations measured in the storm event monitoring. For a third site there was a closer correspondence between modeled and measured TP discharges. For all three sites, ECT measured discharges tended to be higher than SJRWMD modeled for the largest storm events, while modeled discharges tended to be higher for smaller storm events. I recommend that additional stormwater phosphorus concentration measurements should be taken for the two sites with high measured TP concentrations to determine if those measurements were anomalous. If consistently high concentrations continue, it may be desirable to adjust the runoff calculations for those sub-basins to use higher TP concentrations.

For a largely agricultural sub-basin in the Lake Beauclair watershed, Amec (2017) estimates of TP discharges were higher than modeled by SJRWMD, because of high discharges in baseflows that are not accounted for in the SJRWMD modeling. For TN, the SJRWMD modeled discharges exceeded those measured by Amec due to the higher TN concentrations used in the modeling for farm discharges than measured by Amec. SJRWMD has more recently developed HSPF/Basins runoff models for much of the District. If a HSPF/Basins model is available for this sub-basin it could provide a comparison estimate of runoff and/or baseline discharge volumes.

For the Lake Yale watershed, ERD (2017) estimated substantially lower annual runoff volume and TN runoff than modeled by SJRWMD. However, the ERD and SJRWMD estimates of TP runoff were more similar. Here again, a HSPF/Basins model may provide another estimate of runoff volume from the Lake Yale watershed.

There are not consistent differences between SJRWMD modeled stormwater runoff and those estimated or modeled in these three studies. So, these comparison studies don't provide evidence for a systematic problem with the SJRWMD runoff estimates. It could be problematic to make significant changes in methodology from those used during periods for

which TMDLs were developed for the basin lakes, because the resulting nutrient load estimates may not be comparable with those for the TMDL development period, so we would not be able to assess progress in reducing the nutrient loads.

LAKE APOPKA RUNOFF STUDY

ECT (2018) measured stormwater runoff for several tributaries along the south shore of Lake Apopka. For three tributaries, event mean pollutant concentrations and loading were measured between May and August 2018. I compared those data with SJRWMD modeling of stormwater runoff for the same periods:

1. Fullers Cross – compared with modeled runoff for the Crown Point Slough sub-basin, which drains to Lake Apopka through this stream
2. Lulu Creek – compared with modeled runoff for the Lulu Creek sub-basin, which drains to Lake Apopka through this stream. However, the ECT Lulu Creek station was located upstream of a heavily developed area, so this station did not measure the total runoff from this sub-basin.
3. Johns Lake Outlet – compared with modeled runoff for the Johns Lake Outlet sub-basin, which drains to Lake Apopka through this stream. However, the SJRWMD Johns Lake Outlet sub-basin does not include the entire drainage basin. Upstream is Johns Lake, which also discharges through the stream to Lake Apopka, so I also incorporated our estimates of Johns Lake discharges in the comparison with the ECT measurements. In addition, there is a portion of the drainage basin between Johns Lake and the delineated Johns Lake Outlet sub-basin that is not included in the SJRWMD runoff modeling. The part of the drainage basin that is not included in the runoff modeling is mostly undeveloped, but crosses under two major highways, so the modeling may underestimate total runoff from this sub-basin.

ECT (2018) measured discharges at Fullers Cross for 6 storm events (Table 1). Note that the storm event monitoring included only portions of days, but the SJRWMD modeling is of runoff for full days, so I matched up the modeled day(s) that best corresponded with the measurement periods. For 4 of the storm events, the SJRWMD modeled discharges exceeded that measured by ECT, but in the 2 largest storm events the measured discharges were larger than modeled. Averaged across all 6 storm events, the modeled discharges were about 76% of measured.

Table E- 1. Fullers Cross measured and modeled storm discharge volumes.

Storm	Start Date	End Date	ECT measured discharge (cu. ft.)	SJRWMD modeled discharge (cu. ft.)	Difference from ECT measured discharge (cu. ft.)	Percent difference
FC-S1	5/14/2018 8:00	5/15/2018 2:45	1,235,797	1,395,556	159,759	112.9%
FC-S2	5/29/2018 16:15	5/30/2018 6:00	1,639,416	2,333,339	693,923	142.3%
FC-S3	6/8/2018 14:45	6/9/2018 3:00	2,943,161	3,395,296	452,135	115.4%
FC-S4	6/30/2018 12:15	7/2/2018 5:00	7,296,046	2,987,564	-4,308,482	40.9%
FC-S5	7/31/2018 14:45	8/1/2018 13:45	1,716,844	3,613,989	1,897,145	210.5%
FC-S6	8/19/2018 15:15	8/20/2018 9:00	10,238,232	5,228,085	-5,010,147	51.1%
Mean			4,178,249	3,158,971	-1,019,278	75.6%

ECT (2018) measured discharges at Lulu Creek for 6 storm events (Table 2). In 5 of the 6 storm events, the SJRWMD modeled discharges exceeded the ECT measurements. ECT measured discharges slightly exceeded the modeled discharge for the largest storm event. Averaged across all 6 storm events, the modeled discharges were 80% larger than measured. At least part of the reason the modeled discharges exceeded the measurements was probably because the Lulu Creek monitoring station was upstream of a heavily developed area, so did not capture all the discharges from the sub-basin.

ECT (2018) measured discharges at Johns Lake Outlet for 6 storm events (Table 3). For all 6 events, SJRWMD's estimate of discharges from Johns lake were substantially larger than those modeled for the downstream sub-basin. The sum of the SJRWMD estimated Johns lake discharges and the modeled sub-basin discharges exceeded the ECT measured discharges for 5 of the 6 storm events, but the ECT measurements were greater for the largest storm event. Averaged across all 6 storm events, the sum of the SJRWMD estimated Johns lake discharges and the modeled sub-basin discharges were about 80% of ECT measured discharges.

For the three tributary sites, ECT (2018) calculated Event Mean Concentrations (EMCs) for the storm events and average Baseflow concentrations. For comparison with those measurements, I calculated expected nutrient concentrations from SJRWMD runoff modeling for those sub-basins. These were calculated from land-use specific nutrient concentrations used in the modeling, weighted by modeled land-use specific runoff volumes.

Table E- 2. Lulu Creek measured and modeled storm discharge volumes.

Storm	Start Date	End Date	ECT measured discharge (cu. ft.)	SJRWMD modeled discharge (cu. ft.)	Difference from ECT measured discharge (cu. ft.)	Percent difference
LU-S1	5/14/2018 8:00	5/15/2018 12:26	187,926	453,812	265,886	241.5%
LU-S2	5/30/2018 11:30	5/30/2018 17:45	337,252	606,102	268,850	179.7%
LU-S3	6/8/2018 15:15	6/8/2018 20:00	354,809	979,794	624,985	276.1%
LU-S4	6/30/2018 12:30	6/30/2018 18:00	814,209	741,879	-72,330	91.1%
LU-S5	8/1/2018 13:45	8/2/2018 1:00	236,447	394,486	158,039	166.8%
LU-S6	8/16/2018 21:00	8/17/2018 7:30	126,559	524,286	397,727	414.3%
Mean			342,867	616,727	273,860	179.9%

The measured EMCs for TP at Fullers Cross and Lulu Creek were much higher than those used in the SJRWMD modeling (Table 4). The EMC TP concentrations for Fullers Cross exceeded the concentrations used in the modeling and in a more recent Florida Runoff EMC Database (ERD 2017) for all individual land uses occurring in that sub-basin, except for Cropland (0.666 mg/L in the SJRWMD modeling), and Pasture (0.617 mg/L in the Florida Runoff EMC Database), which combined cover only about 9% of the watershed area. The EMC TP concentrations for Lulu Creek exceeded the concentrations used in the modeling and in the Florida Runoff EMC Database (ERD 2017) for all individual land uses occurring in that sub-basin.

The Fullers Cross and Lulu Creek Baseflow TP concentrations were more similar to those used in the SJRWMD modeling. For TN at Fullers Cross and Lulu Creek, both the measured EMC and Baseflow concentrations were similar to those used in the SJRWMD modeling (Table 4).

For Johns Lake Outlet, the ECT measured TP and TN concentrations were lower than SJRWMD modeled values for that sub-basin, and closer to the average measured values for Johns lake discharges (Table 4). This is probably because most of the flow through that sub-basin came from discharges from Johns Lake (Table 3).

Table E- 3. Johns Lake Outlet measured and modeled storm discharge volumes.

Storm	Start Date	End Date	ECT measured discharge (cu. ft.)	SJRWMD modeled Johns Lake Outlet discharge (cu. ft.)	SJRWMD estimated Johns Lake discharge (cu. ft.)	Sum SJRWMD modeled Johns Lake Outlet and Johns Lake discharge (cu. ft.)	Difference from ECT measured discharge (cu. ft.)	Percent difference
JL-S1	5/14/2018 7:45	5/15/2018 12:30	167,832	91,852	621,261	713,113	545,281	424.9%
JL-S2	5/30/2018 11:00	5/30/2018 21:00	583,191	82,071	825,208	907,279	324,088	155.6%
JL-S3	6/8/2018 15:30	6/8/2018 21:30	228,043	129,852	1,009,538	1,139,390	911,347	499.6%
JL-S4	6/29/2018 12:45	6/29/2018 16:30	262,209	64,195	1,203,587	1,267,782	1,005,573	483.5%
JL-S5	8/16/2018 18:00	8/17/2018 1:40	1,862,267	77,911	5,874,790	5,952,702	4,090,435	319.6%
JL-S6	8/19/2018 17:50	8/24/2018 22:10	34,962,207	882,036	19,664,550	20,546,586	-14,415,621	58.8%
Mean			6,344,292	221,320	4,866,489	5,087,809	-1,256,483	80.2%

Table E- 4. ECT and SJRWMD nutrient concentration estimates.

Source	Average TP (mg/L)	Average TN (mg/L)
Fullers Cross – ECT EMC	0.59	2.06
Fullers Cross – ECT Baseflow	0.19	1.08
Fullers Cross – SJRWMD modeling	0.180	1.71
Lulu Creek – ECT EMC	1.43	2.16
Lulu Creek – ECT Baseflow	0.36	0.75
Lulu Creek – SJRWMD modeling	0.285	1.97
Johns Lake outlet – ECT EMC	0.07	0.82
Johns Lake outlet – ECT Baseflow	0.05	0.91
Johns Lake outlet – SJRWMD modeling	0.161	1.65
Johns Lake discharges – mean 2018 SJRWMD	0.052	1.10

Combining the SJRWMD modeled discharge volumes with the SJRWMD nutrient concentration estimates, I estimated TP and TN discharges for the storm events for comparison with the values calculated by ECT. For Johns Lake Outlet, I used the sum of the modeled Johns Lake Outlet and Johns Lake discharge volumes (Table 3), and nutrient concentrations were calculated as flow-weighted averages of the modeled Johns Lake outlet and the average 2018 Johns Lake discharges (Table 4).

For Fullers Cross, the SJRWMD modeled TN discharges were larger than those measured by ECT for 4 of the 6 storm events, but in the 2 largest storm events the measured discharges were larger than modeled (Table 5). Averaged across all 6 storm events, the modeled TN discharges were about 75% of measured. This is similar to the differences in discharge volumes (Table 1) and reflects the similarity of ECT EMC and SJRWMD modeled TN concentrations (Table 4). For TP, the SJRWMD modeled discharges were smaller than those measured by ECT for 5 of the 6 storm events and averaged about 28% of the measured discharges (Table 5). This is due to the measured TP EMCs being substantially higher than the modeled value (Table 4).

Table E- 5. Fullers Cross measured and modeled storm TP and TN discharges.

Storm	ECT measured TP discharge (lb)	SJRWMD modeled TP discharge (lb)	Percent difference	ECT measured TN discharge (lb)	SJRWMD modeled TN discharge (lb)	Percent difference
FC-S1	25	16	62.7%	112	149	132.7%
FC-S2	42	26	62.4%	151	248	164.5%
FC-S3	118	38	32.3%	361	361	100.1%
FC-S4	186	34	18.0%	710	318	44.8%
FC-S5	27	41	150.3%	144	385	267.2%
FC-S6	357	59	16.4%	1,213	557	45.9%
Mean	126	35	28.2%	449	336	75.0%

As with Fullers Cross, for Lulu Creek the TN discharges mirrored the discharge volumes. In 5 of the 6 storm events, the SJRWMD modeled TN discharges exceeded the ECT measurements (Table 6). ECT measured discharges slightly exceeded the modeled discharge for the largest storm event. Averaged across all 6 storm events, the modeled TN discharges were about twice the measured discharges. Again, the similarity with the differences in discharge volumes reflects the similarity of ECT EMC and SJRWMD modeled TN concentrations for Lulu Creek (Table 4). For TP, the SJRWMD modeled discharges were higher in only 3 of the 6 storm events and averaged about 44% of the measured discharges (Table 6). This is primarily due to the measured TP EMCs being substantially higher than the modeled value (Table 4).

Table E- 6. Lulu Creek measured and modeled storm TP and TN discharges.

Storm	ECT measured TP discharge (lb)	SJRWMD modeled TP discharge (lb)	Percent difference	ECT measured TN discharge (lb)	SJRWMD modeled TN discharge (lb)	Percent difference
LU-S1	7	8	115.2%	34	56	164.5%
LU-S2	29	11	37.1%	35	75	213.4%
LU-S3	27	17	64.5%	36	121	335.5%
LU-S4	75	13	17.6%	99	91	92.4%
LU-S5	6	7	116.8%	18	49	270.1%
LU-S6	5	9	186.3%	9	65	718.0%
Mean	25	11	44.1%	39	76	197.4%

For Johns Lake Outlet, the trends in volume, TP, and TN discharges were broadly similar. For all three parameters, the SJRWMD modeled discharges exceeded the ECT measured discharges for 5 of the 6 storm events, but the ECT measurements were greater for the largest storm event (Tables 3, 7). Averaged across all 6 storm events the SJRWMD modeled TP discharges were 67% of measured and the TN discharges were 109% of measured. SJRWMD modeled TN discharges generally exceeded measured discharges by a larger percentage than did modeled TP discharges because of higher SJRWMD-estimated flow-weighted concentrations. The average flow-weighted TN concentration was 1.14 mg /L, exceeding the measured Johns Lake Outlet EMC (Table 4). The average flow-weighted TP concentration was 0.059 mg /L, slightly lower than the measured Johns Lake Outlet EMC (Table 4).

Table E- 7. Johns Lake Outlet measured and modeled storm TP and TN discharges.

Storm	ECT measured TP discharge (lb)	SJRWMD modeled TP discharge (lb)	Percent difference	ECT measured TN discharge (lb)	SJRWMD modeled TN discharge (lb)	Percent difference
JL-S1	1	3	294.1%	10	52	521.3%
JL-S2	2	4	175.2%	24	65	271.4%
JL-S3	1	5	458.3%	12	83	689.3%
JL-S4	2	5	227.6%	16	89	557.9%
JL-S5	6	20	330.9%	109	411	377.5%
JL-S6	149	73	48.8%	1,785	1,441	80.7%
Mean	27	18	67.2%	326	357	109.5%

LAKE BEAUCLAIR RUNOFF STUDY

Amec (2017) conducted a monitoring/modeling study of nutrient discharges from a sub-basin in the Lake Beauclair watershed, along the Apopka-Beauclair (A-B) Canal. The sub-basin they studied is similar to one of the sub-basins used in District modeling of stormwater runoff to Lake Beauclair. The basin studied by Amec had a drainage area of 1,631 acres, smaller than the District sub-basin of 2,179 acres, primarily because the District sub-basin included land on the west side of the A-B Canal, but the Amec study area included only land on the east side of the canal. Most of the land on the west side of the canal is pasture or forested, which contributes relatively little phosphorus runoff (in District modeling pasture and forest lands in the sub-basin, some of which are on the east side of the canal, contribute less than 4% of the TP runoff from that sub-basin). This sub-basin includes the Hurley (or Lake Jem) Farm which pumps discharges into a canal surrounding the farm. The Amec study monitored flows and nutrients in two canals draining other agricultural lands to the east of Hurley Farm, and at two locations where the combined flows from the Hurley Farm and other lands discharge into the A-B Canal.

Amec (2017) measured flows and water quality, separated into baseflows measured during dry periods and storm event samples, during the period July 2016 to January 2017. Annual baseflows were extrapolated from the 6-month monitoring period. Annual stormwater flows were estimated using a continuous simulation ICPR® model to quantify stormwater runoff volume for a 52-inch rainfall year. Annual baseflow volume was estimated as 3,395 acre-feet. From their Figure 3-2, roughly 1/3 of this flow came from the canals draining the eastern agricultural lands, and the remainder was measured at the discharges into the A-B Canal. Some of the baseflow discharging into the A-B Canal could have been pump discharges from Hurley Farm. The modeled stormwater flow from the sub-basin was 931 acre-feet per year. Total discharges, including both baseflows and stormwater were estimated as 4,326 acre-feet.

In SJRWMD modeling for 2019, a year in which rainfall was close to 52 inches, the predicted stormwater runoff volume from the sub-basin was 1,930 acre-feet. A separate estimate of discharges from Hurley Farm, based on a regression equation relating reported discharges from basin muck farms to acreage in production (Fulton 1995) was 1,753 acre-feet for 2019. The SJRWMD estimate of stormwater runoff volume from the sub-basin is roughly twice that of Amec, but when Amec's baseflow runoff estimate is included their estimate of total runoff volume is roughly twice that of the SJRWMD.

Table 8 summarizes average water quality in stormwater and baseflow samples measured by Amec, and water quality concentrations used in the SJRWMD nutrient load estimates. The two CR448 sites are in the ditches draining the agricultural lands on the east side of the sub-basin. The SJRWMD estimates for cropland runoff (based on literature studies, Fulton et al. 2004) are higher for both TP and TN than measured by Amec. However, the cropland runoff estimate is for concentrations at the point of discharge, while the Amec measurements are further downstream, likely incorporating some attenuation in transport from the runoff source. Reported TP concentrations in Hurley Farm discharges are in the range measured by Amec, but Hurley discharges had higher TN concentrations (again the Hurley data are for concentrations at the discharge from the farm). The most recent measurements of Hurley discharge concentrations were lower than previous measurements, perhaps reflecting changes in operations.

Table E- 8. Amec and SJRWMD nutrient concentration estimates.

Source	Average TP (mg/L)	Average TN (mg/L)
Amec – CR448 at Duda stormwater	0.288	1.32
Amec – CR448 at Duda baseflow	0.519	1.39
Amec – CR448A at Long & Scott stormwater	0.311	1.31
Amec – CR448A at Long & Scott baseflow	0.265	1.48
Amec – ABC north ditch baseflow	0.097	1.38
Amec – ABC south ditch baseflow	0.328	1.38
SJRWMD modeling – cropland	0.666	4.56
Hurley Farm permit monitoring 2002-2009	0.304	6.69
Hurley Farm permit monitoring 2008-2009	0.170	5.02

Amec and SJRWMD estimates of nutrient loading from the sub-basin are summarized in Table 9. Amec's estimate of TP load is substantially greater, largely due to the baseflow loads. SJRWMD's estimates also include an attenuation factor based on transport distance from the sub-basin to Lake Beauclair, which reduced the TP load by roughly 30% and the

TN load by about 10%. However, SJRWMD's estimate of TN load is higher, likely due to the higher TN concentrations used for farm discharges.

Table E- 9. Amec and SJRWMD nutrient loading estimates.

Source	TP load (lb/yr)	TN load (lb/yr)
Amec – stormwater	705	4,133
Amec – baseflow	2,004	12,753
Amec – total	2,709	16,886
SJRWMD modeling – 2019	862	29,946

LAKE YALE RUNOFF STUDY

ERD (2017) conducted a study that included runoff estimates for the Lake Yale and Trout Lake watersheds. Their study did not include measurements of stormwater runoff volumes. They estimated generated stormwater runoff volumes using a modeling approach based on SCS curve number methods that appears generally similar to the approach used in the SJRWMD runoff modeling. The ERD estimates of runoff for the Lake Yale watershed used a much more detailed drainage basin network than used in the SJRWMD modeling, but the total watershed area used by ERD is very similar to that used in the SJRWMD modeling. ERD subtracted from the generated runoff volumes estimates of reductions due to stormwater management systems, depressional areas, and wetlands to produce an estimated runoff reaching the lake. The SJRWMD modeling did not include these estimates of volume reductions. ERD estimated runoff volumes for an average annual rainfall of 49.67 inches. Their estimates were a generated runoff volume of 2,253 acre-feet and runoff volume reaching Lake Yale of 1,455 acre-feet. In the SJRWMD modeling, 2019 had watershed rainfall close to the average used by ERD (50.12 inches). However, the runoff volume estimated in that year for the Lake Yale watershed by the SJRWMD modeling was substantially higher than the ERD estimate of generated volume, 8,450 acre-feet. Even dry years had substantially greater runoff volumes in the SJRWMD modeling than the ERD estimate (e.g. 2006, rainfall 33.92 inches, modeled runoff 5,540 acre-feet). I don't know the reasons for this difference.

ERD (2017) manually collected water quality samples on a biweekly basis or following significant storm events which produced measurable runoff from the watershed at 3 sites in the Lake Yale watershed and 4 sites in the Trout Lake watershed. A series of flow-weighted samples were collected during events where measurable discharge was observed. The flow-weighted samples were combined to form a composite sample for each site during each monitored event. The measured nutrient concentrations at these sites (Table 10) tended to be low compared to the land-use specific concentrations used in the SJRWMD modeling and in the Florida Runoff EMC Database. A notable exception was the very high TP concentration measured at site LYD 14-01, but that was only a single sample. The manually collected samples may not have captured complete storm events, particularly may have missed the first flush which usually has the highest pollutant concentrations, so the storm EMCs could be underestimated. Also, the samples collected at biweekly intervals likely reflected baseflows, not stormwater runoff. ERD did note high variability in nutrient concentrations among samples. For example, for site LYD 13-01, Appendix H gives a range in TP concentrations from 0.126 to 0.753 mg/L.

Table E- 10. ERD mean nutrient concentration measurements at Lake Yale and Trout Lake watershed monitoring sites.

Site	Watershed land use	Number of samples	TP (mg/L)	TN (mg/L)
LYD 12-01	Residential, Pasture, Citrus	8	0.166	2.225
LYD 13-01	Agriculture	17	0.291	0.683
LYD 14-01	Industrial	1	8.305	1.342
TL Hicks Ditch	Mixed	16	0.101	1.388
TLW Nature Center	Wetland	3	0.052	0.635
TL 01-02 Inflow 3	Open land, Industrial	1	0.032	0.591
TL 04-05 Inflow 1	Mobile home – high density	4	0.137	1.132

For estimating stormwater nutrient loads to Lake Yale, ERD used a combination of the measured values from the monitoring sites, and concentrations from the Florida Runoff EMC Database for other watershed areas. For the average rainfall year, they estimated a stormwater TP load of 509 kg and a TN load of 3,012 kg. For 2019, a year with similar rainfall, the SJRWMD modeling estimates were 740 kg TP and 11,602 kg TN. The higher TN loading estimate in the SJRWMD modeling is consistent with the difference in estimated stormwater runoff volumes. However, the SJRWMD modeled TP loading is not substantially higher than the ERD estimate, despite the large difference in estimated runoff volumes. Potential reasons for this include:

- Although the SJRWMD modeling does not reduce flow volume reaching the lake from that generated in the watershed, nutrient loads are reduced to incorporate stormwater treatment and attenuation based on the travel distance from the runoff source sub-basin.
- The ERD runoff estimate incorporated the very high TP concentration measurement from site LYD 14-01. Of their total estimated watershed TP runoff, 148.8 kg (29%) came from that sub-basin, which was only 0.6% of the total watershed area.

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