Turning up the Heat: Long-term water quality responses to wildfires and climate change in a hypereutrophic lake

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<u>Open research Statement.</u>

Data sets utilized for this research were retrieved from the following databases at the following links, and query details for retrieving the relevant data are included as follows:

1) Archived Github repository: https://doi.org/10.5281/zenodo.6815236

2) California Department of Forestry and Fire Protection (CAL FIRE). 2021. Fire and Resource Assessment Program (FRAP). frap.fire.ca.gov/mapping/gis-data/
Query Details: Data accessed through the Fire perimeters through 2021 tab, which provides GIS layer files.

3) California Department of Water Resources. 2021. Data Water Library. wdl.water.ca.gov/Map.aspx
Query Details: Select "Water Quality" and select the three Clear Lake stations, Cl-01, Cl-03, Cl-04 from the map points and download relevant data for all records, at all depths for total phosphorus, dissolved oxygen, SRP (Orthophosphate), and temperature from the retrieved field data.

4) PRISM Climate Group 2004, February 4. Oregon State University. prism.oregonstate.edu.
Query details: We used Lake County, CA km PRISM climate data downloaded from the "Explorer" tab and confirmed by the map grid and data selection tool prior to mass download.

Abstract

Clear Lake (Lake County, CA, USA) is hypereutrophic and used for drinking water, tribal use, and supports a significant fishing economy. The Mendocino Complex (2018), one of the largest wildfires in California's post-settlement history, burned 40% of the Clear Lake watershed, providing a timely opportunity to study the impacts of historical and current wildfires on this valuable aquatic resource. Using long-term monthly monitoring data from 1968-2019, paired with historical watershed fire data, we found that for the three largest fire years in the watershed's history, three-year post-fire median July-October epilimnetic total phosphorus (TP) concentrations were below or equal to three years pre-fire TP concentrations. However, both median TP epilimnetic concentrations and deep water temperature across the lake have increased since the late 1960s. Long-term TP increases were more strongly correlated with monthly maximum air temperatures than precipitation, suggesting a potential role of warming-induced water column stratification, dissolved oxygen (DO) depletion, and high potential for internal phosphorus loading. Hypoxic occurrences were correlated with higher hypolimnetic soluble reactive phosphorus (SRP) and TP concentrations, but additional high-frequency monitoring of DO will help determine the duration of anoxia and its contribution to internal phosphorus loading. These long-term data suggest that for this large, hypereutrophic lake, wildfires did not significantly alter in-lake TP concentrations based on long-term, monthly monitoring and that other internal or external sources of TP may mask any wildfire effects. Nonetheless, our study underscores the value of synthesizing decades of water quality, watershed wildfire, and climate data to build a more comprehensive, nuanced picture of multiple long-term threats to aquatic ecosystems under global change. Moreover, monitoring and studying fire effects across a wide range of lake types beyond this study will help promote more effective lake management during changing climates and increasingly frequent large wildfires.

Keywords: anoxia; climate change; hypereutrophic; monitoring; phosphorus; shallow lake; wildfires

Introduction

Recent increases in the size and frequency of large wildfires have been well documented in the western US (Dennison et al. 2014, Holden et al. 2018). Whereas many studies have focused on the effects of wildfires on terrestrial ecosystems, human structures, and air quality, wildfire effects on water quality have been relatively overlooked until recently (Bixby et al. 2015, Williams et al. 2022). Particularly as climate change and growing human populations place further strain on water resources, it is increasingly important to examine how increasing wildfire activity may affect water quality in the future.

Past research on wildfire effects on water quality suffers from several limitations, including a lack of historical context and biases toward certain types of ecosystems and geographic settings. Due to the unpredictability of wildfires, few studies lack extensive pre-fire data or consider only a single fire event, making it difficult to contextualize post-fire observations (Roces-Diaz et al. 2022). Although previous research has shown substantial effects of wildfires on water quality in rivers and streams (Dahm et al. 2015, Rust et al. 2018), there are far fewer studies on lakes and reservoirs (hereafter, lakes) (Bixby et al. 2015). Moreover, the limited lake research has been predominantly conducted in remote, boreal regions (McCullough et al. 2019) and the focus on low-nutrient, subalpine lakes (Allen et al. 2003, Williamson et al. 2016, Scordo et al. 2021) makes it difficult to predict how lakes in different ecological contexts might respond to wildfires.

Nonetheless, some general patterns of wildfire effects on water quality have emerged that can guide future lake-fire research and provide testable hypotheses across a more ecologically diverse set of lakes, landscapes, and fire regimes. Previous studies have generally found increased concentrations of nutrients, ions, organic material, and chlorophyll-*a*, in addition to decreases in water clarity, following post-fire increases in erosion and runoff; however, effects often vary according to time since fire, lake trophic status, and landscape characteristics (McCullough et al. 2019). For example, short-term decreases in light availability can reduce phytoplankton growth within the first season following fire regardless of greater nutrient inputs (Allen et al. 2003, Ranalli 2004). Pre-fire nutrient concentrations matter as well: the eutrophic Lac Francis, Quebec demonstrated smaller increases in diatom-phosphorus following more recent fire events (< 200 years ago) compared to older fire events (> 1000 years ago) based on 2000 years of sediment records (Enache and Prairie 2000). Lac Francis is more eutrophic now than a millennium ago, suggesting that fire-derived nutrient enrichment might not be as detectable in eutrophic lakes as in oligotrophic or mesotrophic lakes. Additionally, McColl and Grigal (1977) found that post-fire nutrient uptake by vegetation and cation absorption by acidic soils negated any potential increases in lake nutrient concentrations in Minnesota lakes. These heterogeneous, nuanced responses of lakes to fires underscore the urgent need for more lake-fire research across ecologically diverse settings, particularly within fire-prone landscapes where humans depend on lakes for drinking water, irrigation, recreation, cultural use, and other ecosystem services (Smith et al. 2011, Hohner et al. 2019, Rhoades et al. 2019).

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Clear Lake, California: a convenient case study

Beginning on July 27, 2018, the record-setting Mendocino Complex Fire burned a combined 1858 km^2 across Mendocino, Lake, Colusa, and Glenn counties in northern California. Consisting of the adjacent Ranch and River fires, the Mendocino Complex was the largest wildfire complex in post-settlement California history prior to the megafires of 2020 (California Department of Forestry and Fire Protection 2019, 2021). Ultimately extinguished on November

7, the fires collectively burned for 72 days, destroying 281 structures and accumulating over USD 267 million in insured property damages and fire suppression costs (NIFC 2018).

Perhaps overlooked among media headlines, however, was the fact that the fires burned within approximately a kilometer of the Clear Lake shoreline in addition to 40% of its watershed (Figure 1a). Originally referred to as (Ha)Ka-ba-tin, or Lupiyoma, meaning “Large or Big Water” by the local, indigenous Pomo (Mauldin 1960, Clark and Williams 1954), Clear Lake is the largest naturally formed lake located completely in California. Clear Lake is used for drinking water, irrigation, and recreation, as well as tribal cultural use and activities, but it is also a hypereutrophic lake with a history of high total phosphorus (TP) concentrations and cyanobacterial blooms (Sharp et al. 2021, Richerson et al. 1994, Suchanek et al. 2003; Richerson et al. 2008). The large extent and proximity of the fires to Clear Lake raised concerns among residents and water managers about the short- and long-term effects of the Mendocino Complex on lake water quality and associated ecosystem services.

A notable aspect of Clear Lake’s monitoring program is the availability of long-term water quality data since the late 1960s. Additionally, statewide fire history records date back to the 1920s. As large wildfires increase in frequency and size across the western US (Dennison et al. 2014), combining long-term water quality and fire history data will help contextualize recent wildfires and provide a long-term perspective on water quality and associated watershed fire history for Clear Lake. In this study, a novel synthesis of long-term water quality and fire history data could help place both the Mendocino Complex fire itself and post-Mendocino Complex lake water quality into an appropriate local historical context of a history of smaller wildfires and declining water quality. Such context is particularly useful for interpreting short-term water quality responses to future fires of varying characteristics in addition to predicting potential long-term effects on water quality, including expected timelines for ecosystem recovery.

Although increasing fire activity has great potential for negative effects on lake water quality and other important ecosystem services, it is also important to consider that other aspects of global change may interact with, exacerbate, or mask fire effects on lakes. In the specific case of Clear Lake, warming air temperatures, combined with the lake's hypereutrophic status, have the potential to increase risks of algal and cyanobacterial blooms, resulting in drinking water advisories and fish kills. Changing climatic conditions, such as rising water temperatures can contribute to anoxia and hypolimnetic oxygen depletion, driving internal phosphorus loading and altering stratification dynamics (Woolway et al. 2021). Furthermore, longer periods of warm droughts interspersed with extreme precipitation events may rapidly increase nutrient and sediment influxes (Woolway et al. 2020; Mishra et al. 2021), particularly following watershed fires that destabilize soils and increase erosion (Lundquist and Smythe 2010; Ranalli 2004; Hohner et al. 2019). Therefore, there is a need to investigate the impacts of both wildfires and climate variability on nutrient concentrations in Clear Lake, given that both have the potential to worsen already hypereutrophic conditions and further increase risks of algal blooms. Such efforts are essential for more informed short- and long-term lake management responses under ongoing global change.

In this study, we combined publicly available long-term water quality, wildfire, and climate data to answer the following questions: 1) What is the historical extent and frequency of fire activity in the Clear Lake watershed? 2) For the three largest wildfire years in recent history, how do post-fire TP concentrations compare to pre-fire concentrations in Clear Lake? 3) How are long-term TP concentrations in Clear Lake affected by variability in climate and dissolved oxygen concentrations? Collectively, these questions will allow us to contextualize the Mendocino Complex with respect to watershed fire history, interpret post-fire Clear Lake water quality in response to the Mendocino Complex and other watershed fires in the historical record, and examine other potential long-term drivers of hypereutrophic conditions in Clear Lake that may be exacerbated by ongoing climate change. Moreover, we help address limitations of previous research on wildfire effects on freshwater ecosystems by conducting long-term research in an understudied region and type of ecosystem (i.e., Mediterranean climate, hypereutrophic lake). Through this investigation, we aim to provide guidance to lake managers and researchers when considering the potential impacts of wildfires, climate change, or both on lake water quality and other lake ecosystem services.

Methods

Study Site: Clear Lake, California

Clear Lake is located (39.07°N, 122.83°W) in the Central Mountain Range in Lake County, California at an approximate elevation of 400 m at the lake surface. Clear Lake is the largest naturally formed, freshwater lake located entirely within California. The lake is estimated to be the oldest North American lake at 2.5 million years old, with subsequent geologic and volcanic activity contributing to the lake's current three-basin configuration (Upper Arm, Lower Arm, and Oaks Arm; Figure 1a) (Horne 1975, Thompson et al. 2013). Clear Lake serves as the primary drinking water source for a population of about 40,000, the majority of which are economically disadvantaged and tribal communities (CDWR 2021). At full capacity, Clear Lake has a surface area of 177 km^2, a shoreline length of 160 km, and a total volume of 1.4 x 10^9 m^3 (Feyrer et al. 2020). The Upper Arm comprises about 70% of the lake's surface area and receives approximately 90% of the contributing runoff from the watershed (Rueda et al. 2005). The lake is fairly shallow with a mean depth of 8 m, but the deepest point is located at the Oaks Arm sample station at 17 m. The other two sampling stations are located at the deepest points in the Upper Arm (7 m) and Lower Arm (12 m).

The Clear Lake watershed covers 968.5 km^2 and reaches a maximum elevation of 1466 m in the northeastern portion (Gesch et al. 2018; Appendix S1: Figure S1). Watershed land cover is predominantly a mixture of forest (28.2%), shrubbery (43.9%), and herbaceous vegetation (13.8%) with some agriculture at lower elevations north and south of the Upper Arm (2.6%) (Yang et al. 2018; Appendix S1: Figure S2). The southern portion of Mendocino National Forest extends into northern areas of the watershed. Developed land (9.8%) is primarily located near the lake shoreline. Wetlands are limited to 1.4% of the watershed, mostly north of the Upper Arm (Yang et al. 2018; Appendix S1: Figure S2).

The watershed has lost over 85% of its natural wetlands, which has contributed to hypereutrophication, particularly during the summer and fall months, and to the high frequency and duration of toxic cyanobacterial blooms (Richerson et al. 2008). Clear Lake is a polymictic lake, with the potential for high rates of internal nutrient loading in the warm summer and fall months. Based on sediment cores, the lake is historically eutrophic due to volcanic, P-rich parent material (Richerson et al. 2008). External nutrient loading has occurred mostly due to anthropogenic activities, starting with grazing and logging, and including a long history of mining (Richerson et al 2008, Suchanek et al. 2008). Clear Lake is fed by several intermittent streams, which are typically dry in the summer and fall months and inundated from seasonal rains during winter and spring, as typical in a temperate, Mediterranean climate. Based on the 1981-2010 normals, average water-year (Oct-Sep) precipitation is 835.8 mm (PRISM Climate

Group 2004). The average maximum and minimum air temperatures are 22.5°C and 5.2°C, respectively, based on the same period. Summers (Jun-Aug) are typically hot (average maximum temperature = 32.4°C), whereas winters (Dec-Feb) are considerably cooler (average minimum temperature = 0.6°C). Snowmelt contribution is negligible as average snow per year within the basin is <1" (LCWPD 2006). The vast majority of annual precipitation falls between November and March (84%) (PRISM Climate Group 2004).

Fire data and analysis

We quantified Clear Lake watershed fire history using fire perimeter polygons from CAL FIRE from 1923 to 2020 (California Department of Forestry and Fire Protection 2021). This dataset contains forest fires ≥ 10 acres (4 ha), brush fires ≥ 30 acres (12 ha), and grass fires ≥ 300 acres (120 ha). We obtained a watershed polygon from the LAGOS-US LOCUS v 1.0 database (Cheruvelil et al. 2021; Smith et al. 2021). We quantified the watershed proportion burned for both wildfires and prescribed burns using GIS tools in the R raster (Hijmans 2019) and rgeos packages (Bivand and Rundel 2019); however, we ultimately focused on wildfires alone because they accounted for > 99% of the watershed area burned over time. From the long-term wildfire dataset, we identified 1981, 1996, and 2018 (Figure 1a, b) as the years with the greatest watershed proportion burned for our analysis of post-fire TP.

Whereas CAL FIRE polygons represent total area burned, another important consideration is burn severity. Fire effects on stream water quality have been shown to increase with burn severity (Rhoades et al. 2011; Rust et al. 2018); however, few studies have specifically examined burn severity impacts on lakes (McCullough et al. 2019). We obtained burn severity data from 1984-2018 for the Clear Lake watershed based on satellite-derived Monitoring Trends in Burn Severity data (Eidenshink et al. 2007; Appendix 1: Figure S3). Traditionally, burn severity is binned into three categories of low-, moderate-, and high-severity based on vegetation mortality (low: < 20%, moderate: 20-70%, high: > 70%). Watershed proportions burned at low-, moderate-, and high-severity were all strongly correlated with total watershed proportions burned from 1984-2018 (Spearman's rho = 0.98-99, $p < 0.05$). Therefore, we opted to focus on total watershed proportion burned owing to these strong correlations, and the fact that burn severity data were unavailable prior to 1984.

Water quality monitoring data and analysis

Regular water quality monitoring has been conducted on Clear Lake as part of the California Department of Water Resources (CDWR) State Water Project (http://wdl.water.ca.gov/) in collaboration with the Lake County Watershed Protection District. Monthly water quality monitoring has been conducted at three sites (Upper Arm, Lower Arm, and Oaks Arm), except rarely in November and February for over 50 years (1968-2019; Figure 1a). Our four primary limnological variables of interest over this period were TP, water temperature, soluble reactive phosphorus (SRP), and dissolved oxygen (DO). TP and SRP water column samples were collected using a Van Dorn style 2.2 L sampler at the surface (0.5 m and 3 m) and 1 meter above the maximum depth of each site, which varied depending on season and lake level. Samples were stored on ice and transported or shipped to either CDWR's Bryte Laboratory in Sacramento, CA, or an outside state-approved laboratory, filtered for dissolved species, (SRP, <0.45 micron) and analyzed following methods described by CDWR (2002). Profile data for DO and water temperature were also collected at 1-meter intervals using a handheld meter and sonde. Sampling equipment changed over time, but the most recent profile instrument used was a YSI Professional Plus handheld meter and EXO2 multiparameter sonde (Yellow Springs Instruments, Yellow Springs, OH). Additional information on the methods and sampling equipment used can be found in the California DWR Water Data Library (https://wdl.water.ca.gov/Map.aspx).

We used median July-October TP data to capture the late summer and fall period that contains long-term annual peak in TP and coincides with periods where external inputs were minimal as described in other studies (Nürnberg et al. 2013). Due to the zero-inflated nature of historical wildfire data (i.e., many years with little or no fire), we analyzed TP three years before and after the three years with the greatest proportion watershed burned (1981, 1996, and 2018), consistent with past studies (Ranalli 2004). Although post-fire TP data following the Mendocino Complex were missing for 2020 owing to COVID-19 state sampling/boating restrictions, we managed to incorporate two observations from each site during both July and August 2021.

We focused on the > 50 year TP and water temperature record for our long-term analyses of Clear Lake water quality trends. To account for limitations of ordinary least squares regression for time series data (i.e., distributional assumptions, outlier sensitivity), we used non-parametric Thiel-Sen slopes (represents median of all pairwise differences between annual timesteps) to detect the estimated annual rate of change in July-October median TP, median surface water temperature, and median deep water temperature and tested for significance with Kendall's p-value using the zyp.trend.vector function in the zyp R package (Bronaugh and Werner 2019). We did not investigate or include nitrogen (N) in this study due to data gaps and previous research has demonstrated that Clear Lake is primarily a P-limited system with low N:P water column ratios (Horne 1975, Richerson et al. 1994, Richerson et al. 2008). Additionally, excess phosphorus has led to the majority of cyanobacterial blooms in Clear Lake and is the focus of the current sediment and nutrient total maximum daily load (TMDL) plan (Suchanek et al. 2003, Tetra Tech 2004). Current research is ongoing in Clear Lake to collect additional nitrogen data to test for N limitation.

For the long-term DO time series analysis, we calculated annual July-October median DO concentrations at the lake surface and bottom from water column profiles from 1968-2019 to coincide with TP and SRP data. The sampling frequency varied over the study period ranging from as frequent as weekly profiles at the beginning of the time series to monthly or bi-monthly profiles in the latter part of the study period. We also analyzed long-term TP and SRP data in relation to DO to determine if anoxic events were correlated with higher concentrations of TP and SRP in the hypolimnion of each site. Spearman's rho correlation coefficients were used to compare the relationship between the minimum DO and TP or SRP concentrations at the maximum depth of each site to identify if internal loading could be a potential contributor to long-term TP trends. Additionally, the number of times when the DO concentration from each profile was hypoxic (DO <2 mg/L) or anoxic (DO <1 mg/L) (Nürnberg 1995, Orihel et al. 2017) was counted and summarized as a percent of the total number of samples per decade to account for inconsistencies in sampling effort among years and decades. The percent of the total number of DO readings that was hypoxic or anoxic was also calculated for each site across the full study period (1968-2019). A threshold of 2 mg/L was chosen as an intermediate estimate of hypoxic conditions and corresponds with recent work that found SRP fluxes from lake sediments rapidly increased when DO concentrations were <1.5 mg/L (Osaka et al. 2022).

Climate data and analysis

We used 4 km PRISM climate data (PRISM Climate Group 2004) to provide a continuous, consistent climate data source rather than piecing together data from various weather stations and locations available for different periods of our water quality data record. We found, however, that PRISM precipitation data were highly correlated with precipitation measured at the nearby Kelseyville (available 1998 to present; 39.00°N, 122.51°W) and Lakeport (available through 2001; 39.03°N, 122.92°W) weather stations (Pearson's $r \geq 0.90$). Another Lakeport station (Lakeport 2.5; 39.07°N, 122.92°W) has precipitation data starting in 2009, but no air temperature data. Although air temperature data were unavailable at the Kelseyville station, Pearson correlations between PRISM and Lakeport observations were ≥ 0.95 for both minimum and maximum temperatures. These strong correlations provided sufficient confidence for using PRISM data to examine climate-TP relationships. We downloaded monthly total precipitation and mean minimum and mean maximum air temperature grids from the lake center (39.07°N, 122.83°W) for 1965 through 2019 and used these to calculate seasonal climate variables. Seasonal time periods were chosen to reflect the locally wet months (November-March), spring (Apr-Jun), and the warm and dry season from mid- to late-summer that is particularly prone to wildfires (July-October). We used Spearman's correlation coefficients to compare these seasonal climate variables with the median July-October TP data to examine climatic predictors of annual mid- to late-summer TP peaks. Similar to the fire analysis described above, we used Spearman's correlations to account for non-normality in the TP data (climate data were approximately normally distributed). All analyses were conducted in R (version 4.0.3). Data and R scripts are available on Zenodo from DePalma-Dow et al. (2022).

Results

Watershed fire history

From 1923-2020, there were relatively frequent fire events in the watershed, including 112 wildfires. Wildfires were most frequent from approximately 1945-1967, with only 7 years experiencing no wildfires. From 1923-1967, there were 23 years without wildfire in the watershed, whereas there were 29 such years from 1968-2020. Despite the high frequency of fires, even in years with multiple fire events, the proportion of the watershed that burned was relatively low in most years (Figure 2). The three years with the greatest proportion burned were 1981 (0.12; 114.76 km^2), 1996 (0.07; 66.91 km^2), and 2018 (0.40; 384.72 km^2). The 1981 fire year consisted of four wildfires (Cow Mountain, Hunter, Sulphur Bank, and School Teacher Hill fires) that ignited in August and September, and the 1996 fire year consisted of a single wildfire (Fork fire) that ignited in August (Figure 1b). The Mendocino Complex in 2018 led to the highest proportion of the watershed burned with high-severity fire (0.08) since 1984 (Appendix S1: Figure S3), whereas all other years since 1984 were below 0.02. Therefore, the 2018 Mendocino Complex Fire appears to be locally anomalous in the Clear Lake watershed in terms of watershed proportion burned in the last approximately 100 years and in terms of burn severity since the mid-1980s.

Fire and lake water quality

Post-fire surface TP concentrations were generally consistent with pre-fire data in Clear Lake with no spikes in TP after the 1981 and 1996 fire events (Table 1, Figure 3). Specifically, median July-October TP concentration in the three years following the 1981 and 1996 fires was below or equal to median July-October TP in the three preceding years across all sites. Maximum July-October TP (0.42 mg/L) at the Upper Arm exceeded all other observations at respective sites over the three years preceding the fire in September 1981, likely while the wildfires were still active (no containment nor extinguishing dates were available). There was a similar peak in July 1996 at the Lower Arm (0.97 mg/L), but this was before that year's fire began.

Moreover, we did not detect substantial increases in surface TP following the 2018 Mendocino Complex compared to the three previous years. Due to state and local COVID-19 boating restrictions during 2020, the only July-October post-fire TP data available were collected in August 2019 and July and August 2021. At both the Upper and Lower Arms, median pre-fire TP ($n = 11$) was greater than median post-fire TP ($n = 3$): 0.64 vs. 0.34 mg/L and 0.38 vs. 0.20 mg/L, respectively (Figure 3). Whereas no post-fire TP concentrations exceeded the pre-fire median at the Upper Arm, one post-fire TP observation at the Lower Arm in August 2021 (0.44 mg/L) exceeded the pre-fire median, but this was still within the pre-fire interquartile range. In contrast, at the Oaks Arm, median post-fire TP ($n = 3$) exceeded pre-fire TP ($n = 11$) (0.55 vs. 0. 46 mg/L), although this was still within the pre-fire interquartile range (Figure 3). Therefore, although post-fire TP data were relatively limited, post-fire concentrations were consistently within the range of pre-fire concentrations up to three years post-fire.

Long-term total phosphorus and water temperature trends

From 1968-2021, July-October TP increased significantly ($p \leq 0.05$) at all sites, with the majority of annual increases occurring between the late-1980s - 2016 (Figure 4). Annual rates of increase (Thiel-Sen slope estimates) were 5.0, 3.0, and 4.0 ug/L at the Upper, Lower, and Oaks Arms, respectively (Kendall's p-values = 0.03, 0.05, and 0.02, respectively; Figure 4). From 1968-2019, surface water temperature increased marginally in the Upper Arm ($p = 0.07$) but did not increase significantly in the Oaks and Lower Arms ($p > 0.05$; Figure 5). The annual rate of increase in surface water temperature (Thiel-Sen slope estimates) in the Upper Arm was 0.03°C (Figure 5). Deep water temperatures increased significantly in all arms over the 1968-2019 time period with Thiel-Sen slope estimate values of 0.06, 0.05, and 0.05°C at the Upper, Lower, and Oaks Arms, respectively (All Kendall's p-values < 0.001; Appendix S1: Figure S8).

Long-term dissolved oxygen and relationships with TP

Over the 50-yr study period, long-term DO concentrations at the surface and bottom of Clear Lake remained fairly constant (Figure 6). Median surface July-October DO concentrations across the three sites ranged from 5.2-15.2 mg/L with a median of 9.2 mg/L, while bottom-water concentrations ranged from 0-7.4 mg/L with a median of 3.6 mg/L. Surface DO concentrations trended down in the Lower and Oaks Arms at the beginning of the study period, but then leveled off. There were periods of hypoxia and anoxia in the bottom-water samples at all three sites throughout the study, primarily in the summer and fall months (July-October). However, there was no noticeable trend in median DO concentrations at the lake bottom across the study period. Uneven sampling across months and years precluded robust analysis of any long-term trends. The percent of the DO samples that were hypoxic or anoxic was fairly consistent across all three sites, ranging from 15-48% for anoxia and 0-18% for hypoxia, but the Lower and Oaks Arms generally had higher percent anoxia (48% and 30%, respectively) than the Upper Arm (21%), particularly in the first three years of the study period (Table 2). The prevalence of anoxia fluctuated across the decades but generally declined over time in the Lower and Oaks Arms while the Upper Arm remained constant and had the highest percent of anoxic samples for the full time period (28%). At all sites, the percent hypoxic was consistently less than the percent anoxic and stayed fairly constant over time in the Upper Arm, but decreased in the Oaks and Lower Arms.

The comparison of bottom-water hypoxia (DO concentrations < 2 mg/L) and July-October median SRP concentrations revealed a negative correlation between DO concentration and SRP concentration at all three sites (Figure 7). The negative relationship was strongest in the Upper Arm (Spearman's rho = -0.44, $p < 0.05$), followed by the Oaks Arm (Spearman's rho = -0.40, $p < 0.05$), and not significant in the Lower Arm (Spearman's rho = -0.04, $p = 0.77$). Across all three sites, the strength of the SRP and DO correlations followed the same order as the long-term TP trends with the Upper Arm showing the greatest increase in TP and strongest negative correlation. However, there was also significant variability in the relationship between SRP and DO because high SRP concentrations were observed across a range of minimum DO concentrations, particularly in the Lower Arm (Figure 7). Correlations between TP and DO were similar to SRP, but overall trends were weaker except in the Upper Arm (Spearman's rho = -0.48, $p < 0.05$; Appendix S1: Figure S9).

Long-term relationships between TP and climate variability

There were stronger negative relationships between precipitation and July-October TP, while maximum and minimum air temperature were generally weaker and not significantly related to TP. Precipitation across all seasons and sites was negatively correlated with median July-October TP, but correlations were the most negative and significant during November-March (Spearman's rho = -0.30 to -0.34; $p < 0.05$) across all sites (Table 3). Precipitation during July-October was also significantly negatively correlated with median July-October TP (Spearman's rho = 0.28, $p < 0.05$) at the Oaks Arm. Overall, precipitation during the previous fall and winter (November-March) was negatively correlated with July-October TP and explained more variability in TP than the maximum or minimum air temperature variables. In general, average maximum air temperature was not correlated with median July-October TP (Spearman's rho = 0.14 – 0.26, $p > 0.05$) with the exception of July-October maximum temperature correlated with TP in the Oaks Arm (Spearman's rho = 0.28, $p < 0.05$). Correlations between median July-October TP and average maximum air temperatures from November-March were positive, but overall weaker ($p < 0.10$), suggesting possible minor lagged effects. Average minimum air temperatures across other seasons were not significantly correlated with median July-October TP. Climate-TP relationships across all seasons in all three sites can be found in the supplementary material (Appendix S1: Figures S5, S6, S7)

Discussion

Interpreting short- and long-term effects of wildfires on Clear Lake water quality: the importance of historical and ecological context

Despite the size and proximity of the very large Mendocino Complex to Clear Lake (40% watershed burned 1 km from shoreline), we did not find considerable changes in TP when comparing concentrations up to three years pre- and post-fire. Similarly, we observed no changes in TP in response to smaller fire events in 1981 and 1996 (12 and 7% watershed burned, respectively) when comparing TP concentrations three years pre- and post-fire (Figure 3). Therefore, our results suggest that relatively small future watershed fires may not lead to major changes in Clear Lake TP concentrations. We could not have discerned this without the benefit of long-term water quality and fire history data. Moreover, the lack of immediate effects of the Mendocino Complex on Clear Lake TP suggests that large watershed fires may also not affect

TP concentrations, but this does not preclude any potential immediate changes in TP missed by the monthly sampling frequency or absence of sampling in 2020 due to the COVID-19 pandemic, changes in other nutrients such as nitrogen and carbon species, or potential further lagged effects in future years.

The lack of post-fire TP increases in Clear Lake may primarily be due to the unique lake morphometry and pre-fire hypereutrophic status of this lake. A study of boreal Alaskan lakes found that nutrient concentrations in shallow, eutrophic lakes as small as 0.5 km^2 may not be sensitive to wildfires up to 2 years post-fire due to large pre-fire nutrient pools (Lewis et al. 2014). Lathrop (1994) found that larger lakes are more well-suited to diluting fire effects, even in cases where 25% of the watershed area was burned, and increases from post-fire runoff might be harder to detect. Clear Lake's large size and hypereutrophic status may mask fire-induced water quality changes despite the recently unprecedented extent of the Mendocino Complex in the watershed. Similarly, such factors may also preclude water quality responses to multiple, smaller wildfires (e.g., 4 wildfires in late summer and fall 1981). Importantly, however, we emphasize that the lack of observed post-fire water quality changes in Clear Lake does not necessarily indicate that other lakes will also not experience wildfire impacts. In particular, less productive lakes may be more sensitive to wildfires than Clear Lake (Enache and Prairie 2000; Williamson et al. 2016). Clearly, ecological context is important, which reinforces the fact that additional studies across a diversity of lakes, landscapes, and fire regimes are necessary to provide more complete and nuanced information when researching or managing impacts of wildfires on lake water quality.

Drivers of long-term total phosphorus concentrations in Clear Lake

Long-term records in Clear Lake allowed us to examine potential climate change effects on TP concentrations that may ultimately be stronger drivers of TP than sporadic, large wildfires. Greater precipitation is commonly expected to increase nutrient concentrations in lakes due to runoff (Hrycik et al. 2021; Mishra et al. 2021), and post-fire runoff has routinely been shown to contain higher concentrations of nutrients (Ranalli 2004). Interestingly, we found significant negative correlations ($p < 0.05$) between median July-October TP and previous November-March precipitation (when the majority of precipitation falls in Mediterranean climates). Furthermore, maximum air temperatures were weakly positively correlated with median July-October TP at the Oaks Arm in the other two sites over the same time period. More generally, maximum air temperature was consistently positively correlated, whereas precipitation was consistently negatively correlated with median July-October TP across all sites and seasons, although sometimes insignificantly. These results suggest that warm, dry conditions are associated with high summer and fall TP concentrations in Clear Lake. This contradicts the long-standing presumption that spring runoff following higher rainfall winters drives increased external sediment and nutrient loading and consequently eutrophication.

Mediterranean-climate lakes such as Clear Lake experience hot and dry summers that feature minimal external inputs, which therefore suggests a potential role of climate warming-driven nutrient resuspension in shaping the long-term TP increase. In other shallow, Mediterranean-climate lakes, phosphorus retention capacity in the sediments has decreased under a warming climate (Özen et al. 2010). Past research has identified that from fall to spring the water column losses coinciding with sediment gains in P indicate the role of both internal and external loading, but during drought years, when external sources are limited, P cycling between sediments and water column increased two-fold (Richerson et al. 1994). In shallow, polymictic lakes that undergo aerated surface exchange, hypoxic indicators may be underestimating the extent of P release from the sediment, particularly in nutrient-rich environments (Nürnberg et al. 2013). Compounding this concept, more frequent wind mixing in shallow lakes may entrain hypolimnetic phosphorus up to the epilimnion, contributing to high TP concentrations at the lake surface (Welch & Cooke 1995).

Lake surface water temperature trends indicate that many lakes are warming at the global scale (Pilla et al. 2020, O'Reilly et al. 2015) and Clear Lake is no exception. Specifically, since around 1980, Clear Lake surface water temperatures visually and statistically started steadily increasing, while trends were decreasing at the beginning of the sampling period (Figure 5, Appendix S1: Figure S8). These warming trends post-1980 may indicate some local or regional alteration or, more likely, coincide with the global temperature regime shift (Reid et al. 2016) that occurred within the same decade and should be explored in detail in future research efforts. Warming surface waters can also intensify the strength and duration of water column stratification, reducing hypolimnetic DO concentrations and increasing internal loading, which can exacerbate eutrophication when water column mixing and hypolimnetic entrainment eventually occur (Jankowski et al. 2006, Jane et al. 2021, Woolway et al. 2020). Interestingly, deep water temperatures also increased in Clear Lake over the study period, lending more probability to the influence of internal loading on TP concentrations due to lower solubility of DO in warmer water. Taken together, these phenomena suggest that ongoing climatic shifts toward more frequent, longer periods of droughts, combined with warm air and water temperatures, while also more conducive to wildfires, promote lake conditions that may lead to higher internal loading of TP.

Whereas TP has increased over the 50-year study period, DO has generally not changed at the lake surface or bottom over this period. Additionally, the frequency of hypoxic and anoxic events in Clear Lake did not vary significantly over the 50-year period, but sampling inconsistencies and reduced sampling frequency in recent years complicate drawing conclusions about changes in the duration of anoxia. Nonetheless, we found some evidence of internal nutrient loading occurring in Clear Lake with some of the highest SRP and TP concentrations occurring at low DO concentrations (Figure 6, Appendix S1: Figure S9). Internal P loading associated with low DO has been well documented in many other shallow, eutrophic lakes (Welch and Cooke 1995, Søndergaard et al. 2003, Søndergaard et al. 2013) and is the focus of current research at Clear Lake. Particularly in baseline eutrophic lakes further enriched through cultural eutrophication, once the feedback loop of internal loading begins it can be hard to reverse (Gächter and Wehrli 1998) and can contribute to delayed external load abatement response (Nürnberg et al. 2013). Therefore, future research in Clear Lake should address full-ecosystem nutrient budgets, internal and external sediment loading, and DO dynamics in response to variable environmental conditions, particularly increasing air and water temperatures and associated shifting lake levels, which are expected for the region with extreme drought-flood cycles (Mishra et al 2021). Such modeling efforts would help determine the volume of the lake that may go hypoxic or anoxic during the summer and to what extent that volume has changed over the study period. Furthermore, these studies would help evaluate the potential ecosystem impacts of reducing external nutrient inputs (Schindler 2012, Paerl et al. 2016), a common management objective, versus measures focused on reducing internal loading (e.g., sediment removal, oxygenation of the water column, and intentional mixing to prevent persistent periods of anoxia; Bormans et al. 2016, Kiani et al. 2020).

The long-term increases in July-October TP in Clear Lake may be linked with other water quality indicators and have implications for phytoplankton biomass and water transparency. Total suspended solids (TSS) and chlorophyll *a* (ChlA) data were collected beginning in 2004 and 2007, but no significant changes were observed for either variable over the short-term period. Both ChlA and TSS were significantly positively correlated with TP in all arms on an annual basis, (Spearman's correlations for ChlA: 0.5-0.65 and TSS: 0.52-0.69; all $p < 0.05$) but the lack of long-term data precluded further analysis in response to wildfires and climate change impacts. Although these short-term associations potentially indicate ecologically important interactions among TP, TSS, and primary productivity, further long-term research is required to disentangle the extent to which wildfires and climate influence these ecosystem properties and expected interactions. This is particularly important to identify direct influences of TP on potential harmful algal blooms (HABs) and bloom impacts (e.g., hypoxia or anoxia) in Clear Lake. Additional research, combining all short and long-term monitoring, along with information provided by enhanced internal loading studies or lab core-incubation studies, would provide more relevant guidance for future research and management of eutrophication drivers in Clear Lake.

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~~While not directly influenced by internal loading, we conducted some additional analysis of TP and short-term chlorophyll A (ChlA) and total suspended solids (TSS) data, and while these constituents were only collected since 2007, and 2004, respectively, we did notice some interesting associations. Spearman's correlations between TP and ChlA (2007-2020) for all months ranged from 0.5-0.65 for all arms (all $p < 0.05$) and ranged between 0.12-0.17 for months between July-October (all $p > 0.05$). Spearman's correlations between TP and TSS (2004-2020)~~

Commented [JAB3]: This isn't really true so I would prefer to delete but I don't know how to better integrate this paragraph with everything else. Also not sure what the main point of the paragraph is and if we can summarize better? I think this was added to appease a reviewer but would be better to try to integrate a little more I think. Definitely felt like it came up out of the blue

Commented [JAB4]: I'm not really sure what this means - on an annual basis? or each month by itself?

~~for all months and all arms ranged from 0.52-0.69 (all $p < 0.05$) and between 0.31 – 0.59 (all $p < 0.05$) for the months between July–October.~~

~~Statistical relationships among TSS, TP, and ChlA are generally reasonable and consistent with known, established principles in limnology. Although these short-term associations potentially indicate ecologically important interactions among TP, TSS, and primary productivity, further long-term research is required to disentangle the extent to which wildfires and climate influence these ecosystem properties and expected interactions. This is particularly important to identify direct influences on potential harmful algal blooms (HABs) and bloom impacts (e.g., hypoxia or anoxia) in Clear Lake. Additional research, combining all short and long-term monitoring, along with information provided by enhanced internal loading studies or lab core-incubation studies, would provide more relevant guidance for future research and management of eutrophication drivers in Clear Lake.~~

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The complex history of fire in the Clear Lake watershed and the western US

The history of wildfire in the Clear Lake watershed in many ways represents a microcosm of the complex, coupled human-natural history of fire in the western US. Prior to large-scale Euro-American settlement (~1850-1900), the combination of mixed-conifer forests at high elevations and shrub vegetation at lower elevations likely produced (and reinforced) variable fire regimes throughout the large Clear Lake watershed. Specifically, estimated mean fire return intervals ranged from 11–55 years throughout the watershed, likely resulting in a heterogeneous patchwork of vegetation composition, vegetation structure, and soil properties (Safford and Van de Water 2012). Throughout the 20th century, however, mean fire return intervals lengthened as much as 90% in forested areas of the watershed (Appendix S1: Fig S10).

Such shifts are likely due to the same fire exclusion practices that resulted in widespread forest densification, homogenization (i.e., shifts toward more abundant and larger patches of less fire-adapted species), and fuel accumulation throughout the western US (Hessburg et al. 2015). In other words, land and fire management practices have produced a landscape template that, compounded by climate change, is more prone to larger and more frequent wildfires, including particularly damaging wildfires such as the Mendocino Complex (Westerling et al. 2006, Dennison et al. 2014). Accordingly, densely forested watersheds that have not recently experienced extensive fires may be accumulating a "fire deficit" and be vulnerable to large, higher-severity fires in the future, particularly under continued climate change (Parks et al. 2015). Although the last approximate 100 years of fire history in the Clear Lake watershed were dominated by relatively small wildfires, the trajectory of increasing fire frequency and the currently anomalous Mendocino Complex may represent a harbinger of future watershed conditions as wildfire activity continues to increase across the western US owing to the "perfect storm" combination of climate change and the legacy of land and fire management.

Although climate change and historical land and fire management practices amplify current and future fire risk in the western US, recent and ongoing human activities also play a major role in increasing exposure of lakes to wildfire. Human-caused ignitions accounted for 84% of wildfire events and 44% of wildfire area burned from 1992-2012 throughout the conterminous US (Balch et al. 2017). Increasing wildfire frequency in the wildland-urban interface (WUI) has been partly attributed to increasing human populations in these fire-prone locations (Radeloff et al. 2018). Such increases have likely contributed to the over 6,000 lake watersheds experiencing wildfire throughout the conterminous US since 1984, including regions outside the western US (e.g., Florida, Southern Great Plains; McCullough et al. 2019). In the case of Clear Lake, mean fire return intervals are as much as 76% shorter in shrublands since 1850 (Appendix S1: Fig S10; Safford and Van de Water 2012), which may be partly due to expanding human activities in the WUI. In California more broadly, increasing wildfire activity, expansion of the WUI, and a growing population (increased from 10 to 40 million since 1950; PPIC 2020) underscore not only the socio-ecological complexity of wildfire issues, but also the growing need to investigate the impacts of wildfire on important aquatic resources such as Clear Lake.

Conclusion: A burning need for long-term monitoring data and synthesis of lake-fire research

The ability to incorporate long-term lake water quality data into this study to provide context for understanding the impacts of wildfires was invaluable. As climate change and large fires increase, the need for consistent and dependable water quality monitoring should be a standard requirement for any lake management team, especially for those trying to identify specific wildfire impacts and charged with planning and executing an effective response. This is especially true as climate change is strongly contributing to extreme droughts, floods, and wildfire events that impact water quality conditions in unpredictable ways (Mishra et al. 2021). A lack of pre- and post-fire monitoring data makes it difficult to identify and contextualize both climatic and wildfire-associated impacts on water quality, further solidifying that consistently collected, long-term data are especially useful for teasing out direct wildfire impacts on water quality vs. those associated with other drivers, such as climate variability. Although our reliance on monthly long-term data may have obscured possible immediate TP responses to wildfires, the highly seasonal runoff patterns in Mediterranean-climate landscapes suggest that targeted, seasonal sampling in the future could help detect potential TP spikes as a result of wildfires.

To build a more comprehensive understanding of the effects of fire on physical, chemical, and biological properties of lakes, broad-scale studies across a diversity of lakes, landscapes, climates, and fire regimes (i.e., variability in fire extent, frequency, and burn severity) are needed. The fact that most existing lake-fire studies were conducted in boreal regions, had only 1-3 years of post-fire water quality data, and often lacked pre-fire water quality data entirely (i.e., relying on nearby reference lakes in unburned watersheds) makes it difficult to predict longer-term responses of lakes to fires, particularly for large, hypereutrophic, Mediterranean-climate lakes such as Clear Lake. Continued development of large, public databases of water quality observations (e.g., LAGOS: Soranno et al. 2017; National Lakes Assessment: Pollard et al. 2018) and remotely sensed water quality (e.g., AquaSat: Ross et al. 2019), and fires (e.g., Monitoring Trends in Burn Severity; Eidenshink et al. 2007) offer emerging opportunities for amassing the necessary data for broad-scale lake-fire studies within the conterminous US. Further, it is necessary to integrate broad-scale observational studies with local- to landscape-scale case studies that can help identify mechanisms of fire effects (or lack of effects) on lake water quality. For example, Scordo et al. (2021) showed that wildfire smoke, which ranges more widely than watershed burned areas, can reduce light availability and consequently lake temperatures and primary productivity. Understanding the mechanisms by which smoke alters lake ecosystems is useful for anticipating future effects of fires on lakes, but this would not be possible with broad-scale, retrospective studies alone that only consider area burned and/or burn severity. While lake-fire management frameworks may not yet exist to facilitate the synthesis of such various studies, lake managers implementing routine or post-fire response monitoring should be encouraged to share their experiences and data with other researchers and agencies to support synthesis of knowledge and research. Such open science research applications will help managers anticipate and respond to the effects of future climate change and increasing fire activity on lakes, improve the design of water resource management plans considering fire and climate adaptations, communicate needs to drinking water system managers, and allocate limited resources more efficiently and effectively.

Acknowledgments

We would like to thank and acknowledge the data providers for this study including the California Department of Water Resources, the State Water Project, and the Data Water Library Database available at http://wdl.water.ca.gov/waterdatalibrary/index.cfm. We also want to acknowledge the California Central Valley Regional Water Quality Control Board for supporting sampling lab analysis from 2004 through 2020. Additional thanks to Sarah Ryan from Big Valley Band of Pomo Indians Environmental Department, Richard Muhl for helping us coordinate and acquire funding for the post-fire response, Amy Little of the California State Water Resources Control Board Drinking Water Division Mendocino District, UC Davis Tahoe Environmental Research Center, and the Schladow Lab. We extend appreciation to the county managers and staff who have helped facilitate the collection of past water quality data including, but not limited to S. De Leon, T. Smythe, W. Evans., M. Deligiannis, R. Carey, L. Healey, W. Fox, J. Thorup, D. Cazares, S. Webb, and researchers involved on the Clean Lakes Report series. Special thanks to P. Jerrett for conducting monthly sampling on behalf of the CDWR. Funding to summarize and analyze data and prepare the manuscript was provided by the Lake County Watershed Protection District, a partnership with Habematolel Pomo of Upper Lake, US EPA Region 9 (NEIEN Grant # 83969501), and the US National Science Foundation Macrosystems

Biology program (EF #1638679). Additional thanks to A. Cortes for your thoughtful review pre-submission.

References

Allen, E. W., E. E. Prepas, S. Gabos, W. Strachan, and W. Chen. 2003. Surface water chemistry of burned and undisturbed watersheds on the Boreal Plain: an ecoregion approach. Journal of Environmental Engineering and Science 2:S73–S86.

Balch, J. K., B. A. Bradley, J. T. Abatzoglou, R. C. Nagy, E. J. Fusco, and A. L. Mahood. 2017. Human-started wildfires expand the fire niche across the United States. Proceedings of the National Academy of Sciences 114:2946–2951.

Bivand, R. and C. Rundel. 2019. Rgeos: Interface to geometry engine - open source ('GEOS'), R package version 0.5–1 https://cran.r-project.org/package=rgeos

Bixby, R. J., S. D. Cooper, R. E. Gresswell, L. E. Brown, C. N. Dahm, and K. A. Dwire. 2015. Fire effects on aquatic ecosystems: an assessment of the current state of the science. Freshwater Science 34:1340–1350.

Bormans, M., B. Maršálek, and D. Jančula. 2016. Controlling internal phosphorus loading in lakes by physical methods to reduce cyanobacterial blooms: a review. Aquatic Ecology 50:407–422.

Bronaugh, D. and A. Werner. 2019. Package 'zyp' 0.10-1.1 for the Pacific Climate Impacts Consortium. R Package version 3.6.3.

California Department of Forestry and Fire Protection (CAL FIRE). 2019. Top 20 Largest California Wildfires. CalFire Statistics and Events https://www.fire.ca.gov/media/5510/top20_acres.pdf

California Department of Forestry and Fire Protection (CAL FIRE). 2021. Fire and Resource Assessment Program (FRAP). https://frap.fire.ca.gov/mapping/gis-data/

California Department of Water Resources. 2002. Bryte Chemical Laboratory Quality Assurance Manual. 2002. Quality Assurance Technical Document 8. Department of Water Resources Publications, Sacramento, California. 48 pp.

California Department of Water Resources. 2015. California's most significant droughts: Comparing historical and recent conditions. Department of Water Resources Publications, Sacramento, California. https://water.ca.gov/LegacyFiles/waterconditions/docs/California_Signficant_Droughts_2015_small.pdf

California Department of Water Resources. 2021. Data Water Library. https://wdl.water.ca.gov/Map.aspx

California Department of Water Resources. 2021. California Disadvantaged Community (DAC) Mapping Tool: Designed to assist with responsibilities related to IRWM, SGMA, and the CA Water Plan. https://gis.water.ca.gov/app/dacs/.

Carey, C. C., B. W. Ibelings, E. P. Hoffmann, D. P. Hamilton, and J. D. Brookes. 2012. Eco-physiological adaptations that favour freshwater cyanobacteria in a changing climate. Water Research 46:1394–1407.

Carignan, R., P. D'Arcy, and S. Lamontagne. 2000. Comparative impacts of fire and forest harvesting on water quality in Boreal Shield lakes. Canadian Journal of Fisheries and Aquatic Sciences 57:105–117.

Cheruvelil, K. S., P. A. Soranno, I. M. McCullough, K. E. Webster, L. K. Rodriguez, and N. J. Smith. 2021. LAGOS-US LOCUS v1.0: Data module of location, identifiers, and physical characteristics of lakes and their watersheds in the conterminous U.S. Limnology and Oceanography Letters 6:270–292.

Clark, C. and T.B. Williams. 1954. Pomo Indians Myths and Some of their Sacred Meanings. Vantage Press, Inc. New York, NY.

Dahm, C. N., R. I. Candelaria-Ley, C. S. Reale, J. K. Reale, and D. J. Van Horn. (2015). Extreme water quality degradation following a catastrophic forest fire. Freshwater Biology, 60:2584-2599.

DePalma-Dow, A., I. M. McCullough, and J. A. Brentrup. (2022). Data and R code for long-term study of fire and climate effects on water quality in Clear Lake, California (1.0) [Data set]. Zenodo. https://doi.org/10.5281/zenodo.6815236

Dennison, P. E., S. C. Brewer, J. D. Arnold, and M. A. Moritz. 2014. Large wildfire trends in the western United States, 1984-2011. Geophysical Research Letters 41:2928–2933.

Eidenshink, J., B. Schwind, K. Brewer, Z.-L. Zhu, B. Quayle, and S. Howard. 2007. A Project for Monitoring Trends in Burn Severity. Fire Ecology 3:3–21.

Enache, M., Y.T. Prairie. 2000. Paleolimnoligical reconstruction of forest fire induced changes in lake biogeochemistry (Lac Francis, Abitibi, Quebec, Canada): Canadian Journal Fisheries and Aquatic Sciences 57(2):146–154.

Feyrer, F., M. Young, O. Patton, and D. Ayers. 2020. Dissolved oxygen controls summer habitat of Clear Lake Hitch (*Lavinia exilicauda chi*), an imperilled potamodromous cyprinid. Ecology of Freshwater Fish 29:188–196.

Gächter, R., and B. Wehrli. 1998. Ten Years of Artificial Mixing and Oxygenation: No Effect on the Internal Phosphorus Loading of Two Eutrophic Lakes. Environmental Science & Technology 32:3659–3665.

Gesch, D. B., G. A. Evans, M. J. Oimoen, and S. Arundel. 2018. The National Elevation Dataset:83–110.

Havens, K. E., and G. Ji. 2018. Multiyear oscillations in depth affect water quality in Lake Apopka. Inland Waters 8:1–9.

Hayes, N. M., M. J. Vanni, M. J. Horgan, and W. H. Renwick. 2015. Climate and land use interactively affect lake phytoplankton nutrient limitation status. Ecology 96:392–402.

Hessburg, P. F., D. J. Churchill, A. J. Larson, R. D. Haugo, C. Miller, T. A. Spies, M. P. North, N. A. Povak, R. T. Belote, P. H. Singleton, W. L. Gaines, R. E. Keane, G. H. Aplet, S. L. Stephens, P. Morgan, P. A. Bisson, B. E. Rieman, R. B. Salter, and G. H. Reeves. 2015. Restoring fire-prone Inland Pacific landscapes: seven core principles. Landscape Ecology 30:1805–1835.

Hijmans, R.J. 2019. Spatial Data Science with R. https://rspatial.org/

Hislop, S., A. Haywood, S. Jones, M. Soto-Berelov, A. Skidmore, and T. H. Nguyen. 2020. A satellite data driven approach to monitoring and reporting fire disturbance and recovery across boreal and temperate forests. International Journal of Applied Earth Observation and Geoinformation 87:102034.

Hohner, A. K., C. C. Rhoades, P. Wilkerson, and F. L. Rosario-Ortiz. 2019. Wildfires Alter Forest Watersheds and Threaten Drinking Water Quality. Accounts of Chemical Research 52:1234–1244.

Holden, Z. A., A. Swanson, C. H. Luce, W. M. Jolly, M. Maneta, J. W. Oyler, D. A. Warren, R. Parsons, and D. Affleck. 2018. Decreasing fire season precipitation increased recent western US forest wildfire activity. Proceedings of the National Academy of Sciences, 115:E8349-E8357.

Horne, A.J. 1975. The ecology of Clear Lake phytoplankton. Clear Lake Algal Research Unit Special Report, Lakeport, CA.

Hrycik, A. R., P. D. F. Isles, R. Adrian, M. Albright, L. C. Bacon, S. A. Berger, R. Bhattacharya, H. Grossart, J. Hejzlar, A. L. Hetherington, L. B. Knoll, A. Laas, C. P. McDonald, K. Merrell, J. C. Nejstgaard, K. Nelson, P. Nõges, A. M. Paterson, R. M. Pilla, D. M. Robertson, L. G. Rudstam, J. A. Rusak, S. Sadro, E. A. Silow, J. D. Stockwell, H. Yao, K. Yokota, and D. C. Pierson. 2021.

Earlier winter/spring runoff and snowmelt during warmer winters lead to lower summer chlorophyll- *a* in north temperate lakes. Global Change Biology 27:4615–4629.
Jane, S. F., G. J. A. Hansen, B. M. Kraemer, P. R. Leavitt, J. L. Mincer, R. L. North, R. M. Pilla, J. T. Stetler, C. E. Williamson, R. I. Woolway, L. Arvola, S. Chandra, C. L. DeGasperi, L. Diemer, J. Dunalska, O. Erina, G. Flaim, H.-P. Grossart, K. D. Hambright, C. Hein, J. Hejzlar, L. L. Janus, J.-P. Jenny, J. R. Jones, L. B. Knoll, B. Leoni, E. Mackay, S.-I. S. Matsuzaki, C. McBride, D. C. Müller-Navarra, A. M. Paterson, D. Pierson, M. Rogora, J. A. Rusak, S. Sadro, E. Saulnier-Talbot, M. Schmid, R. Sommaruga, W. Thiery, P. Verburg, K. C. Weathers, G. A. Weyhenmeyer, K. Yokota, and K. C. Rose. 2021. Widespread deoxygenation of temperate lakes. Nature 594:66–70.
Jankowski, T., D. M. Livingstone, H. Bührer, R. Forster, and P. Niederhauser. 2006. Consequences of the 2003 European heat wave for lake temperature profiles, thermal stability, and hypolimnetic oxygen depletion: Implications for a warmer world. Limnology and Oceanography 51:815–819.
Kiani, M., P. Tammeorg, J. Niemistö, A. Simojoki, and O. Tammeorg. 2020. Internal phosphorus loading in a small shallow Lake: Response after sediment removal. Science of The Total Environment 725:138279.
Lake County Water Resources Department. 2018. Quality Assurance Project Plan for Lake Chlorophyll A and Sediment Nutrient Sampling in Clear Lake, CA. County of Lake. http://www.lakecountyca.gov/Assets/Departments/WaterResources/Algae+Images/LC+Quality+Assurance+Plan.pdf
Lathrop, R. G. 1994. Impacts of the 1988 wildfires on the water quality of Yellowstone and Lewis Lakes, Wyoming. International Journal of Wildland Fire, 4:169-175.
Lewis, T. L., M. S. Lindberg, J. A. Schmutz, and M. R. Bertram. 2014. Multi-trophic resilience of boreal lake ecosystems to forest fires. Ecology 95:1253–1263.
Lundquist E., T. Smythe. 2010. Clear Lake Integrated Watershed Management Plan. Calfed Watershed Program. http://www.lakecountyca.gov/Government/Directory/WaterResources/database/watershedplan/CLIWMP_Docs.htm
Mauldin, H. K. 1960. Your lakes, valleys, and mountains. History of Lake County. Vol. 1. Clear Lake and Mt. Konocti. San Francisco: East Wind Printers. 64 pp
McColl, J. G., and D. F. Grigal. 1977. Nutrient Changes Following a Forest Wildfire in Minnesota: Effects in Watersheds with Differing Soils. Oikos 28:105.
McCullough, I. M., K. S. Cheruvelil, J. Lapierre, N. R. Lottig, M. A. Moritz, J. Stachelek, and P. A. Soranno. 2019. Do lakes feel the burn? Ecological consequences of increasing exposure of lakes to fire in the continental United States. Global Change Biology 25:2841–2854.
Mishra, A., A. Alnahit, and B. Campbell. 2021. Impact of land uses, drought, flood, wildfire, and cascading events on water quality and microbial communities: A review and analysis. Journal of Hydrology 596:125707.
National Interagency Fire Center. 2018. National Large Incident Year-To-Date Report; https://gacc.nifc.gov/sacc/predictive/intelligence/NationalLargeIncidentYTDReport.pdf
Nürnberg, G. K. 1995. Quantifying anoxia in lakes. Limnology and Oceanography 40:1100–1111.
Nürnberg, G. K., B. D. LaZerte, P. S. Loh, and L. A. Molot. 2013. Quantification of internal phosphorus load in large, partially polymictic and mesotrophic Lake Simcoe, Ontario. Journal of Great Lakes Research 39:271–279.
O'Reilly, C. M., S. Sharma, D. K. Gray, S. E. Hampton, J. S. Read, R. J. Rowley, P. Schneider, J. D. Lenters, P. B. McIntyre, B. M. Kraemer, G. A. Weyhenmeyer, D. Straile, B. Dong, R. Adrian,

M. G. Allan, O. Anneville, L. Arvola, J. Austin, J. L. Bailey, J. S. Baron, J. D. Brookes, E. Eyto, M. T. Dokulil, D. P. Hamilton, K. Havens, A. L. Hetherington, S. N. Higgins, S. Hook, L. R. Izmest'eva, K. D. Joehnk, K. Kangur, P. Kasprzak, M. Kumagai, E. Kuusisto, G. Leshkevich, D. M. Livingstone, S. MacIntyre, L. May, J. M. Melack, D. C. Mueller-Navarra, M. Naumenko, P. Noges, T. Noges, R. P. North, P. Plisnier, A. Rigosi, A. Rimmer, M. Rogora, L. G. Rudstam, J. A. Rusak, N. Salmaso, N. R. Samal, D. E. Schindler, S. G. Schladow, M. Schmid, S. R. Schmidt, E. Silow, M. E. Soylu, K. Teubner, P. Verburg, A. Voutilainen, A. Watkinson, C. E. Williamson, and G. Zhang. 2015. Rapid and highly variable warming of lake surface waters around the globe. Geophysical Research Letters 42, 10: 773-781.

Orihel, D. M., H. M. Baulch, N. J. Casson, R. L. North, C. T. Parsons, D. C. M. Seckar, and J. J. Venkiteswaran. 2017. Internal phosphorus loading in Canadian fresh waters: a critical review and data analysis. Canadian Journal of Fisheries and Aquatic Sciences 74:2005–2029.

Osaka, K., R. Yokoyama, T. Ishibashi, and N. Goto. 2022. Effect of dissolved oxygen on nitrogen and phosphorus fluxes from lake sediments and their thresholds based on incubation using a simple and stable dissolved oxygen control method. Limnology and Oceanography: Methods 20:1–14.

Özen, A., B. Karapınar, İ. Kucuk, E. Jeppesen, and M. Beklioglu. 2010. Drought-induced changes in nutrient concentrations and retention in two shallow Mediterranean lakes subjected to different degrees of management. Hydrobiologia 646:61–72.

Paerl, H. W., J. T. Scott, M. J. McCarthy, S. E. Newell, W. S. Gardner, K. E. Havens, D. K. Hoffman, S. W. Wilhelm, and W. A. Wurtsbaugh. 2016. It Takes Two to Tango: When and Where Dual Nutrient (N & P) Reductions Are Needed to Protect Lakes and Downstream Ecosystems. Environmental Science & Technology 50:10805–10813.

Parks, S. A., C. Miller, M.-A. Parisien, L. M. Holsinger, S. Z. Dobrowski, and J. Abatzoglou. 2015. Wildland fire deficit and surplus in the western United States, 1984–2012. Ecosphere 6:art275.

Pilla, R. M., C. E. Williamson, B. V. Adamovich, R. Adrian, O. Anneville, S. Chandra, W. Colom-Montero, S. P. Devlin, M. A. Dix, M. T. Dokulil, E. E. Gaiser, S. F. Girdner, K. D. Hambright, D. P. Hamilton, K. Havens, D. O. Hessen, S. N. Higgins, T. H. Huttula, H. Huuskonen, P. D. F. Isles, K. D. Joehnk, I. D. Jones, W. B. Keller, L. B. Knoll, J. Korhonen, B. M. Kraemer, P. R. Leavitt, F. Lepori, M. S. Luger, S. C. Maberly, J. M. Melack, S. J. Melles, D. C. Müller-Navarra, D. C. Pierson, H. V. Pislegina, P.-D. Plisnier, D. C. Richardson, A. Rimmer, M. Rogora, J. A. Rusak, S. Sadro, N. Salmaso, J. E. Saros, É. Saulnier-Talbot, D. E. Schindler, M. Schmid, S. V. Shimaraeva, E. A. Silow, L. M. Sitoki, R. Sommaruga, D. Straile, K. E. Strock, W. Thiery, M. A. Timofeyev, P. Verburg, R. D. Vinebrooke, G. A. Weyhenmeyer, and E. Zadereev. 2020. Deeper waters are changing less consistently than surface waters in a global analysis of 102 lakes. Scientific Reports 10:20514.

Pollard, A. I., S. E. Hampton, and D. M. Leech. 2018. The Promise and Potential of Continental-Scale Limnology Using the U.S. Environmental Protection Agency's National Lakes Assessment. Limnology and Oceanography Bulletin 27:36–41.

Public Policy Institute of California. 2020. Just the Facts: California's Populations https://www.ppic.org/publication/californias-population/

PRISM Climate Group 2004, February 4. Oregon State University. http://prism.oregonstate.edu.

Radeloff, V. C., D. P. Helmers, H. A. Kramer, M. H. Mockrin, P. M. Alexandre, A. Bar-Massada, V. Butsic, T. J. Hawbaker, S. Martinuzzi, A. D. Syphard, and S. I. Stewart. 2018. Rapid growth of the US wildland-urban interface raises wildfire risk. Proceedings of the National Academy of Sciences 115:3314–3319.

Ranalli, A.J. 2004. A Summary of the Scientific Literature on the Effects of Fire on the Concentration of Nutrients in Surface Waters: U.S. Geological Survey Open-File Report 2004-1296, 23 p.
R Development Core Team. 2019. A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.
Reid, P. C., R. E. Hari, G. Beaugrand, D. M. Livingstone, C. Marty, D. Straile, J. Barichivich, E. Goberville, R. Adrian, Y. Aono, R. Brown, J. Foster, P. Groisman, P. Hélaouët, H.-H. Hsu, R. Kirby, J. Knight, A. Kraberg, J. Li, T.-T. Lo, R. B. Myneni, R. P. North, J. A. Pounds, T. Sparks, R. Stübi, Y. Tian, K. H. Wiltshire, D. Xiao, and Z. Zhu. 2016. Global impacts of the 1980s regime shift. Global Change Biology 22:682–703.
Richerson, P.J., T.H. Suchanek, S.J. Why.1994. The causes and control of algal blooms in Clear Lake: clean lakes diagnostic/feasibility study for Clear Lake, California. Final Report. Division of Environmental Studies, University of California, Davis, California, USA.
Richerson, P. J., T. H. Suchanek, R. A. Zierenberg, D. A. Osleger, A. C. Heyvaert, D. G. Slotton, C. A. Eagles-Smith, and C. E. Vaughn. 2008. Anthropogenic Stressors and changes in the Clear Lake Ecosystem as Recorded in Sediment Cores. Ecological Applications 18:A257–A283.
Roces-Díaz, J. V., C. Santín,, J. Martínez-Vilalta, and S. H. Doerr. 2022. A global synthesis of fire effects on ecosystem services of forests and woodlands. Frontiers in Ecology and the Environment, 20: 170-178.
Ross, M. R. V., S. N. Topp, A. P. Appling, X. Yang, C. Kuhn, D. Butman, M. Simard, and T. M. Pavelsky. 2019. AquaSat: A Data Set to Enable Remote Sensing of Water Quality for Inland Waters. Water Resources Research 55:10012–10025.
Rueda, F. J., S. G. Schladow, S. G. Monismith, and M. T. Stacey. 2005. On the effects of topography on wind and the generation of currents in a large multi-basin lake. Hydrobiologia 532:139–151.
Rust, A. J., T. S. Hogue, S. Saxe, and J. McCray. 2018. Post-fire water-quality response in the western United States. International Journal of Wildland Fire, 27:203-216.
Safford, H. D., and K. M. V. de Water. 2014. Using fire return interval departure (FRID) analysis to map spatial and temporal changes in fire frequency on national forest lands in California. Res. Pap. PSW-RP-266. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 59 p 266.
Schindler, D. W. 2012. The dilemma of controlling cultural eutrophication of lakes. Proceedings of the Royal Society B: Biological Sciences 279:4322–4333.
Scordo, F., S. Chandra, E. Suenaga, S. J. Kelson, J. Culpepper, L. Scaff, F. Tromboni, T. J. Caldwell, C. Seitz, J. E. Fiorenza, C. E. Williamson, S. Sadro, K. C. Rose, and S. R. Poulson. 2021. Smoke from regional wildfires alters lake ecology. Scientific Reports, 11:1-14.
Sharp, S. L., A. L. Forrest, K. Bouma-Gregson, Y. Jin, A. Cortés, and S. G. Schladow. 2021. Quantifying Scales of Spatial Variability of Cyanobacteria in a Large, Eutrophic Lake Using Multiplatform Remote Sensing Tools. Frontiers in Environmental Science 9:612934.
Smith, H. G., G. J. Sheridan, P. N. J. Lane, P. Nyman, and S. Haydon. 2011. Wildfire effects on water quality in forest catchments: A review with implications for water supply. Journal of Hydrology 396:170–192.
Smith, N. J., K. E. Webster, L. K. Rodriguez, K. S. Cheruvelil, and P. A. Soranno. 2021. LAGOS-US LOCUS v1.0: Data module of location, identifiers, and physical characteristics of lakes and their watersheds in the conterminous U.S. Environmental Data Initiative.
Søndergaard, M., R. Bjerring, and E. Jeppesen. 2013. Persistent internal phosphorus loading during summer in shallow eutrophic lakes. Hydrobiologia 710:95–107.

Søndergaard, M., J. P. Jensen, and E. Jeppesen. 2003. Role of sediment and internal loading of phosphorus in shallow lakes. Hydrobiologia 506–509:135–145.

Soranno, P. A., L. C. Bacon, M. Beauchene, K. E. Bednar, E. G. Bissell, C. K. Boudreau, M. G. Boyer, M. T. Bremigan, S. R. Carpenter, J. W. Carr, K. S. Cheruvelil, S. T. Christel, M. Claucherty, S. M. Collins, J. D. Conroy, J. A. Downing, J. Dukett, C. E. Fergus, C. T. Filstrup, C. Funk, M. J. Gonzalez, L. T. Green, C. Gries, J. D. Halfman, S. K. Hamilton, P. C. Hanson, E. N. Henry, E. M. Herron, C. Hockings, J. R. Jackson, K. Jacobson-Hedin, L. L. Janus, W. W. Jones, J. R. Jones, C. M. Keson, K. B. S. King, S. A. Kishbaugh, J.-F. Lapierre, B. Lathrop, J. A. Latimore, Y. Lee, N. R. Lottig, J. A. Lynch, L. J. Matthews, W. H. McDowell, K. E. B. Moore, B. P. Neff, S. J. Nelson, S. K. Oliver, M. L. Pace, D. C. Pierson, A. C. Poisson, A. I. Pollard, D. M. Post, P. O. Reyes, D. O. Rosenberry, K. M. Roy, L. G. Rudstam, O. Sarnelle, N. J. Schuldt, C. E. Scott, N. K. Skaff, N. J. Smith, N. R. Spinelli, J. Stachelek, E. H. Stanley, J. L. Stoddard, S. B. Stopyak, C. A. Stow, J. M. Tallant, P.-N. Tan, A. P. Thorpe, M. J. Vanni, T. Wagner, G. Watkins, K. C. Weathers, K. E. Webster, J. D. White, M. K. Wilmes, and S. Yuan. 2017. LAGOS-NE: a multi-scaled geospatial and temporal database of lake ecological context and water quality for thousands of US lakes. GigaScience 6:gix101.

Suchanek, T.H., P.J. Richerson, D.C. Nelson, C.A. Eagles-Smith, D.W. Anderson, J.J. Cech, G. Schladow, R. Zierenberg, J.F. Mount, S.C. McHatton, D.G. Slotton, L.B. Webber, A.L. Bern, B.L. Swisher. 2002. Evaluating and managing a multiply-stressed ecosystem at Clear Lake, California: A holistic ecosystem approach. pp. 1233-1265 *in* Rapport, D., editor. 2003. Managing for healthy ecosystems. Lewis Publishers, Boca Raton

Suchanek, T. H., C. A. Eagles-Smith, D. G. Slotton, E. J. Harner, D. P. Adam, A. E. Colwell, N. L. Anderson, and D. L. Woodward. 2008. Mine-Derived Mercury: Effects on Lower Trophic Species in Clear Lake, California. Ecological Applications 18:A158–A176.

Sugihara, N. G., J. W. Van Wagtendonk, J. Fites-Kaufman, K. E. Shaffer, and A. E. Thode. 2006. Fire in California's ecosystems. University of California Press.

Tetra Tech. 2004. Total Maximum Daily Load for Nutrients in Clear Lake, Lake County, California Technical Report Final. December 1st, 2004. Prepared for the Central Valley Regional Water Quality Control Board. https://www.waterboards.ca.gov/rwqcb5/water_issues/tmdl/central_valley_projects/clear_lake_nutrients/cl_nut_tech_tmdl.pdf

Thompson, L.C., G.A. Giusti, K.L.Weber, R.F. Keiffer. 2013. The native and introduced fishes of Clear Lake: a review of the past to assist with decisions of the future. California Fish and Game Bulletin 99(1):7-41

Welch, E. B., and G. D. Cooke. 1995. Internal Phosphorus Loading in Shallow Lakes: Importance and Control. Lake and Reservoir Management 11:273–281.

Westerling, A. L., H. G. Hidalgo, D. R. Cayan, and T. W. Swetnam. 2006. Warming and Earlier Spring Increase Western U.S. Forest Wildfire Activity. Science 313:940–943.

Williams, A. P., B. Livneh, K. A. McKinnon, W. D. Hansen, J. S. Mankin, B. I. Cook, J. E. Smerdon, A. M. Varuolo-Clarke, N. R. Bjarke, C. S. Juang,, and D. P. Lettenmaier. 2022. Growing impact of wildfire on western US water supply. Proceedings of the National Academy of Sciences, 119:e2114069119.

Williamson, C. E., E. P. Overholt, J. A. Brentrup, R. M. Pilla, T. H. Leach, S. G. Schladow, J. D. Warren, S. S. Urmy, S. Sadro, S. Chandra, and P. J. Neale. 2016. Sentinel responses to droughts, wildfires, and floods: effects of UV radiation on lakes and their ecosystem services. Frontiers in Ecology and the Environment 14:102–109.

Woolway, R. I., E. J. Anderson, and C. Albergel. 2021. Rapidly expanding lake heatwaves under climate change. Environmental Research Letters, 16:094013.
Woolway, R. I., B. M. Kraemer, J. D. Lenters, C. J. Merchant, C. M. O'Reilly, and S. Sharma. 2020. Global lake responses to climate change. Nature Reviews Earth & Environment 1:388–403.
Wurtsbaugh, W. A., and A. J. Horne. 1983. Iron in Eutrophic Clear Lake, California: Its Importance for Algal Nitrogen Fixation and Growth. Canadian Journal of Fisheries and Aquatic Sciences 40:1419–1429.
Yang, L., S. Jin, P. Danielson, C. Homer, L. Gass, S. M. Bender, A. Case, C. Costello, J. Dewitz, J. Fry, M. Funk, B. Granneman, G. C. Liknes, M. Rigge, and G. Xian. 2018. A new generation of the United States National Land Cover Database: Requirements, research priorities, design, and implementation strategies. ISPRS Journal of Photogrammetry and Remote Sensing 146:108–123.

Table 1. Pre- and post-fire July-October total phosphorus (mg/L) data in Clear Lake for 1981, 1996, and 2018 fire years. Bolded row indicates fire year. No Data for 2020 due to boating restrictions.

1981 fire year	**Upper**			**Lower**			**Oaks**		
	Min	**Median**	**Max**	**Min**	**Median**	**Max**	**Min**	**Median**	**Max**
1978	0.03	0.12	0.16	0.02	0.05	0.07	0.02	0.07	0.10
1979	0.04	0.16	0.17	0.03	0.05	0.06	0.02	0.09	0.13
1980	0.02	0.10	0.15	0.02	0.03	0.03	0.02	0.04	0.15
1981	**0.03**	**0.09**	**0.42**	**0.03**	**0.06**	**0.08**	**0.03**	**0.08**	**0.34**
1982	0.03	0.07	0.31	0.04	0.04	0.12	0.02	0.06	0.16
1983	0.03	0.11	0.15	0.02	0.04	0.06	0.03	0.05	0.07
1984	0.10	0.12	0.26	0.03	0.05	0.13	0.02	0.06	0.17

1996 fire year	**Upper**			**Lower**			**Oaks**		
	Min	**Median**	**Max**	**Min**	**Median**	**Max**	**Min**	**Median**	**Max**
1993	0.12	0.22	0.30	0.14	0.23	0.24	0.18	0.21	0.24
1994	0.19	0.46	0.62	0.08	0.20	0.26	0.28	0.35	0.44
1995	0.12	0.35	0.41	0.08	0.11	0.12	0.08	0.20	0.31
1996	**0.13**	**0.32**	**0.44**	**0.10**	**0.13**	**0.97**	**0.16**	**0.22**	**0.23**
1997	0.07	0.33	0.44	0.10	0.20	0.41	0.20	0.24	0.39
1998	0.10	0.12	0.22	0.03	0.06	0.08	0.04	0.09	0.20
1999	0.07	0.13	0.15	0.02	0.04	0.10	0.05	0.12	0.12

2018 fire year	**Upper**			**Lower**			**Oaks**		
	Min	**Median**	**Max**	**Min**	**Median**	**Max**	**Min**	**Median**	**Max**
2015	0.64	0.73	0.79	0.38	0.42	0.52	0.59	0.65	0.70
2016	0.49	0.76	1.04	0.38	0.53	0.65	0.40	0.62	0.78
2017	0.24	0.30	0.45	0.18	0.25	0.29	0.17	0.28	0.39
2018	**0.22**	**0.47**	**0.56**	**0.14**	**0.27**	**0.34**	**0.16**	**0.38**	**0.48**
2019*	0.34	0.34	0.34	0.13	0.13	0.13	0.19	0.19	0.19
2021**	0.28	0.42	0.55	0.13	0.16	0.20	0.55	0.58	0.60

* Single August 2019 measurement at each arm; all other years had a minimum of 3 measurements per arm; ** July and August 2021 only

Table 2. Total number of dissolved oxygen readings and percent of readings that were hypoxic (1-2 mg/L) or anoxic (<1 mg/L) from July-October for each decade in the Upper, the Lower, and the Oaks Arm of Clear Lake.

	Upper			Lower			Oaks		
Time Period	**Total Samples**	**% Hypoxic**	**% Anoxic**	**Total Samples**	**% Hypoxic**	**% Anoxic**	**Total Samples**	**% Hypoxic**	**% Anoxic**
1968-1970	28	11	21	29	10	48	27	4	30
1971-1980	66	12	27	65	14	23	65	9	25
1981-1990	38	5	21	38	5	26	38	18	16
1991-2000	34	15	15	37	8	30	35	6	23
2001-2010	33	3	24	33	6	21	33	3	15
2011-2019	26	12	23	26	8	27	26	0	23
Total	228	9	28	224	8	22	225	10	23

Table 3. Spearman's rho correlation coefficients between seasonal climate and median July-October total phosphorus (mg/L) data in Clear Lake from 1968-2019. Tmin = mean air temperature minimum, Tmax = mean air temperature maximum, Precip = total precipitation. See Appendix S1: Figures S5-S7 for scatterplot data.

Season	Climate variable	Upper	Lower	Oaks
Nov-Mar	Tmin	-0.08	-0.09	-0.09
	Tmax	0.22	*0.26*	*0.24*
	Precip	**-0.30**	**-0.34**	**-0.31**
Apr-Jun	Tmin	0.18	*0.25*	0.19
	Tmax	0.14	0.22	0.18
	Precip	-0.06	-0.02	-0.03
Jul-Oct	Tmin	-0.11	-0.06	-0.12
	Tmax	0.22	0.19	**0.28**
	Precip	**-0.30**	-0.21	*-0.25*

bold: $p < 0.05$, *italics*: $p < 0.1$

Figure Legends.

Figure 1. (a) Clear Lake and its watershed in relation to the 2018 Mendocino Complex Fire (Ranch and River fires). These wildfires collectively burned 40% of the Clear Lake watershed from July-November 2018. Background is a false color composite (red, green, blue (RGB) 7, 5, 2) Landsat 8 (satellite) image from November 15, 2018, approximately 1 week after the Complex ceased. Brown indicates fire-scarred vegetation. (b) Wildfire extent in the Clear Lake watershed in 1981 (12% burned) and 1996 (7% burned), the next largest fire years besides the 2018 fire year.

Figure 2. Annual proportion of the Clear Lake watershed burned by wildfires from 1923-2020. Note that 1981, 1996, and 2018 were the three biggest fire years in the watershed. Source: CAL FIRE.

Figure 3. Total phosphorus (TP) concentration data in the (a) Upper Arm, (b) Lower Arm, and (c) Oaks Arm for the three largest wildfire years based on annual proportion watershed burned (1981: 0.12, 1996: 0.09, 2018: 0.40) in the Clear Lake watershed. Boxplots show July–October

surface TP data (interquartile range, median, and outliers) from all sites for the 3 years prior to the fire (blue) compared to three years post-fire (red). For the 2018 fire year, no post-fire data were available for 2020.

Figure 4. Median surface total phosphorus (TP) concentration data from July-October in Clear Lake in the (a) Upper Arm, (b) Lower Arm, and (c) Oaks Arm from 1968-2021 (note: no data were available for 2020).

Figure 5. Median surface (≤ 3 m) water temperature data from July-October in Clear Lake in the (a) Upper Arm, (b) Lower Arm, and (c) Oaks Arm from 1968-2019. Median deep water temperature data for the same time period for all Arms is available in Appendix 1: Figure S8.

Figure 6. Long-term surface (light blue) and bottom (dark blue) median dissolved oxygen (DO) concentration data for July-October for the (a) Upper Arm, (b) Lower Arm, and (c) Oaks Arm from 1968–2019 in Clear Lake. The dotted gray line is an indicator of hypoxia (< 2 mg/L) and the solid gray line is an indicator of anoxia (< 1 mg/L).

Figure 7. Correlation between hypoxic dissolved oxygen (DO) concentration data (< 2 mg/L) from July–October to soluble reactive phosphorus (SRP) concentration data from the same depth for the (a) Upper Arm, (b) Lower Arm, and (c) Oaks Arm for 1968-2019 (n = 169 total samples: n = 54 at Upper, n = 62 at Lower, n = 53 at Oaks). *r* = Spearman's rho. Correlations for DO compared to TP are provided in Appendix 1: Figure S9.