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Cite this: *Environ. Sci.: Adv.*, 2025, **4**, 2138

Climate variability and flow management impact phytoplankton biomass in a shallow reservoir

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Received 7th April 2025
Accepted 24th October 2025

DOI: 10.1039/d5va00094g

rsc.li/esadvances

Shallow, eutrophic lakes often exhibit high and extremely variable phytoplankton biomass. This variability makes drinking water supply from shallow lakes particularly vulnerable to rapid change, as phytoplankton blooms can strongly impact treatment processes. Using 39 years of water quality data (typically bi-weekly), this study investigates the roles of climate variability and flow management in driving change in chlorophyll *a* (an indicator of phytoplankton biomass) in a shallow, hydrologically managed drinking water reservoir. Generalized additive modelling shows a significant increase in phytoplankton biomass (1992–1997), leading to a doubling of average chlorophyll *a* concentrations. Interestingly, we also see a shift towards smaller spring blooms and larger summer blooms. Our results show this variability in phytoplankton is largely related to climate variability, nutrients, and flow source. Specifically, the increase in phytoplankton biomass coincided with periods with greater precipitation and associated nutrient-rich inflows from an agriculturally dominated catchment, along with strong El Niño events that potentially contributed to a warm, stable water column during the growing season. During high flows from the local catchment, flows from the upstream supply reservoir are typically reduced to prevent downstream flooding. However, flows from the supply reservoir have a diluting impact due to its lower nutrient concentrations. Thus, where reservoir flow sources vary in chemistry, considering water quality in flow management could help to ameliorate bloom severity and reduce drinking water treatment costs. Importantly, understanding management and climatic drivers of reservoir water quality contributes to advance planning and mitigation of risks to water treatment.

Environmental significance

Understanding and anticipating risks to water treatment is critical to ensuring reliable and affordable drinking water and achieving UN SDG 6: clean water and sanitation. An important risk to water treatment is rapid changes in water quality, including increases in phytoplankton biomass. Here, we identify the conditions leading to a major increase in phytoplankton biomass and show how hydrological management can help manage source waters. Understanding and anticipating the conditions contributing to increases in phytoplankton will be critical to ensuring drinking water security, especially for shallow drinking water sources experiencing large climatic and hydrologic changes.

1. Introduction

Cultural eutrophication and algal blooms have been worsening in many lakes and reservoirs globally,[1–4] including in shallow, eutrophic lakes, where phytoplankton biomass is often both high and extremely variable.[5–8] Eutrophication and the risk of bloom formation can be exacerbated by point source pollution, catchment land use, and climate change.[9–11] Conversely, eutrophication and blooms can be abated through management, which can include control of nutrient sources and, in some cases, hydrological manipulation *via* altering flow velocity and turbulence.[1,12,13] However, the source, chemistry, and volume of inflows are important to water quality. Understanding the conditions that have contributed to increases in phytoplankton biomass and opportunities for mitigation are critical to the provision of safe, affordable drinking water.

Phytoplankton biomass is strongly affected by climate variability, for example through changes in water temperature and precipitation, and associated impacts on hydrology and hydrologic management. Higher temperatures are known to favour certain phytoplankton, including cyanobacteria.[14,15] Higher catchment runoff is often associated with influxes of nutrients and organic matter, especially in regions where watersheds are dominated by agricultural land use.[16–18] In contrast, flows from

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upstream supply reservoirs can improve downstream water quality *via* multiple effects which can include impacts on temperature and stratification, or by diluting or flushing nutrient-rich waters,[19–22] creating an important opportunity for bloom management. However, these benefits are dependent on the quality of flows from supply reservoirs relative to downstream systems,[21,23] as well as on water availability and infrastructure capacity. For example, controlled flows may be paused or minimized to prevent downstream flooding during particularly wet periods.[12,17,24]

Globally, an important driver of interannual climate variability is large-scale climate oscillations, such as the El Niño-Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO).[25–29] On the Canadian Prairies (a region spanning the provinces of Alberta, Saskatchewan, and Manitoba), for example, both ENSO and PDO are strong drivers of climate variability.[29–32] Here, the positive ENSO phase, or La Niña, is characterized by cooler, wetter conditions, with later initiation of snowmelt.[30] In contrast, the negative phase of ENSO, commonly referred to as El Niño, is associated with warmer temperatures, slower surface wind speeds, and moisture deficits.[31,32] Additionally, PDO has a positive (warm) and negative (cool) phase that can enhance ENSO effects when in-phase.[29,30] Hence, the warm phases of ENSO and PDO are typically associated with low natural streamflow and thus lower catchment inputs, whereas the cold phases are typically associated with higher regional runoff.[29] Although ENSO and PDO typically have stronger influences on winter conditions,[29] these cycles can last several months to a year (ENSO) or longer (PDO) and can contribute to moisture deficits or surpluses in summer.[33] These changes in temperature and flows have important implications for nutrient loading, that combined, can affect the distribution, timing, and severity of algal blooms, including blooms of cyanobacteria and other taxa such as taste and odor forming chlorophytes (*e.g.*, ref. 14, 30, and 31). However, few studies have assessed the combined effects of flow source and interannual climate variability—linked to large-scale climate oscillations—on long-term phytoplankton dynamics.[34,35,52,59]

Changes in bloom dynamics create critical challenges for water treatment, especially in regions where watersheds are dominated by agricultural land use, thus commonly experience periods of high nutrient influx and blooms. For example, high phytoplankton biomass and associated high turbidity can foul equipment such as filters and increase the level of disinfection byproducts from chlorination.[8,36] In addition to increasing the costs to treat water for domestic, industrial, and agricultural uses,[37–40] high phytoplankton biomass can lead to temporary water shortages associated with treatment problems,[17] or—in extreme events—shutdowns at water treatment plants and drinking water advisories.[39] Concerningly, climate change is expected to exacerbate eutrophication and algal blooms.[1,3,4,10] Thus, without intervention, water shortages and shutdowns at water treatment plants are likely to occur more frequently.

Here, we use 39 year record of chlorophyll *a*, an indicator of phytoplankton biomass, in an important drinking water reservoir in Saskatchewan, Canada (Buffalo Pound Lake). This water supply is in a relatively water scarce, dryland region and has been subject to substantive variation in climate. Hydrological management of the system, which includes inter-basin transfers from a mesotrophic (lower nutrient) supply reservoir, creates a unique opportunity to observe how water management decisions and hydrological variability, linked to interannual climate variability, have influenced phytoplankton biomass. Indeed, use of such a long-term dataset enables analysis of these relationships over the timescale in which some of the climate oscillations operate (decades). This rare combination of long-term ecological monitoring that captures signals of interannual climate variability and source water manipulation provides valuable insights into the potential risks and opportunities for ensuring safe, reliable drinking water.

2. Methods

2.1. Study site

Buffalo Pound Lake (BPL) in Saskatchewan, Canada (Fig. 1) serves as the main source of drinking water for approximately 25% of Saskatchewan's population.[41] BPL is a shallow, polymictic reservoir with a mean depth of 3 m, a length of 29 km, and a width of 1 km.[42] The outlet of the BPL was dammed in 1952, raising the water level by approximately two meters and reducing water level fluctuations to less than a meter per year.[43] Catchment land-use is dominated by agriculture, including crops and livestock operations.[44] Importantly, BPL is eutrophic and often dominated by cyanobacteria in the summer months, including *Dolichospermum* (formerly *Anabaena* spp.), *Microcystis*, *Oscillatoria*, and *Aphanizomenon*.[43,45,46] During prolonged warm, calm conditions, BPL can experience transient stratification events (lasting hours to days) and bottom waters can become anoxic.[17,47] These brief periods of stratification may increase internal nutrient loading—although internal P loading may also occur under oxic conditions.[47,48] Depending on volumes of flow from the supply reservoir, Lake Diefenbaker, as well as watershed sources and occasional backflows from the downstream tributary (the Moose Jaw River) during particularly wet years, the residence time of water in BPL fluctuates from approximately 6 to 36 months.[20,49]

The gross catchment area for BPL is 3310 km^2; however, due to the complex nature of Prairie hydrology and variable, intermittent precipitation, the effective drainage area (or the area that contributes to flow in an average runoff year) is only 38% of the gross catchment area.[18] Additionally, due to this variable, intermittent precipitation, input from the catchment is transient and akin to overland flows, occasional backflows from the downstream tributary (Moose Jaw River), and flows from upstream catchment tributaries that are indirectly connected to BPL *via* the Qu'Appelle River. Rather than being dominated by catchment sources, flow into BPL is often dominated by the mesotrophic (lower nutrient) supply reservoir, Lake Diefenbaker (ref. 46; Fig. S1–S3, SI), which receives most of its water from rain and snow melt in the rocky mountains of Alberta.[50] Water is released from Lake Diefenbaker through the Qu'Appelle River Dam—these releases are managed by the water security agency[46] for the purpose of maintaining adequate

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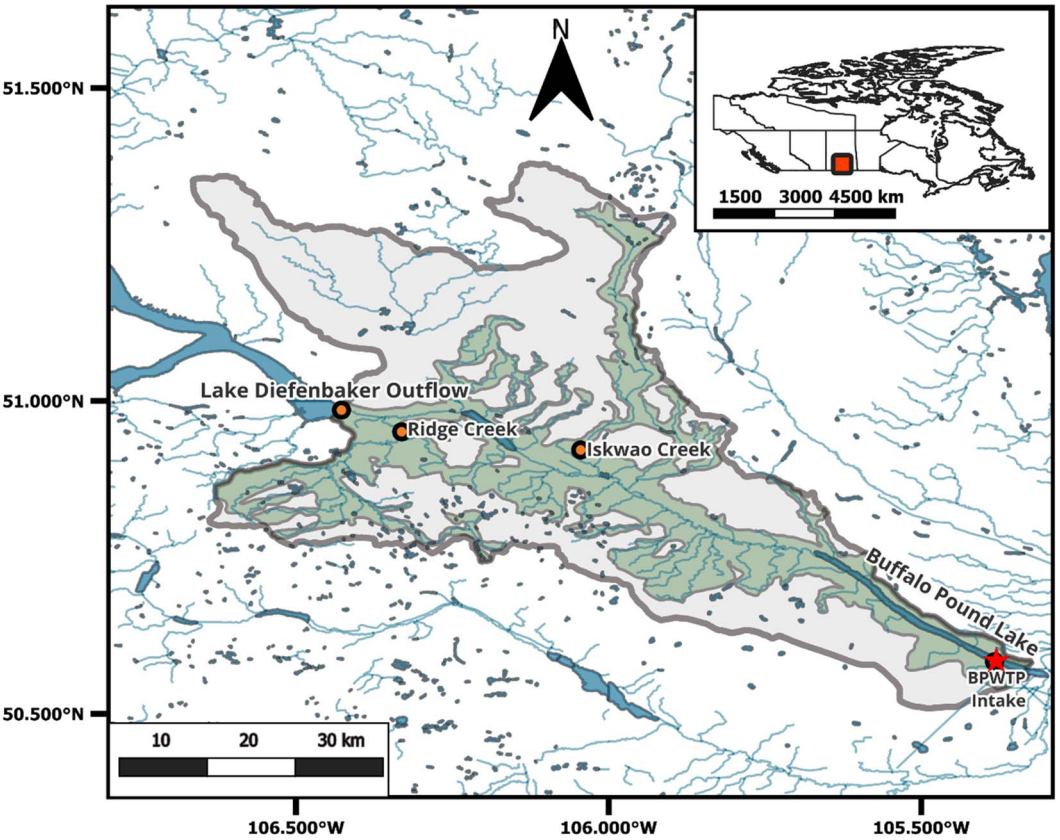

Fig. 1 Map of Buffalo Pound Lake in Saskatchewan, Canada and its gross (gray) and effective (green) catchment area. Map includes Lake Diefenbaker, the hydrometric station measuring daily average flow from Lake Diefenbaker (Lake Diefenbaker Outflow), the two primary catchment tributaries (Ridge Creek and Iskwao Creek), as well as the location of the intake pipes for the Buffalo Pound Water Treatment Plant (red star). The inset map with the red square shows the location of Buffalo Pound Lake within Canada. Geospatial data were downloaded from CanVec and catchment area shapefiles were provided by Prairie Farm Rehabilitation Administration. Bathymetry of Buffalo Pound Lake is available in ref. 20.

water supply for drinking and industrial uses, as well as to control downstream flooding.[17]

Because water from Lake Diefenbaker has been diverted into BPL through the Upper Qu'Appelle River since 1963,[22,51] BPL and Lake Diefenbaker do not share natural catchment boundaries—rather, they belong to the Assiniboine-Red River and the South Saskatchewan River drainage basins, respectively.[17,18] Therefore, the chemistry of water from Lake Diefenbaker differs from catchment inflows to Buffalo Pound Lake,[17,18] with flows from Lake Diefenbaker typically being lower in nutrients compared to runoff generated in BPL's catchment (Fig. S4). BPL's catchment is dominated by agricultural cropland, and both surface runoff and riverine inputs exhibit higher concentrations of nutrients than are found in Lake Diefenbaker.[17,18,20] Water released from Lake Diefenbaker travels nearly 100 km through the Qu'Appelle River system.[17,20] Given the distance between flows from Lake Diefenbaker, catchment tributaries, and BPL, as well as the difference in natural catchment boundaries between BPL and Lake Diefenbaker, water chemistry and flow rates change as it travels downstream towards BPL (ref. 47 and Fig. S4).

2.2. Variables and data

Here, we use regular (typically bi-weekly) observations across a 39 year dataset (1984–2022) provided by the Buffalo Pound Water Treatment Plant (BPWTP) to assess long-term and seasonal variability in phytoplankton biomass. From this dataset, we included concentrations of chlorophyll *a* (an indicator of phytoplankton biomass) and nutrients (soluble reactive phosphorus and dissolved inorganic nitrogen) in analyses. Lake water is brought into BPWTP through two intake pipes (Fig. 1), each one metre above the bottom of the lake and approximately 170 m apart,[53] where water is then analyzed prior to treatment and distribution. The BPWTP laboratory has an internal quality control program and is accredited by the Canadian Association for Laboratory Accreditation to ISO/IEC 17025 standard.[54] Briefly, dissolved inorganic nitrogen (DIN) concentrations are represented as the sum of nitrate and ammonium concentrations. Nitrate concentrations are determined using ion chromatography using a Dionex ICS-1100 Ion Chromatograph, and ammonium concentrations are determined using colorimetry *via* buffering in boric acid, distillation, and then addition of Nessler's reagent, each measured as concentrations of N.[18]

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Concentrations of soluble reactive phosphorus (SRP) were measured using spectrophotometry and the ammonium molybdate method. Chl *a* was measured by spectrophotometry after extracting pigments with acetone. The presence of pheophytin was not corrected for in this analysis.[18]

The sampling frequency for these water quality parameters (SRP, DIN, and Chl *a*) changed throughout the long timescale of this dataset. Typically, water quality parameters were measured bi-weekly or monthly, with an average of 32 weeks with observations each year. In most years, Chl *a* was analyzed at least once a month (except March 2002, January and February 2003, December 2007, December 2017, and August and September 2022). Weeks that did not have Chl *a* data were excluded from analysis, for a total of 1084 observations used in analysis. Where nutrient concentrations of zero were recorded, the zeros were replaced with half the limit of detection for each analysis method, with the limits of detection being 86 μg/L N for ammonia, 57 μg/L N for nitrate, and SRP = 3.0 μg/L.

To assess the effect of large-scale climate oscillations on phytoplankton biomass, two indices were used: the Southern Oscillation Index (SOI) which measures ENSO phases, as well as the PDO index. Index values for the SOI and PDO were downloaded from the National Centers for Environmental Information's website.[55,56] The SOI and PDO values were included as three-month averages of index values to reduce noise and capture broader climate trends. Average annual SOI and PDO index values are shown in Fig. S5.

Additionally, to investigate the effect of hydrological management, the effect of daily average flows from Lake Diefenbaker on Chl *a* were assessed. Daily average flows from Lake Diefenbaker were measured at a hydrometric station (Water Survey of Canada station 05JG006) located at Highway 19 downstream of the Qu'Appelle Dam.[57] Catchment flows are highly zero-inflated given highly intermittent precipitation in the region leading to intermittency of streamflow, thus are not included in formal analysis. See the SI for a summary of flow sources to BPL, including a comparison of water chemistry across these sources, comparisons of flow rates between Lake Diefenbaker and the catchment, and the proportion of high catchment flows (75th percentile of all observations) observed within each decade.

2.3. Model development

Generalized additive mixed models (GAMMs) were used to understand long-term and seasonal changes in Chl *a* concentrations, and generalized additive models (GAMs) were used to assess the predictors of this change, all fitted using restricted maximum likelihood method (REML). Both GAMMs and GAMs are a type of data-driven regression, particularly useful for modelling non-linear relationships between response and predictor variables, and are well-described elsewhere (*e.g.*, ref. 51–53). GAMs assume independent and identically distributed errors, whereas GAMMs relax this assumption by incorporating random effects and correlation structures to explicitly model residual autocorrelation.[58,60] Both models use smooth terms, also called splines, which are constructed from multiple basis functions (k) that can take on different shapes. This flexibility allows them to model various types of non-linear relationships.[58] The models report the effective degrees of freedom (EDF), where higher EDF values suggest a stronger influence of the predictor on the response variable.[60] They also report the reference degrees of freedom (Ref. DF), which corresponds to the basis dimension ($k - 1$) for each predictor. Significance of smooth terms was assessed using approximate tests ($p < 0.05$) based on comparisons of the EDF and Ref. DF.

For the time series analysis, a GAMM was specified using a cubic regression spline for year to capture smooth, long-term nonlinear trends, and a cubic cyclical spline for day of year (DOY) to capture the continuous seasonal cycle. A tensor product smooth between year and DOY was included to investigate whether the seasonal Chl *a* patterns varied over time.

$$y = \beta_0 + f_1\,(\mathrm{DOY}) + f_2\,(\mathrm{year}) + g_1\,(\mathrm{DOY}, \mathrm{year}) + Z_i b_i + \varepsilon_{ij} \quad (1)$$

where y corresponds to Chl *a*, β_0 is the intercept, f_1 (DOY) and f_2 (year) are the smooth functions for year and DOY, h_2 (DOY, year) is a tensor product interacting year and DOY, $Z_i b_i$ are random effects, and ε_{ij} is the error term that incorporates autoregressive (AR) correlation structures.[60] Comparison of model fit using eqn (1) resulted in an AR(1) structure being used in the GAMM.[61] Chl *a* exhibited a positive skew; therefore, a Gamma distribution was specified for eqn (1). Periods of significant change in average Chl *a* concentrations were identified by computing the first derivatives of the smooths for year and for DOY, and identifying instances where confidence intervals for these derivatives did not include zero, indicating that the trend was significantly changing.[17,58]

Secondly, we specified a GAM that assessed the smooth effects of dissolved nutrients and interbasin transfers, all with thin plate regression splines, as they provide flexible and efficient smooths without requiring knot placement to be pre-specified.[62] Additionally, tensor products were used to capture the main and interactive effects of climate oscillations (SOI and PDO) and time (year and DOY) on Chl *a* concentrations. Due to the positively skewed data, a Tweedie distribution ($p = 1.99$) was used.

$$\begin{aligned} y = \beta_0 &+ h_1\,(\mathrm{SRP}) + h_2\,(\mathrm{DIN}) + h_3\,(\mathrm{QLD}) \\ &+ i_1\,(\mathrm{SOI}, \mathrm{PDO}) + i_2\,(\mathrm{DOY}, \mathrm{year}) + \varepsilon \end{aligned} \quad (2)$$

where y corresponds to Chl *a*, β_0 is the intercept, h_1 (SRP), h_2 (DIN), and h_3 (QLD) are the smooth functions for SRP, DIN, and flow from Lake Diefenbaker (QLD), i_1 (SOI, PDO) and i_2 (DOY, year) are tensor products capturing the main and interacted SOI and PDO and year and DOY, and ε is the error term. By including seasonal dynamics in eqn (2), predictors were required to explain variation outside typical seasonal patterns to be identified as having a significant effect on Chl *a* concentrations.[63] The size of effect that each predictor had on Chl *a* concentrations was quantified as the range between the minimum and maximum GAM-simulated Chl *a* concentrations in response to specific predictors.

Using eqn (2), we developed an additional model with a 6 month lag applied to the 3 month averages of SOI and PDO

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Table 1 Summary of chlorophyll *a* time series GAMM showing estimated effects of year and day of year, and the tensor product interacting year and day of year[a]

Predictor	EDF	Ref. DF	*p*-Value
Year	5.8	5.8	<0.0001
Day of year	8.4	10	<0.0001
ti(DOY, year)	6.2	12	<0.0001

[a] Note: EDF = estimated degrees of freedom; Ref. DF = reference degrees of freedom; model adjusted $R^2 = 0.4$; $n = 1206$.

index values to account for potential lagged effects of ENSO and PDO. Outputs from the models with and without the 6 month lag were compared, showing negligible differences in model fit and outputs (Table S1 and Fig. S6, SI). Therefore, outputs from the model without the 6 month lag are presented here.

Model fit was assessed using adjusted R^2 values and visual inspection (Fig. S7 and S8). All data analyses were performed using *R* statistical software version 4.4.2 using the *mgcv R* package version 1.9-1[64] and *gratia R* package version 0.9.0.9001.[65]

3. Results

3.1. Time series analysis

Time series analysis using GAMM showed that the fitted smooth terms for day of year (DOY) and year, as well as the tensor product interacting year and DOY, were all significant (*p*-value < 0.0001; Table 1).

Chl *a* concentrations increased at the beginning of the time series (Fig. 2A) from approximately 1984 until 2000, after which variation is less pronounced. The fitted cubic regression spline for year in Fig. 2 (A) shows that average Chl *a* concentrations

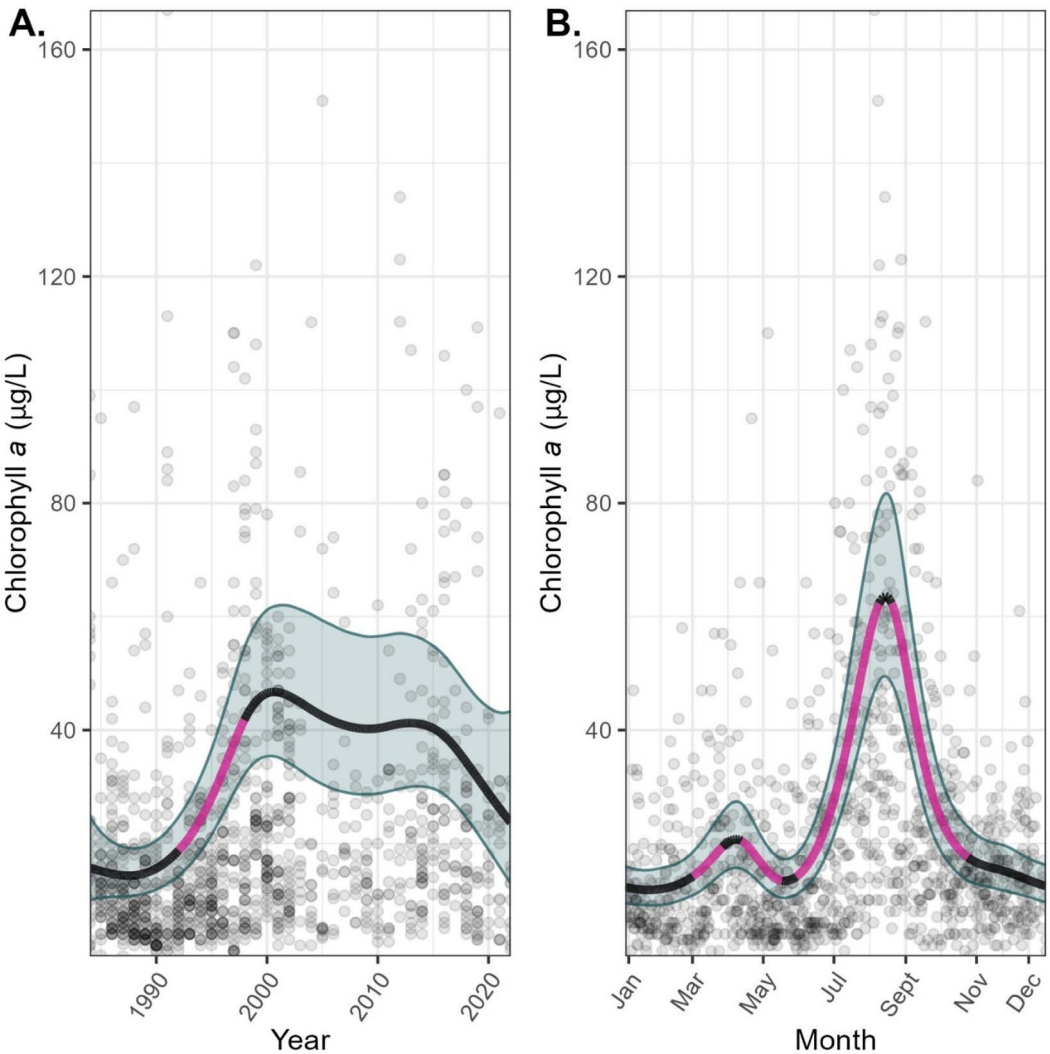

Fig. 2 Time series analysis showing long-term (A) and seasonal (B) trends in chlorophyll *a* concentrations (µg/L) in Buffalo Pound Lake across 1984–2022. The black lines represent the GAMM-estimated trends, while the points represent observed Chl *a* concentrations. The blue-grey ribbons represent the 95% confidence intervals of GAM-estimated trends. The pink highlights in both panels represent periods of significant change in chlorophyll *a* concentrations.

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significantly increased from 1992 through 1997 (represented as the pink highlighted section of the spline) with this trend peaking around the years 2000–2002. The highest Chl *a* concentrations were measured in August of 1991 (167 μg/L), 2005 (151 μg/L), and 2012 (134 μg/L). The period of significant increase was followed by a period where average Chl *a* concentrations slightly decreased (∼2003–2008), and then slightly increased again (∼2013–2015; note observations with high concentrations of Chl *a* in 2013–2019). The cubic cyclical spline for DOY in Fig. 2 (B) shows a clear seasonal pattern with a spring bloom typically beginning in early March and peaking in mid-April, followed by a larger, longer-lasting summer bloom starting in early June and peaking in mid-August.

Between 1984–1990, spring blooms were larger and started earlier in the year (January–early April), while summer bloom peaks were smaller (Fig. 3). Beginning around 1995, Chl *a* concentrations were lower during winter and spring, and higher in summer, with the largest summer peaks occurring in the late '90s and early 2000s. Near the end of the time series (2010 onwards), this trend shifts to greater Chl *a* concentrations in late fall and winter (late October to late January) and much smaller spring peaks in biomass. Note that BPL is typically ice-covered from November to late April/early May, with the duration of ice cover ranging from 124–184 days.[53]

3.2. Predictors of chlorophyll *a*

Generalized additive modelling showed that all included environmental variables were significant predictors of variability in Chl *a* concentrations (p-values < 0.0001; Table 2). The model explained 60% of deviance in Chl *a* concentrations over the 39-year dataset.

GAMs also showed that nutrients exhibited bimodal relationships with Chl *a* (Fig. 4). At concentrations below ∼37.5 μg/L, SRP had a negative relationship with Chl *a* (Fig. 4A), above which the relationship changed to positive, exhibiting a peak around 125 μg/L, though the number of observations of concentrations greater than 100 μg/L SRP is low (see the rug

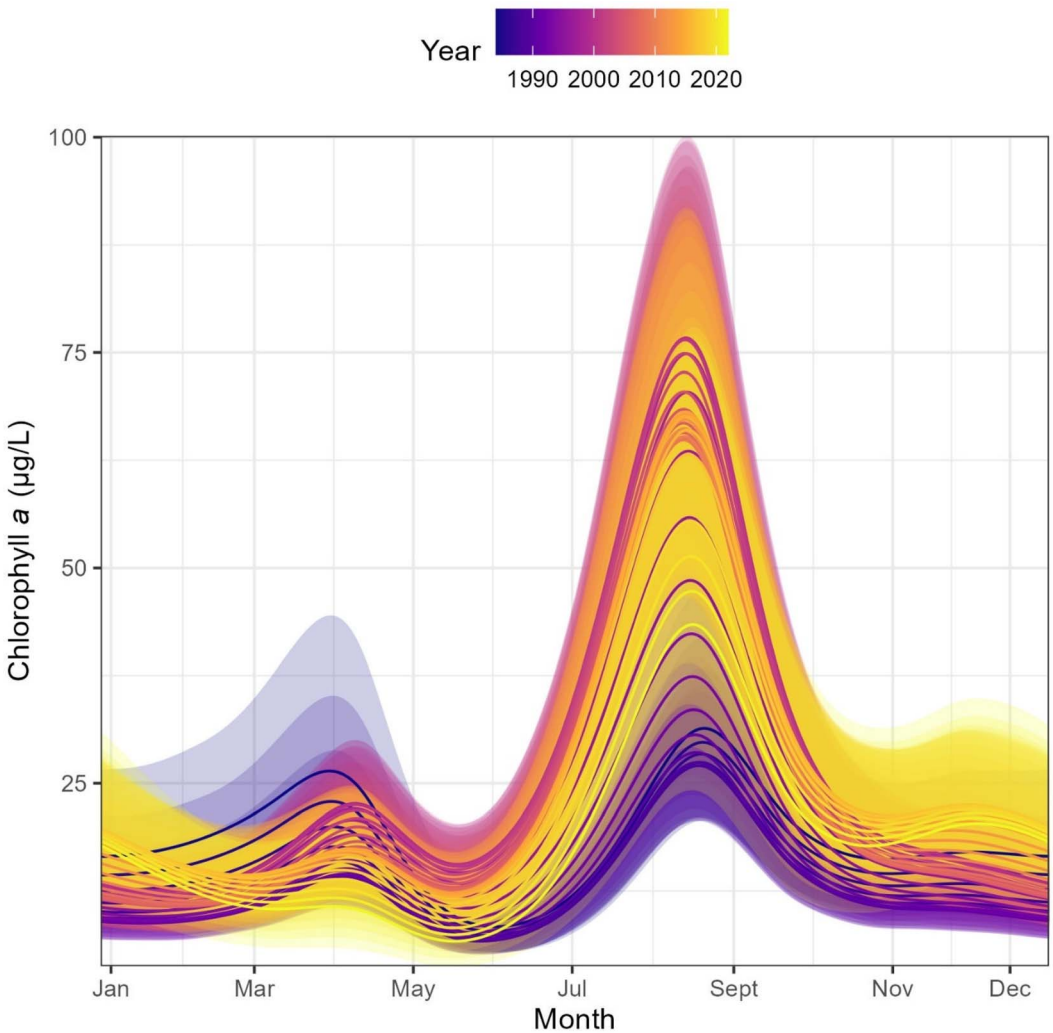

Fig. 3 Time series analysis showing interannual variability in chlorophyll *a* concentrations (μg/L) in Buffalo Pound Lake across 1984–2022. Each line represents the estimated seasonal trend for a given year, with darker purples representing earlier years and lighter yellows representing more recent years in the time series. Shaded ribbons for each line represent 95% confidence intervals.

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Table 2 Summary of additive modelling of the estimated effect of environmental predictors on chlorophyll *a* concentrations[a]

Predictor	EDF	DF	*p*-Value
Soluble reactive phosphorus	5.3	9	<0.0001
Dissolved inorganic nitrogen	6.2	9	<0.0001
Lake Diefenbaker flow	0.96	6	<0.0001
te(SOI, PDO)	12.1	24	<0.0001
te(Year, DOY)	64.9	109	<0.0001

[a] Note: EDF = estimated degrees of freedom; DF = degrees of freedom; adjusted $R^2 = 0.501$; $n = 1084$; deviance explained = 60%.

along the *x*-axis). DIN had a positive effect on Chl *a* at low concentrations (up to ~0.25 mg/L DIN), after which the relationship becomes negative, before turning positive and exhibiting a second peak between 0.75 and 1.25 mg/L (Fig. 4B). Similar to SRP, there were fewer observations of concentrations of DIN greater than 1.0 mg/L, resulting in larger uncertainty bands above this concentration. In contrast to nutrients, flow from Lake Diefenbaker (QLD) showed a linear, negative effect on Chl *a* (Fig. 4C). Note that QLD is negatively related to flow from the watershed, so when watershed flows are higher, QLD is lower, and *vice versa* (Fig. S1 and S2).

The 3 month averages of SOI and PDO index values showed interactive effects on Chl *a* concentrations (Fig. 5). Most notably, Chl *a* concentrations were higher at negative SOI values (less than −2) and positive PDO index values (between 0 and 2), both which are typical of warmer, drier phases, with PDO interacting with ENSO to intensify El Niño events. Additionally, the interactive effects of SOI and PDO showed lower Chl *a* concentrations at negative SOI values (between 0 and −2) and negative PDO values (between −1 and −2), as well as at positive

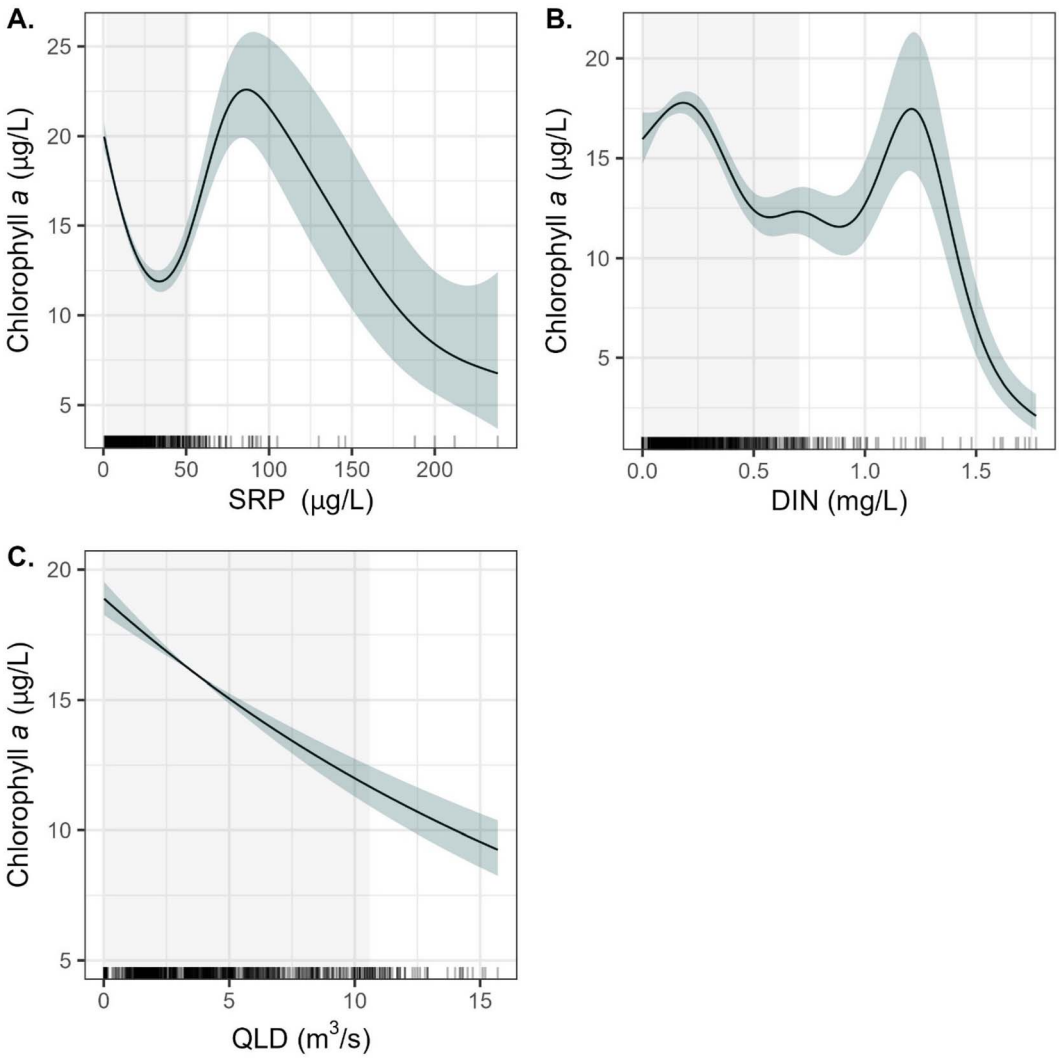

Fig. 4 Chlorophyll *a* responses (in µg/L) to smooth effects of predictors including soluble reactive phosphorus (SRP, µg/L Panel A), dissolved inorganic nitrogen (DIN, mg/L, Panel B), and flows from Lake Diefenbaker (QLD, m^3/s, Panel C), all estimated using generalized additive modelling. The rug along the *x*-axis indicates data points. The light grey underlay indicates the middle 90% of all observations and the blue-grey ribbons represent 95% confidence intervals. Interannual variability in relationships (specified as a tensor product of year and DOY) have also been accounted in this model but are not shown here. Plots generated using code adapted from Wiik *et al.*[66]

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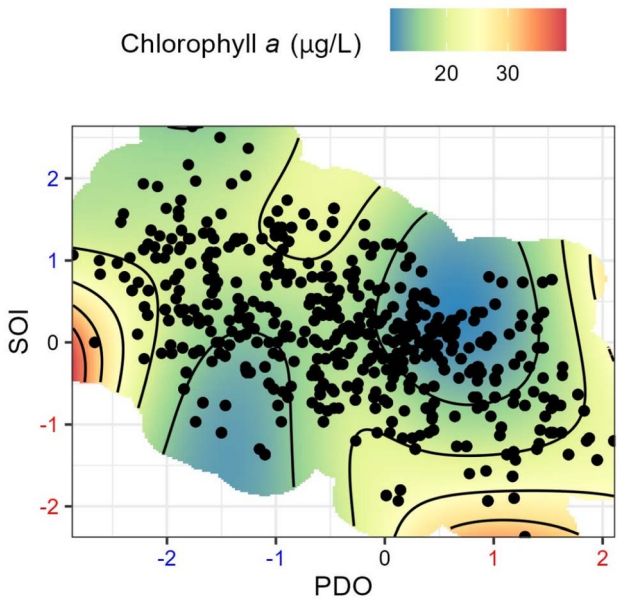

Fig. 5 Fitted chlorophyll *a* estimates (μg/L) predicted using additive modelling of a tensor product interaction between the 3 month average of SOI and PDO index values. Lower ends of the colour spectrum (blues and greens) represent lower modelled chlorophyll *a* concentrations whereas higher ends (yellow, orange, and red) indicate higher modelled chlorophyll *a* concentrations. Axis labels are colour-coded according to the phase they represent, with blue representing cooler and wetter phases, and red representing warmer and drier phases. Zero is neutral.

PDO values (between 0 and 1.5) and positive SOI values (between 0 and 1), which are typical of wetter, cooler periods and La Nina events.

Each predictor had varying influence over Chl *a* concentrations (Table 3). The predictor with the strongest effects on Chl *a* was the tensor product interacting SOI and PDO, followed by SRP, DIN, then QLD (Table 3). A timeline showing the median, maximum, and minimum values for environmental variables with ENSO and PDO phases overlain is provided in Fig. S9. Temporal trends in water temperature are presented in Fig. S10 and S11.

Table 3 Size of effects that predictors had on Chl *a* concentrations (μg/L) according to GAM simulations, with 95% confidence intervals in brackets

Predictor	Predictor effect size (all in μg/L) (95% confidence interval)[a]
SRP	15.8 (7.4, 22.0)
DIN	15.7 (14.1, 17.0)
QLD	9.6 (7.9, 11.3)
SOI × PDO	28.6 (19.7, 39.8)

[a] Range of predictor effect is the difference between the minimum and maximum GAM-simulated Chl *a* concentration in response to specific predictors.

4. Discussion

Effectively managing drinking water sources depends on understanding how the development of algal blooms responds to climatic variability, hydrologic management, and nutrients, helping to anticipate future change and inform adaptation options. This research reveals a significant increase in phytoplankton biomass (as measured by Chl *a* concentrations) from 1992 through 1997 in an important drinking water reservoir in Saskatchewan, Canada (BPL). Our analyses indicate that climate variability, hydrological management, and nutrients play a significant role in influencing phytoplankton biomass, with large-scale atmospheric teleconnections, namely ENSO and PDO, showing the strongest influence on phytoplankton biomass. Specifically, ENSO and PDO index values that indicate the warm/dry phase, when El Niño events co-occurred with the warm PDO phase, showed higher Chl *a* concentrations. These relationships are complex and interlinked, with ENSO-PDO interactions affecting flow source and dissolved nutrient concentrations. The combined effects of these relationships regulate water quality, with dissolved nutrients increasing phytoplankton biomass and flows from supply reservoirs reducing biomass.

To our knowledge, this is the first study to quantify long-term, significant increases in phytoplankton biomass and to attribute them to modes of climate variability and associated changes in flow source and nutrient concentrations in a drinking water reservoir. Past work has primarily relied on shorter-term analyses of variability in climate and flow source impacts, also showing increases in nutrient and Chl *a* concentrations in other hydrologically-managed reservoirs.[67,68] Indeed, there is growing evidence of the impacts of climate variability and the value of climate indices in informing changes in water chemistry in diverse ecosystems, with insights spanning from impacts on seasonal bloom severity,[28,69] as well as the intensity of spring blooms in marine[70,71] and freshwater systems.[72] Trophic status has increased globally[1,3,4,10] and will continue to do so under the 'business-as-usual' scenario,[73] underscoring the importance of understanding these relationships, climatic influences, and management options to handle these interacting stressors. Through using indices of large-scale climate oscillations and a flexible modelling tool, the approach employed in this work is highly adaptable for understanding broad risks of climatic variability to water quality in diverse ecosystems.

4.1. Influence of climate on nutrients and phytoplankton biomass

The warm and calm conditions produced by strong El Niño events have important implications for phytoplankton biomass that can create challenging and potentially disastrous consequences for water treatment if not effectively managed. These effects can be particularly severe when these warm, calm conditions are preceded by higher volumes of nutrient-rich inflows. For example, after a severe drought on the Canadian Prairies in the 1980s,[74] the 1990s were a particularly wet period

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with higher catchment flows, instances of spring flooding, and lower flows from the mesotrophic supply reservoir (Lake Diefenbaker).[22,31,75] Two El Niño events occurred—one in 1991 and another stronger event that was in-phase with PDO in 1997. The combination of higher catchment runoff and two El Niño events in the 1990s contributed to the highest average water temperature, highest nutrient concentrations, and highest Chl *a* concentration across the entire timeseries (Figures S9–S11). Concerningly, in BPL and another shallow lake in this region (Wascana), the abundance of colonial cyanobacteria peaked during the strong El Niño event in 1997.[69] Similar events recur through the timeseries: during a strong El Niño in 2015, amongst another wet period (2010–2016), an early peak in phytoplankton biomass, elevated particulates in the water column, and prolonged thermal stratification led to issues at the water treatment plant, resulting in a drinking water shortage for the first time in BPWTP's operating history.[17,49]

Periods with greater precipitation and nutrient export, especially if followed by warm, stable conditions, are particularly problematic for water treatment because these conditions are conducive to the growth of phytoplankton, especially cyanobacteria.[76,77] Rainfall can increase nutrient export to lakes—especially if following a prolonged drought, which can result in large pulses of nutrients into lake systems.[78] When combined with (or followed by) warm, stable conditions that can contribute to thermal stratification, cyanobacteria can thrive—in part due to their ability to regulate buoyancy.[77,79] Additionally, because El Niño events are associated with warm and calm conditions, they can contribute to enhanced internal P loading.[47,48] These warm/dry conditions and increased internal P loading have been shown to contribute to blooms of cyanobacteria in other reservoirs.[80] Our observation that these conditions that occurred in the '90s contributed to a significant increase in phytoplankton biomass at this time is further supported by BPL's sediment record, which suggests a doubling of Chl *a* between 1990 and 1996[43] and associated increases in total and diazotrophic cyanobacteria during the late 1990s.[22] This highlights important ecological change at this time.

Intriguingly, similar changes were underway around this timeframe elsewhere in the region and beyond, with evidence of similar drivers. For example, spring flooding was an important trigger for increased phytoplankton biomass and dominance of cyanobacteria in Lake Winnipeg.[81] Estimates suggest that without the increased frequency of spring flooding, intensified urbanization and agricultural land use would have only contributed one third of the increased nutrient loading between 1970 and the mid-1990s in Lake Winnipeg.[81] Similar observations have been made in Lake Erie, with increases in phytoplankton biomass since the mid-1990s,[82] suggesting that these conditions in the 1990s were an important period of change in many Canadian lakes. Indeed, recent evidence has shown increases in phytoplankton biomass in Canadian lakes since the mid-1960s.[83] Yet these observations are not unique to Canada—global increases in the intensity of summer blooms have occurred since the 1980s, with lakes in Thailand, China, Russia, and the U.S. showing increases in bloom intensity since the 1990s, often coinciding with changes in temperature, precipitation, and/or fertilizer use.[4] These examples highlighting the 1990s as an important period of change for phytoplankton suggest that the findings presented here may also apply to other temperate lakes, especially shallow lakes and reservoirs in regions with substantial agriculture in their catchments.

While the overarching increase in Chl *a* in BPL was primarily driven by summer trends, there was a second important change—a decrease in spring phytoplankton biomass over time and recent (~2010 onwards) increases in fall biomass. This decrease in spring biomass is concerning given importance of spring blooms to aquatic food webs[84] and potential biogeochemical implications for seasonal nutrient cycling.[85,86] Variation in fall and spring biomass may reflect changes in winter conditions associated with climate change, with periods of ice cover and snow coverage on ice declining.[87] Shortening periods of ice cover can result in higher productivity in the fall and winter.[88] Combined with reduced snow coverage, this can contribute to earlier peaks in spring biomass, a prolonged mixing period, and earlier depletion of nutrients.[72,88] Furthermore, because El Niño is associated with warmer winters, reduced snowpack, and slower wind speeds that can last several months or longer on the Canadian Prairies,[29,32] these events could contribute to reduced mixing and nutrient availability—especially when reinforced by the warm PDO phase—potentially reducing spring phytoplankton biomass in freshwater systems, as has been observed in marine systems.[70,71] Due to safety issues (*i.e.*, unsafe ice), sampling during the late winter period, when the spring bloom can occur, is often limited. As a result, this dataset is among a small number of multi-decadal under-ice datasets—presenting important and rare insights to changing seasonality in north temperate lakes.

Following the period with the greatest change—when phytoplankton biomass significantly increased—annual average phytoplankton biomass generally remained higher while nutrient concentrations decreased. Nearly all observations with high SRP and DIN—where the relationships between dissolved nutrients and Chl *a* inflect—occurred in the late 1990s. These observations imply a key inflection or transition that was sustained even after the abnormally wet period ended—an effect seen in other shallow lakes (*e.g.*, ref. 69). Evidence that biomass has generally remained higher in BPL while dissolved nutrient concentrations have decreased may indicate a potential regime shift towards a more turbid state, which can be triggered by large perturbations such as floods.[89–91] Concerningly, regime shifts are often difficult and slow to reverse.[90] The recovery of lakes following reductions in external nutrient loading often requires decades or longer—if recovery occurs at all—due to shifting baselines, legacy nutrient stores, and internal P loading.[92–94] However, alternative unstable states have also been observed in this region,[95] leaving questions about the diversity of factors affecting when shifts to high biomass states are sustained. Nonetheless, given the persistence of the shift towards a higher biomass state in this system, careful consideration of the potentially long-lasting impacts of water management decisions is critical.

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Under future climate projections—including increased temperatures, frequency and extent of droughts, and frequency of extreme runoff events[22,96–98]—the conditions that contributed to significant increases in phytoplankton biomass may become more common. Indeed, these conditions are widely anticipated to increase the severity of cyanobacteria blooms.[99–101] For example, increasingly frequent summer heatwaves and increased temperatures, along with slower wind speeds and the associated risk of internal P loading, are expected to promote blooms of cyanobacteria.[101,102] Increasingly frequent extreme precipitation events can increase nutrient-rich runoff events.[103,104] As was observed here, these nutrient pulses can create suitable conditions for the growth of phytoplankton, including cyanobacteria and the production of cyanotoxins, especially if followed by warm, stable conditions.[77] In cold regions, changes may be particularly pronounced. For example, warmer springs, changing ice cover and snow thickness, and associated changes to nutrient loading and photoperiod present potentially large disruptions to seasonality and thus community composition, including decreases in diatoms and increases in the dominance of cyanobacteria.[105,106] Furthermore, extreme El Niño events—which were associated with the significant increase in Chl *a* concentrations observed here—are expected to double in frequency as a result of climate change.[107] Thus, the findings presented here, in combination with these climate change projections, suggest that future conditions may induce important changes to bloom timing and severity, likely shifting towards increasing dominance of cyanobacteria. Changes in the timing and severity of blooms, especially cyanobacteria, have important consequences for water treatment, challenging proactive and robust water treatment,[108,109] and raising questions about the role of water management in adaptation.

Although a variable and changing climate, along with complex hydrology, make it challenging to anticipating peaks in runoff, and thus, peaks in nutrients and phytoplankton biomass,[110] our work suggests that climate oscillations may provide advance warning of bloom risk. Importantly, incorporating water chemistry and climate indices such as ENSO and PDO into predictive models can contribute to earlier and more accurate seasonal predictions of bloom severity.[28] Additionally, ENSO projections have been improving,[27] and the ENSO amplitude and phase can be predicted reasonably well up to a year (amplitude) and two years (phase) in advance.[111] Using these predictions to infer whether a strong El Niño is likely to occur during a period with greater precipitation may help treatment plant operators prepare and proactively respond to potential risks to treatment operations and potential water shortages.

4.2. Flow source and hydrological management influence phytoplankton biomass

Flow source—driven both directly and indirectly by climate—is a key influence on phytoplankton biomass and thus presents an important tool to leverage in hydrologically managed systems. In contrast to the poor water quality conditions observed during wet periods in the 1990s and 2010s—when flows were dominated by agricultural catchments and thus, contributed greater nutrient loads—flow from the mesotrophic supply reservoir, Lake Diefenbaker, had a nearly linear, negative effect on Chl *a* concentrations. This suggests that flow from Lake Diefenbaker had a positive effect on water quality in BPL *via* decreasing phytoplankton biomass, highlighting the importance of interbasin transfers. While changes in hydrodynamic conditions in reservoirs are known to affect bloom development (*e.g.*, ref. 12, 67, 99, and 100), these effects are often linked to increased turbulence, which can disrupt thermal stratification and affect mixing regimes, dispersing nutrients and algal cells and thus reducing bloom formation.[112–115] Given the distance water travels from Lake Diefenbaker (~100 km), as well as the long, narrow nature of BPL, the negative relationship observed between flow from Lake Diefenbaker and Chl *a* concentrations is expected to be primarily a dilution effect—where mesotrophic water from Lake Diefenbaker dilutes nutrient-rich waters in BPL.[20,22] Despite the potential benefits of increasing flows from the mesotrophic supply reservoir, channel capacity presents an important constraint on flow releases from supply reservoirs. For example, the Upper Qu'Appelle River has a maximum capacity of 14 m^3/s during open water season, generally limiting flow releases from Lake Diefenbaker to this rate.[24]

Managing interbasin transfers necessitates consideration of many factors, including risk of erosion at higher flows,[116] lags and chemical change associated with transit time,[24] the chemistry and temperature of flows from the supply reservoir relative to downstream systems,[21,23] managing competing demands for water including irrigation and hydropower production,[117] and even broader considerations such as flooding.[12,118] For example, during wet periods with high catchment flows, flows from the upstream reservoir are reduced or paused to prevent downstream flooding.[12,17,24] Thus, the ability to dilute or flush nutrients during these wet periods is limited, contributing to observed water quality issues and water shortages in the 1990s and in 2015. In contrast, during mid- to late-summer in dryland agricultural regions, blooms are exacerbated by low natural flows, high water demand, and greater potential for thermal stratification and internal nutrient loading.[48,97,119] Although drier conditions, channel capacity, and management priorities such as irrigation and hydropower generation may challenge the ability to increase flow releases from supply reservoirs in summer, strategically timing flow releases during these drier mid- to late-summer periods may provide enough disturbance to sufficiently combat larger, potentially problematic cyanobacterial blooms.[68]

Reservoir management strategies are considered more cost-effective and environmentally-friendly than other interventions (*i.e.*, physical, chemical, and biological interventions) for mitigating algal blooms.[12] If flow management to BPL—and indeed, other shallow, hydrologically managed reservoirs—is refined to balance these complex priorities and minimize trade-offs, leveraging hydrological management regimes (*i.e.*, environmental flows for water quality) could offer an important, cost-effective tool to help manage blooms. Developing appropriate management regimes requires scientific understanding

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of links among climate variability, hydrology, and water quality, as well as collaboration between scientists, water treatment operators, water managers, and other stakeholders and rightsholders to identify opportunities for balancing multiple priorities for water—thus presents an important area for future work.

Clearly, leveraging hydrological management regimes is challenging. It also cannot fully mitigate long-standing water quality challenges in this, and many other, eutrophic water supplies.[22,120] Given the importance of nutrients to phytoplankton growth, which here showed stronger effects on Chl *a* than hydrological management, investing in measures to reduce the export of nutrients from land to lake remains a critical leverage point for protecting source water quality. In highly agricultural regions such as this, broader adoption of beneficial land management practices that are robust to climate change will become increasingly critical as risks to water treatment increase.[69,121,122]

5. Conclusion

Significant inter- and intra-annual variation in phytoplankton biomass occurred over the past 39 years at a shallow, hydrologically managed reservoir in the Canadian Prairies (Buffalo Pound Lake, Saskatchewan). Climate, together with linked impacts on hydrology, hydrological management, and nutrient concentrations, were important predictors of change in phytoplankton biomass, measured as Chl *a* concentrations. A period with greater precipitation (and associated nutrient-rich inflows from an agriculturally dominated catchment), interspersed with cycles of warm temperatures and calmer winds associated with strong El Niño events,[30–32,123] contributed to high nutrient concentrations and a significant increase in phytoplankton biomass in the 1990s, as well as a water shortage at the drinking water treatment plant in 2015. This period of significant increase was marked by increased summer bloom intensity and a shift towards increased cyanobacterial dominance, with a persistent high biomass state remaining for many years after initial increases—mirroring observations in lakes across the world during this same time period.[4,22,82,83] In addition to shifting towards larger summer blooms, changes in seasonal bloom dynamics included shifts to smaller blooms in spring and, in recent years, higher biomass in late fall and winter. These shifts may have important ecosystem-level consequences given importance of bloom timing to food webs and seasonal biogeochemical nutrient cycling.[84–86] Combined, these results indicate a dynamic balance between climatic and management controls on nutrient status and phytoplankton. They also raise concerns about shifts to more turbid regimes impacting ecosystem health and human uses, including water treatment.

This research presents a cautionary tale, where wet periods with high catchment flows combined with warm, stable conditions, along with lower flows from a supply reservoir with a diluting function, appear to have contributed to deterioration of water quality in a key water source. Our work suggests that climate oscillations may provide advance warning of bloom risk and is among the first to link long-term, significant increases in phytoplankton biomass to modes of climate variability and associated changes in flow source and nutrient concentrations in shallow drinking water reservoir. Although intervention options are limited and uncertain in BPL, interbasin transfers have had clear benefits in terms of reducing phytoplankton biomass. Therefore, environmental flow management for water quality may be merited. However, as is common in managed water systems,[124–126] this is a complex, social-ecological system. Management change may not be feasible due to limitations of channel capacity, and myriad (sometimes changing) priorities including irrigation and hydropower. Ultimately, reducing nutrient inputs into reservoirs will be critical for protecting water quality. In governing a system with complex trade-offs in water management decisions and water insecurity, deepening our understanding of temporal change, trade-offs, and the role of climatic variability can help support adaptive decisions, aiding water management and delivery of reliable, safe, and affordable drinking water. Because the approach used here to assess climate impacts on water quality is adaptable, it can contribute to deepening these understandings in freshwater systems globally.

Author contributions

Danielle S. Spence (University of Saskatchewan): conceptualization, methodology, formal analysis, writing – original draft preparation, writing – review and editing. Kristin J. Painter (University of Saskatchewan): conceptualization, methodology, writing – review and editing. Ali Nazemi (Concordia University): data curation, writing – review and editing. Jason J. Venkiteswaran (Wilfrid Laurier University): conceptualization, writing – review and editing. Helen M. Baulch (University of Saskatchewan): conceptualization, writing – review and editing, funding acquisition.

Conflicts of interest

Helen M. Baulch has been funded by the Buffalo Pound Water Treatment Corporation who uses water from this system in drinking water supply, and by the Saskatchewan Water Security Agency, which is responsible for water management and decisions, including agricultural drainage. There are no other conflicts to declare.

Data availability

Flow and water chemistry data for Lake Diefenbaker and Buffalo Pound Lake's catchment can be requested through the Saskatchewan Water Security Agency (**https://www.wsask.ca/contact-us/**). Water chemistry data are not publicly available but can be requested for research purposes, subject to agreements, through the Buffalo Pound Water Treatment Plant (Blair Kardash, blairk@buffalopoundwtp.ca). *R* code for all statistical analyses and figures are available here: **https://github.com/daniellespence/bpl_chla**.

Supplementary information is available. See DOI: **https://doi.org/10.1039/d5va00094g**.

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Acknowledgements

This study is supported by FORMBLOOM, a project of Global Water Futures funded by the Canada First Research Excellence Fund, NSERC Discovery Grant to HMB, and scholarship funding to DSS from the School of Environment and Sustainability. We are grateful to Blair Kardash, Laboratory Manager at the Buffalo Pound Water Treatment Plant, for stewardship of treatment plant data and supporting this research, as well as the Buffalo Pound Water Treatment Plant for chemical analyses. Furthermore, we are grateful to John-Mark Davies and Heather Sauer with the Water Security Agency, as well as Blair Kardash for reviewing and providing helpful feedback that greatly improved this manuscript. We are grateful to reviewers for the constructive comments that improved this manuscript.

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