

Blooms and flows: Effects of variable hydrology and management on reservoir water quality

Kristin J. Painter^{1*}, Jason J. Venkiteswaran², and Helen M. Baulch¹

¹ School of Environment and Sustainability, Global Institute for Water Security, University of Saskatchewan, Saskatoon, SK S7N 5C8, Canada

² Department of Geography and Environmental Studies, Wilfrid Laurier University, Waterloo,

ON N2L 3C5, Canada

*Corresponding author: kristin.painter@usask.ca

ABSTRACT

Flow management has the potential to significantly affect water quality. Shallow lakes in arid regions are especially susceptible to flow management changes which can have important implications for the formation of cyanobacterial blooms. Here, we reveal water quality shifts across a gradient of managed source water inflow regimes. Using *in situ* monitoring data, we studied a seven-year time span during which inflows to a shallow, eutrophic drinking water reservoir transitioned from primarily natural landscape runoff (2014 to 2015) to managed flows from a larger upstream reservoir (Lake Diefenbaker; 2016 to 2020) and identified significant changes in cyanobacteria (as phycocyanin) using generalized additive models to classify cyanobacterial bloom formation. We then connected changes in water source with shifts in chemistry and the occurrence of cyanobacterial blooms using principal components analysis. Phycocyanin was greater in years with managed reservoir inflow from mesotrophic Lake Diefenbaker (2016 to 2020) but dissolved organic matter (DOM) and specific conductivity, important determinants of drinking water quality, were greatest in years when landscape runoff dominated lake water source (2014 to 2015). Most notably, despite changing rapidly, it took multiple years for lake water to return to a consistent and reduced level of DOM after managed inflows from upstream Lake Diefenbaker were resumed, an observation that underscores how resilience may be hindered by weak resistance to change and slow recovery. Environmental flows for water quality are rarely defined yet here it appears trade-offs exist between poor water quality via elevated conductivity and DOM, and higher bloom risk. Taken together, our findings have important implications for water managers who must protect water quality while adapting to projected hydroclimatic change.

Keywords: cyanobacterial blooms; dissolved organic matter; flow management; resilience; water quality; environmental flows

OPEN DATA STATEMENT

The data, figures and associated code used to describe cyanobacterial blooms and water quality in Buffalo Pound Lake are available at the following URL: https://github.com/biogeochem/buffalopound_interannual

INTRODUCTION

The widely celebrated “Winning the Prairie Gamble” exhibit unveiled for Saskatchewan’s centenary in 2005 tells the story of a province that would succeed against hardship to become Canada’s agricultural powerhouse (Saskatchewan, 2002; B. Waiser, 2006). More than 40% of Canada’s cultivated land is located in Saskatchewan and the province is the world’s largest exporter of globally important food crops including canola, lentils, flaxseed, oats, and durum wheat (Saskatchewan, 2022). Today, the people of Saskatchewan and the prairies at large are faced with a new prairie gamble – mitigating and adapting to the looming and uncertain impact of a changing climate and increasingly uncertain water security.

Climatic changes are anticipated to reduce or alter water supply and quality across the prairies in the coming decades (Schindler and Donahue, 2006; Wheater and Gober, 2013; Clarke et al., 2015). As a result, Saskatchewan is now betting \$4 billion on the most expensive project in the province’s history, increasing reservoir and irrigation capacity (Lake Diefenbaker Irrigation Projects, 2021), in hopes of enhancing economic resiliency by doubling the province’s irrigable land (Saskatchewan, 2020). However, like the original prairie gamble, not all Saskatchewan stories about building agricultural capacity have been success stories (Stonechild, 2006; B. Waiser, 2006), and scientists, environmental groups, and local First Nations have cautioned that these projects could have unintended consequences (Anderson, 2022). As anthropogenic pressures on critical water resources increase, there is a dire need to fill knowledge gaps and reduce uncertainty around the environmental and hydrologic conditions that affect source water quality and resilience, particularly in water-scarce regions such as this, where supply and quality stressors are already substantive.

Surface waters are heavily managed on the prairies, typically to enhance agricultural production (e.g., via drainage and irrigation) and manage flood risk. Larger lakes and rivers are often impounded to create reservoirs that supply drinking water, industry, and hydropower. Such intensive management often results in tightly controlled water levels and flow regimes to maintain the most critical water supplies. Human intervention has reduced intermittency and aided the creation of stable flow regimes in these systems; however, the same cannot be said for water quality. Water quality issues are especially challenging in prairie systems where soils are naturally phosphorus (P) rich and agriculture dominates watersheds (Schindler and Donahue, 2006; Schindler et al., 2012). Prairie lakes and reservoirs are often bloom-prone with frequent mid-summer blooms of harmful cyanobacteria (Pick, 2016; Hayes et al., 2020; Painter et al., 2022). Yet, water quality issues remain understudied and flow management for water quality is rarely prioritized.

Both intra- and interannual variability are important drivers of water quality. Many prairie lakes are shallow, polymictic, and prone to internal P loading (Taranu et al., 2010; Orihel et al., 2017). As such, stratifying conditions (e.g., calm, hot days) may be key to cyanobacterial bloom formation (Orihel et al.,

2017). Interannual variations in the timing of snowmelt, and wet and dry periods can have drastic effects on watershed nutrient and solute concentrations (Ali and English, 2019). Here, streamflow is limited, and the regional hydrology is defined by depressional wetlands (i.e., prairie potholes) that are isolated during dry periods but “fill and spill” into terminal waterbodies during wet periods when subsurface connections increase contributing areas (Shook and Pomeroy, 2011; Nachshon et al., 2014). Prairie soils are high in sulfate salts (Bowman and Sachs, 2008) and thus isolated wetlands are typically saline and have high dissolved organic matter owing to evapoconcentration (M. Waiser, 2006). However, in rain-dominated extreme wet years, spillover from wetlands has been shown to markedly increase salinity of more dilute, downstream systems (Nachshon et al., 2014) and such years with a greater degree of watershed connectivity have been linked to downstream cyanobacterial bloom occurrence in some lakes (Ali and English, 2019). The alteration of flow regimes in response to these wet periods may also be consequential. For example, an extended period of poor water quality in Buffalo Pound Lake, Saskatchewan in 2015 caused a critical drinking water shortage affecting a quarter of Saskatchewan’s population (CTV, 2015; BPWTP, 2015). Increased inflow from Buffalo Pound Lake’s upstream source, Lake Diefenbaker, was used to try to ameliorate water quality issues. This event highlighted the importance of source water quality, not just quantity, to prairie water security.

The Buffalo Pound Water Treatment Plant serves approximately 260,000 people in the cities of Regina and Moose Jaw and surrounding and regularly monitors raw water quality aided by an *in situ* water quality monitoring buoy located near the plant’s intake. Successful collaboration with the water treatment plant has allowed us to assess changes in water quality over seven years across a gradient of managed source water inflow regimes and highlight the complexity of bloom formation in highly variable systems like polymictic Buffalo Pound Lake. We use *in situ* sensor data to identify cyanobacterial blooms occurring in this key drinking water reservoir over the seven study years. We describe how bloom periods, lake water chemistry, and associated environmental conditions differ between years. We underscore how interannual variability, expected to become more pronounced with a changing climate, makes predicting water quality issues difficult. Additionally, we show the potential for unintended consequences of flow alteration by providing evidence that upstream flow management changes impact water quality, the effects of which may last multiple years.

MATERIALS AND METHODS

Buffalo Pound Lake (Figure 1) is a polymictic, eutrophic lake in south-central Saskatchewan, Canada. The climate of the region is classified as semi-arid to subhumid with annual precipitation of 365 mm, 275 mm of which typically falls as rain (based on 1980-2010 Canadian Climate Normals for Moose Jaw, SK [ECCC, 2022]). The lake is narrow, long (approx. 1 km by 29 km) and shallow with a mean depth of approximately 3.0 m and a maximum depth of 5.8 m. The lake has a tightly regulated water level because it is managed with multi-

ple goals, including drinking water provision (necessitating near constant-water levels), industrial uses, and flood control for downstream lakes. Buffalo Pound receives regulated inflow from upstream Lake Diefenbaker via the Qu'Appelle River. Water is released from Lake Diefenbaker through the Qu'Appelle Dam into a channelized section of the river where it flows southeast towards Buffalo Pound Lake. The 35 km channelized section of the river runs parallel to a small marshy lake (Eyebrow Lake) before transitioning back to its naturally meandering form for a farther 62 km to Buffalo Pound Lake. A hydrometric station (Water Survey of Canada station 05JG006) is located downstream from the Qu'Appelle Dam and provides daily discharge data (ECCC, 2022b; Appendix1: Figure S1). Lake Diefenbaker is mesotrophic (North et al., 2015) and water chemistry (total dissolved/suspended solids, dissolved organic carbon) at the hydrometric station has been shown to be more dilute than that measured in Buffalo Pound Lake (WSA, 2018). In wet years, flows from the mesotrophic upstream reservoir are decreased due to limited channel capacity and desire to prevent downstream flooding. However, despite the importance of this water resource, there is limited chemical monitoring of inflows, and direct catchment inflows (i.e., from watershed tributaries) are not consistently measured.

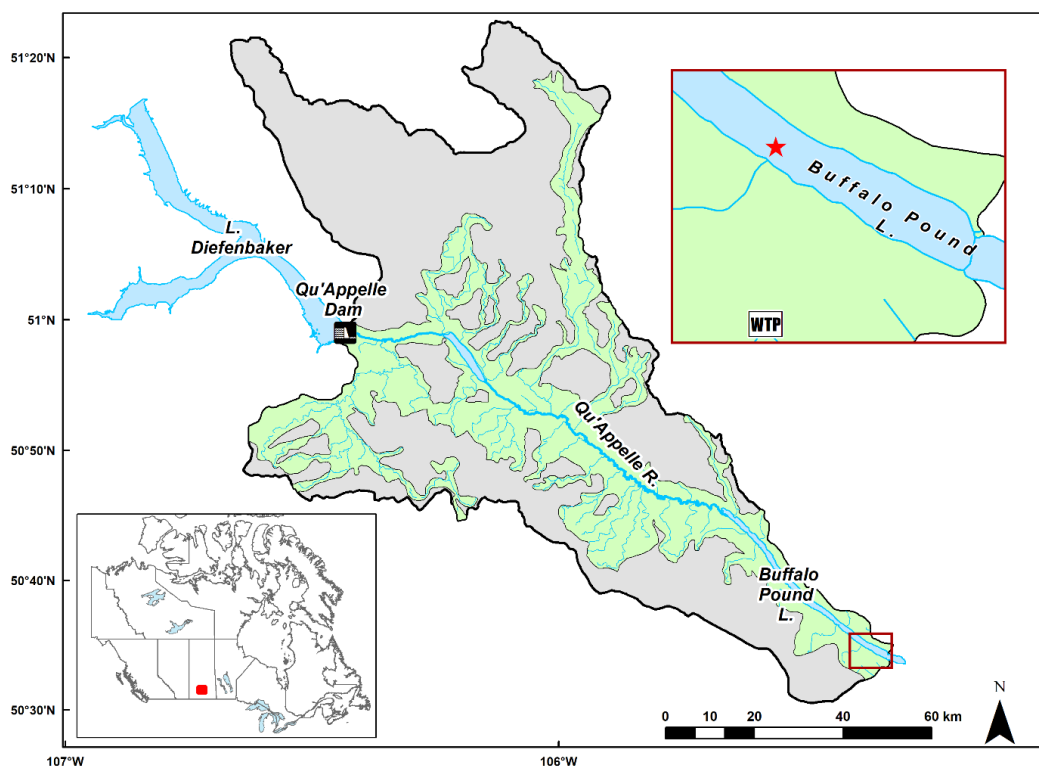


Figure 1) Map of Buffalo Pound Lake, Saskatchewan, Canada, and its gross

(grey) and effective (green) drainage area, including upstream Lake Diefenbaker (excludes L. Diefenbaker drainage area). The inset map (red outline) indicates the location of the Buffalo Pound Water Treatment Plant (WTP symbol) and the sampling site (star symbol; 50°35'8.8" N, 105°23'0.24" W) near its intakes. The Qu'Appelle River Dam, which controls water delivery from Lake Diefenbaker to Buffalo Pound Lake, is shown at the southeast arm of Lake Diefenbaker (black and white dam symbol). Geospatial data used to create this map was downloaded from CanVec and ArcGIS Online.

A water quality monitoring buoy has been located near the Buffalo Pound Water Treatment Plant (BPWTP) raw water intakes since 2014. The buoy hosts suites of water quality sensors placed on multiparameter YSI sondes (YSI Inc., USA) at 0.8 m depth including: phycocyanin (Relative Fluorescence Units, RFU), chlorophyll-a (RFU), turbidity (NTU), specific conductivity ($\mu\text{S}/\text{cm}$), photosynthetically active radiation (PAR, $\mu\text{mol}/\text{s}/\text{m}^2$), pH, temperature ($^{\circ}\text{C}$), and dissolved oxygen (mg/L). A deep-water sonde is also present at 2.8 m but was not included in this study due to issues with intermittency of operation. An additional vertical string of five T-Node (NexSens Technology Inc., USA) temperature sensors (at 0.45, 0.77, 1.23, 2.18, and 3.18 m depths), and an above-surface weather station (Vaisala, Finland) that measures wind speed (m/s) are also attached to the buoy. Data are recorded at 10-minute intervals from deployment (typically mid-May to early June) to retrieval (typically late September to early October). Deployment periods vary year to year depending on ice-off date in spring and weather conditions in autumn which affect safety of retrieval operations. During its deployment, the buoy is maintained approximately weekly. The water treatment plant also collects approximately weekly to monthly (depending on parameter) samples from the incoming raw water to assess levels of dissolved organic matter, nutrients, salts, and other chemical and physical parameters.

Cyanobacterial bloom classification

While blooms remain poorly defined (Smayda, 1997), we classified them using generalized additive models (GAMs) to identify periods of significant change in daily phycocyanin measured by the buoy in each study year. Phycocyanin, (i.e., the blue-green pigment associated with cyanobacteria) is strongly correlated with cyanobacterial biomass (e.g., Pasztaleniec et al., 2020). Prior to the construction of GAMs, data were reduced from 10-minute intervals to daily means to reduce the influence of sub-daily temporal autocorrelation. GAMs were then fit to the phycocyanin data with thin plate splines using the restricted maximum likelihood (REML) method in the R package *mgcv* (Wood, 2003; Simpson, 2018; Pederson et al., 2019). The number of basis functions (k) was adjusted for each individual model. Details and model output are available in Appendix S1 (Appendix S1:Table S1, Appendix S1:Figure S2). We also assessed the models as generalized additive mixed models (GAMMs) with the addition of a continuous time first order correlation process (CAR1) to model residual autocorrelation; however, GAMMs did not perform better than GAMs and in some cases resulted

in over smoothing therefore we proceeded with GAMs (see Simpson, 2018). Periods of significant increase and/or decrease in phycocyanin were identified by using the derivatives function in the R package gratia (Simpson, 2022) to compute first derivatives of all smooths using central finite differences. Confidence intervals around first derivatives were then assessed to identify time periods during which confidence intervals did not include zero (Simpson, 2018), thereby indicating periods of significant change in phycocyanin. Blooming periods were identified as those days during which significant changes in phycocyanin indicated an increase, followed by a non-significant plateau, and then a significant decrease. One exception to the increase-plateau-decrease sequence was made to include the period of sharp increase at the end of 2016 which did not culminate due to seasonal removal of the monitoring buoy.

Analysis of environmental variables

Principal components analysis (PCA) was used to assess associations between daily environmental variables and blooming days in Buffalo Pound Lake. Variables were selected *a priori* with the intent of reducing redundancy and collinearity while still representing variables thought to be important drivers of cyanobacterial bloom formation and water quality in Buffalo Pound. Daily mean values were included in the PCA with exception of temperature, pH, and dissolved oxygen for which daily maxima were used in order to incorporate the potential influence of diel variability. Overall, 11 variables were included in the PCA including eight *in situ* variables: phycocyanin, specific conductivity and turbidity, PAR, (max) temperature, (max) pH, (max) dissolved oxygen, and daily Schmidt stability calculated from the T-Node temperature sensors using the R package LakeAnalyzeR (Winslow et al., 2019). The three hydroclimatic variables included in the PCA were daily mean wind speed from the buoy weather station, daily precipitation from the nearest ECCC weather station, and daily discharge at the upstream hydrometric station near the Qu’Appelle dam. Prior to analysis, missing values were omitted from the daily dataset. Remaining data ($n = 690$) were then $\log_{10}+1$ transformed and scaled (to mean = 0) and analysis was carried out using R package factextra (Kassambara and Mundt, 2020).

Following the PCA, permutational multivariate analysis of variance (PERMANOVA; see Anderson, 2017) was used to assess differences between bloom and non-bloom periods, and differences between years. Data were first transformed to a Euclidean distance matrix using the vegdist function in R package vegan (Okansen et al., 2022) and then analysed using the pairwise.perm.manova function in the R package RVAideMemoire (Hervé, 2021) with 9999 permutations. We considered groups significant different at $\alpha = 0.01$ given our large sample size.

RESULTS

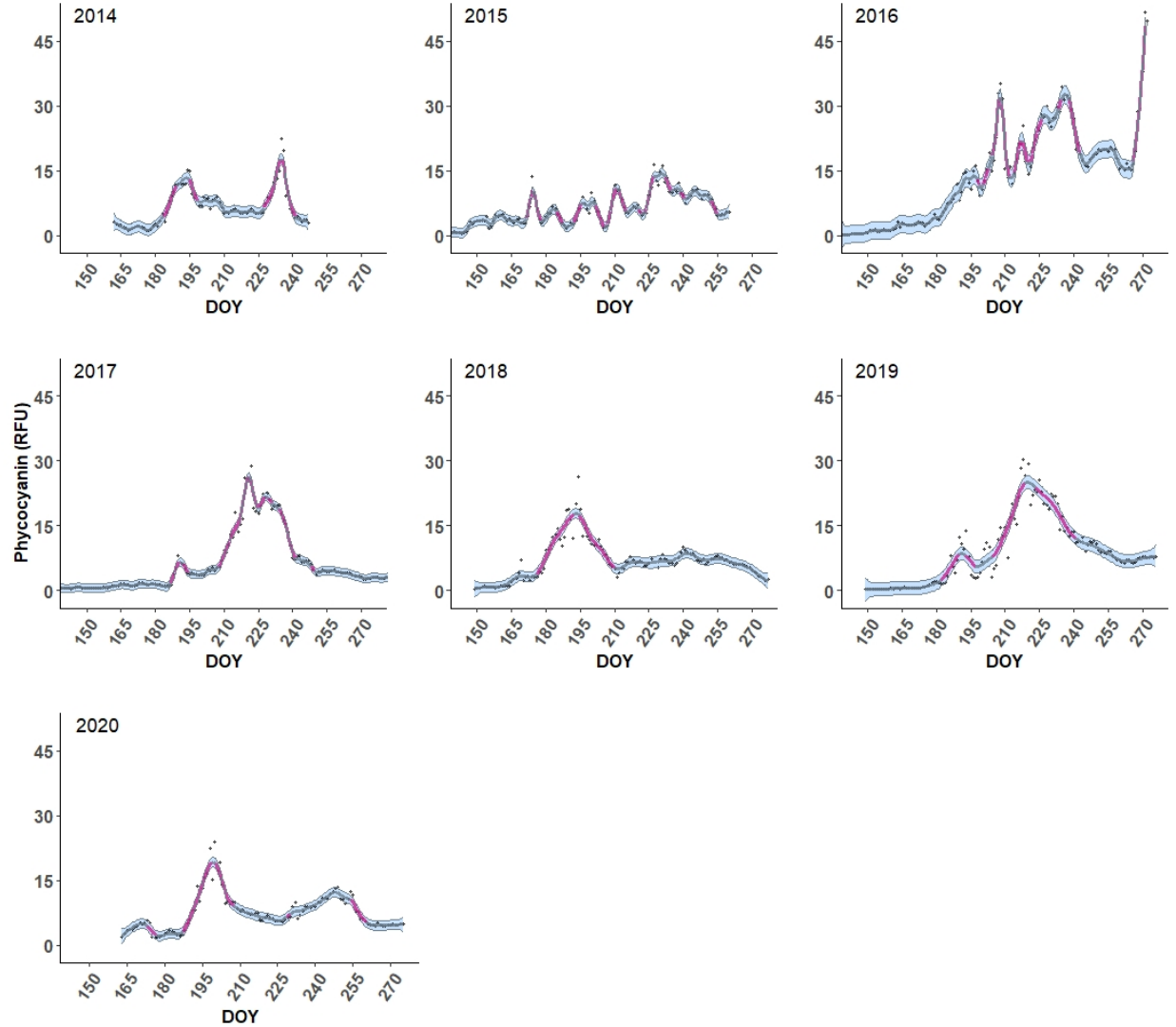


Figure 2) Periods of significant change (pink line segments) in phycocyanin (RFU) over time (day of year [DOY]) in each study year (from 2014 to 2020) calculated using first derivatives of smooth functions from generalized additive models in Buffalo Pound Lake. Periods of no change are represented by gray segments of the trend line. The modeled trend is overlaid onto daily mean phycocyanin (points). 95% confidence intervals are shown (shaded ribbon) above and below the line.

Periods of significant change with time periods following an increase-plateau-decrease sequence, hereafter referred to as blooms or bloom periods, occurred

in every study year (Figure 2). However, the number of bloom periods, their frequency, duration, and magnitude varied from year to year (Table 1 [included with supplementary files], Figure 3). There was a > two-fold variation in annual bloom days during the study period, and the number of annual bloom events ranged from one to four. Blooms in the early years (2014 and 2015) did not reach maximum phycocyanin levels seen in the following years. For example, 2014 had a median phycocyanin value of 5.61 RFU and max of 22.42 RFU with 35.5% of days classified as “in bloom” (evident as two bloom sequences in Figure 2). In 2015, phycocyanin was in a state of change 38.1% of the time and with four bloom sequences spanning 31.7% (40) days, although this percentage increased to 47.6% when the dataset is restricted to the 86-day period (from DOY 163 to 247) in which the buoy was deployed in all years. Median phycocyanin was 5.13 RFU and that year (2015) saw the lowest recorded maximum phycocyanin value (16.46 RFU).

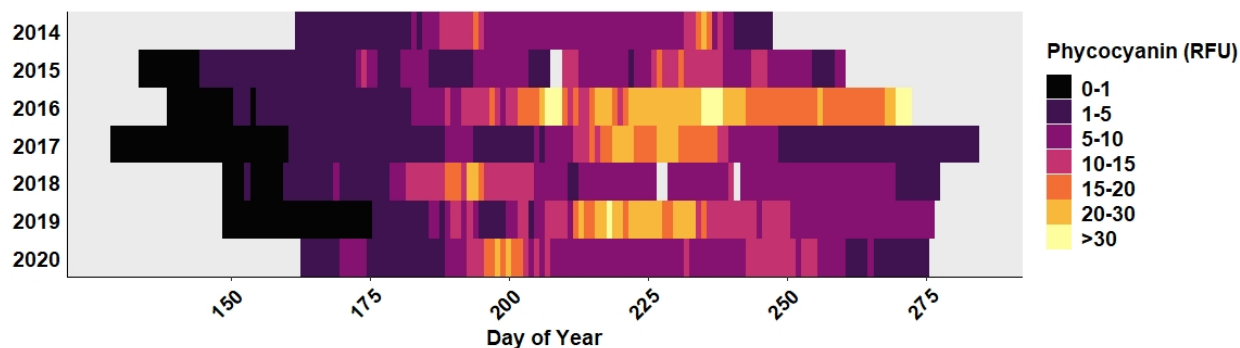


Figure 3) Heatmap showing daily phycocyanin (RFU) levels over the study years (2014 to 2020) in Buffalo Pound Lake. Phycocyanin increases from dark (e.g., black: 0-1 RFU) to light colours (light yellow: >30 RFU). Blank areas indicate no data was collected.

The most intense periods in terms of bloom magnitude were recorded in 2016 and 2019 when phycocyanin levels above 20 RFU were observed for the greatest proportion of days (26% and 12.5% of days, respectively). However, despite having the greatest phycocyanin intensity (median = 15.23 RFU, max = 51.67 RFU), time classified as “in bloom” in 2016 was only 24.4% (33 days) with a total of 30.4% of days identified as changing significantly (Figure 2) reflecting a relatively late onset of significant change in phycocyanin (DOY 204). Indeed, the percentage of days in bloom only increased to 29.1% when only common days of buoy deployment were considered. The greatest number of days (duration) spent in bloom were 52 both in 2019 and 2020 (40.6 and 46% days blooming and 40.6 and 48.7% of days changing, respectively). In 2019, >60% of common days where blooming and 2019 had the greatest median and maximum phycocyanin after 2016 – 7.67 and 30.23 RFU, respectively. 2017 had the least number of days in bloom (20.5% or 32 days) and median phycocyanin of only 3.57 RFU albeit the bloom period reached a max of 28.85 RFU.

The first two principal components (PCs) of our PCA explained >50% of the total variance in the data, with 33.1% and 20.2% explained by PC1 and PC2, respectively (Figure 4a). Nearly 75% of the variation was explained within the first three PCs (12% by PC3, see Appendix S1: Figure S3 for scree plot). PC1 was most strongly positively correlated with max pH (0.849), turbidity (0.825), phycocyanin (0.810), max dissolved oxygen (0.648) and max temperature (0.512) and to a lesser extent with Schmidt stability (0.429). PC1 was most strongly negatively correlated with PAR (-0.600) and had weaker (< 0.4) negative correlations with specific conductivity and wind speed. PC2 was most strongly positively correlated with Schmidt stability (0.700) and PAR (0.690) and to a lesser extent max dissolved oxygen (0.468), max temperature (0.467), and specific conductivity (0.410). PC2 was negatively correlated with wind speed (-0.450) and phycocyanin (-0.427). PC3 was strongly negatively correlated with Qu'Appelle R. discharge (-0.881) as well as temperature (-0.444). A full table of correlations between variables and principal components is shown in Table 2 [included with supplementary files].

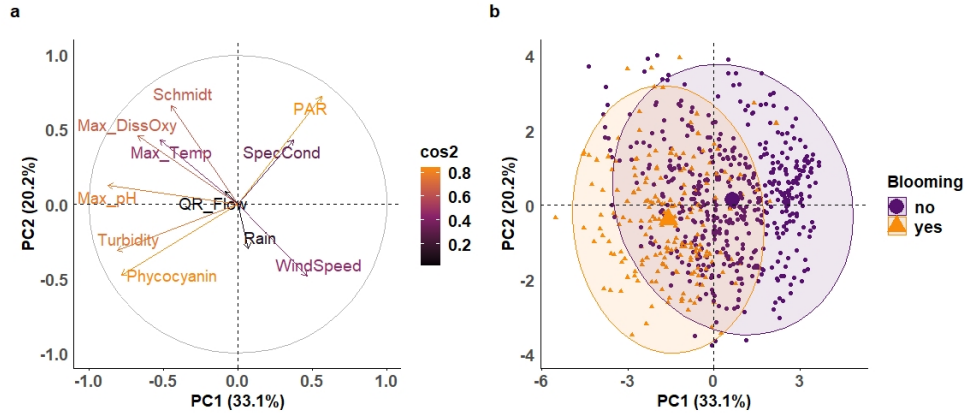


Figure 4) Results of principal components analysis (PCA) of daily environmental data (log10+1 transformed and scaled) from 2014 to 2020 in Buffalo Pounds Lake. **Figure 4a)** is a variable correlation plot with environmental variables displayed as vectors coloured by their quality of representation (\cos^2) on the principal components from low (dark purple, closest to the origin of correlation circle) to high (orange, closest to the circumference of the correlation circle). Points and ellipses (conf. int. = 0.95) on **Figure 4b)** indicate coordinates of individual days coloured by absence (dark points) or presence (light points) of a bloom period (as determined by GAMs previously). Large points indicate the

centroid mean of each ellipse.

Days considered blooming ($n = 202$) and non-blooming ($n = 488$) clustered separately (Figure 4b) and results of PERMANOVA confirmed they were indeed significantly different ($p < 0.001$). Days with cyanobacterial blooms clustered more closely with *in situ* variables (e.g., temperature, pH, oxygen, turbidity) while days without blooms clustered more closely with climatic environmental variables like wind speed, rain, and PAR as well as specific conductivity (discussed below).

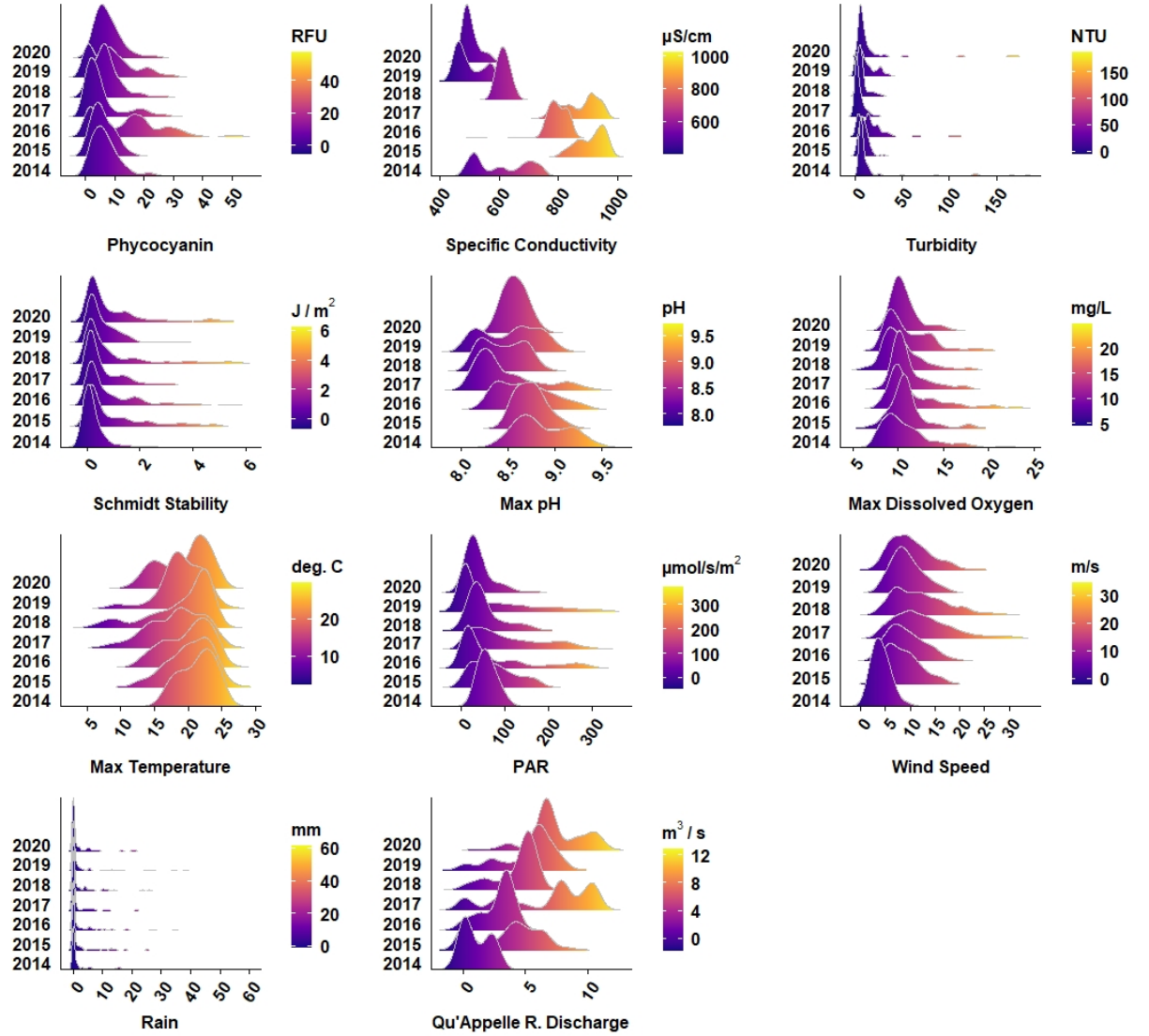


Figure 5) Ridge density plots showing the distributions of the 11 daily environmental variables analysed from 2014 to 2020 in Buffalo Pound Lake: phycocyanin, specific conductivity, turbidity, Schmidt stability, pH maxima, dissolved oxygen maxima, temperature maxima, photosynthetically active radiation (PAR), wind speed, rain, and upstream discharge in the Qu'Appelle River. All parameters are daily mean values unless otherwise noted.

Environmental variables measured by the buoy and nearby weather and hydro-metric stations varied between years (Figure 5). Indeed, PERMANOVA with year as the grouping variable indicated significant differences between years. All years were significantly different from one another with $p < 0.01$ with the exception of 2019 and 2020 for which $p = 0.021$. For example, despite calm winds and warm water temperature, 2014 had low phycocyanin as described above. However, 2014 experienced greater than typical rainfall (419 mm over the 87-day study period), and more frequent large, rain events. For example, the greatest number ($n = 6$) of days with large rain events (receiving > 25 mm) and the maximum daily rainfall recorded of all years (60 mm) was recorded in 2014. Water was more turbid on average (17.7 NTU) in 2014 than other years (range 4.12 NTU in 2017 to 15.69 NTU in 2016) – a likely influence of decreased inflow from Lake Diefenbaker and greater than normal contribution of particulates and organic matter from the surrounding watershed due to high rainfall (Water Security Agency, 2018). While we do not have a record of watershed inflow data in 2014, when springtime flows from Lake Diefenbaker were reduced (Appendix S1:Figure S1) and watershed inflows to the lake were highest, conductivity and dissolved organic matter in the lake increased leading to an uncharacteristically steep slope in conductivity values (from ~ 480 to $750 \mu\text{S}/\text{cm}$) during that year until inflows from Diefenbaker were increased in mid-summer 2015 (Figure 6a; Appendix1: Figure S1). Conductivity in the lake remained elevated for the following three years with a return to 2014 values not recorded until 2019 (mean $\sim 500 \mu\text{S}/\text{cm}$ in 2019 and 2020).

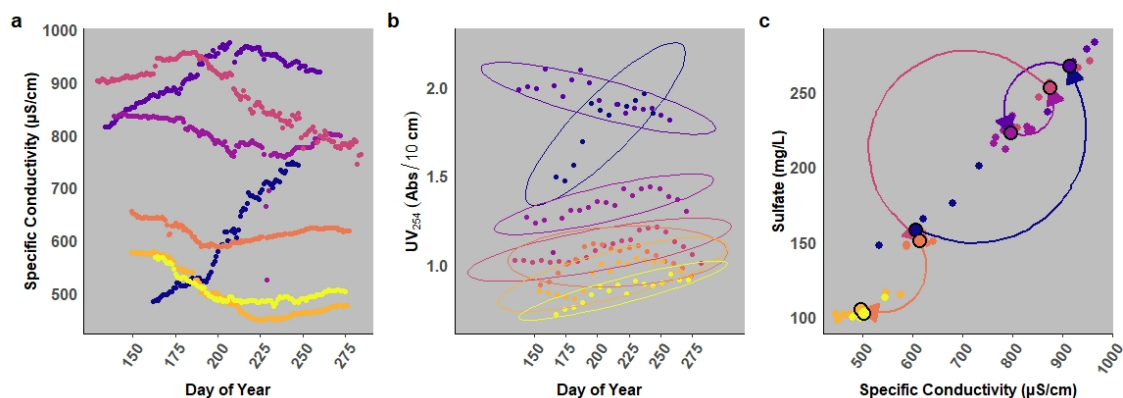


Figure 6 a) Mean daily specific conductivity from the in situ buoy in Buffalo Pound Lake (0.8m depth), **b)** approximately weekly dissolved organic matter

(as UV_{254}) from the raw water intake of the water treatment plant (WTP, 2.8 m depth) and, **c**) the conductivity – sulfate relationship (data from buoy and WTP) showing changes in water quality following upstream flow management changes from 2014 to 2020 in Buffalo Pound Lake. Arrows on Figure 6c indicate the evolution of time by connecting the annual means.

Data recorded at the water treatment plant raw water intake provides additional evidence of a shift in water source – DOM, measured as UV absorbance at 254 nm (Figure 6b, increased from 1.47 to 1.97 Abs/10 cm) and sulfate (Figure 5c; increased from 101 to 200 mg/L) showed similar trends to specific conductivity. Sulfate, the dominant anion in the region (Bowman and Sachs, 2008) was linearly correlated with conductivity ($r=0.99$). Throughout 2015, the smallest bloom year in terms of max. phycocyanin reached, DOM remained elevated (1.95 ± 0.09 Abs/10 cm as UV_{254}) before decreasing in 2016. A back-flow event due to exceedance of channel capacity also occurred in 2015 when water flowed back into Buffalo Pound Lake from its downstream outflowing tributary, the Moose Jaw River, potentially bringing with it an external input of solutes (Water Security Agency, 2018; Terry et al., 2022). However, 2016, the largest bloom year (greatest max. phycocyanin), still saw elevated conductivity (796 ± 38 $\mu S/cm$) and sulfate (222 ± 6 mg/L) concentrations compared to the proceeding years.

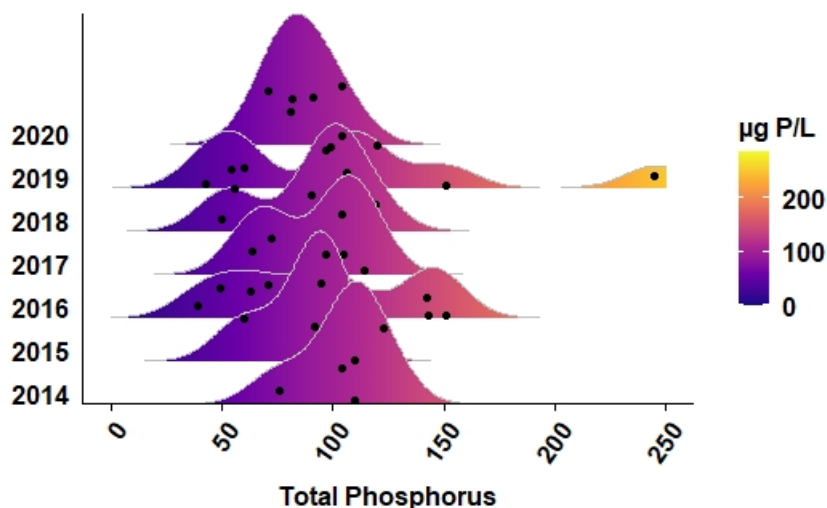


Figure 7) Ridge density plot of total phosphorus ($\mu g P/L$) recorded in the Buffalo Pound Water Treatment Plant raw water intake (weekly to monthly samples collected at 2.8 m depth). Points (randomly distributed within each ridge) indicate number of samples collected within each study year (from 2014 to 2020).

Despite being the largest bloom years, 2016 and 2019 did not stand out in any particular buoy-recorded variable other than phycocyanin, and indeed 2019 and

2020 were barely significantly different. However, nutrient data recorded by the water treatment plant, although infrequent, indicates periods of elevated phosphorus in raw water in 2016 and 2019 (Figure 7) – a finding consistent with reports of elevated nutrient concentrations in the lake in 2016 found by provincial water agency monitoring (Water Security Agency, 2018). Meanwhile, 2017 and 2018 experienced only moderate blooms and had the greatest wind speeds (12.3 ± 5.6 and 12.0 ± 5.5 m/s, respectively) and coolest max. water temperatures (18.4 ± 3.9 and 19.1 ± 4.7 °C, respectively). Finally, all years from 2014 to 2020 exhibited high variability in Schmidt stability ($CV > 100\%$) indicating occurrence of transient stratification events, possibly driving increased internal P loading, in a highly polymictic system (mean daily Schmidt stability range from 0.2 to 0.9 J/m²). Together, these findings point to the complex interplay between source water input, local climatic influence, stochastic events, and internal lake dynamics.

DISCUSSION

Blooms occur every year in Buffalo Pound Lake regardless of environmental conditions, but they are variable in number, magnitude, and timing. Our findings illustrate blooms that vary widely with phycocyanin peaks that rarely resemble one another from year to year. Such variability makes blooms like those in Buffalo Pound Lake difficult to predict and is indicative of environmental stochasticity (Isles and Pomati, 2021). Certainly, external variability in climatic conditions is a key driver of water quality and quantity in prairie regions (Schindler and Donahue, 2006; Vogt et al., 2018) and it is thus unsurprising that such variability would translate to chaotic bloom dynamics.

Nonetheless, we found some key features of days with blooms vs. those without. Based on the results from our PCA, days with blooms are more strongly associated with periods of calm wind, higher temperatures, higher pH, higher dissolved oxygen, greater water column stability and lower specific conductivity. Some of these variables – increases in pH and dissolved oxygen – are likely products of the blooms themselves. As blooms of cyanobacteria photosynthesize, they deplete dissolved carbon dioxide and produce oxygen, shifting the pool of dissolved inorganic carbon towards its carbonate/bicarbonate forms, which increases pH (Huisman et al., 2018). Meanwhile, calm, warm conditions are known to favour bloom formation, allowing cyanobacteria to proliferate, especially during periods of thermal stratification (Persaud et al., 2015; Huisman et al., 2018). Stratification events in highly polymictic systems like Buffalo Pound Lake may last only hours to days; however, our finding that Schmidt stability was associated with bloom periods highlights the importance of these transient events.

Transient stratification events can also release large pulses of internal P from sediments, particularly in polymictic systems where stratification and P release may occur repeatedly rather than (semi-)annually during turnover (Orihel et al., 2015; North et al., 2015). Polymictic lakes with low iron are demonstrably prone to internal P loading due to the importance of Fe in P sequestration through the

formation of ferrous-phosphate minerals (Orihel et al., 2015). Weekly samples from the Buffalo Pound Water Treatment Plant revealed dissolved iron concentrations below levels of detection (0.01 mg Fe/L) throughout the spring and summer months (BPWTP unpublished data). Combined with strong polymixis and past evidence of internal P loading (D’Silva, 2017), it is highly likely that P release from sediments during transient stratification is an important source of P to mid-summer algal blooms like those observed in Buffalo Pound Lake. Additionally, the majority (60% or more) of the annual land-based nutrient load is delivered to prairie waterbodies via snowmelt runoff (Glozier et al., 2006; Coriveau et al., 2013; Rattan et al., 2019) when a greater proportion of total P is likely to be in particulate form bound to soil or sediment (Koiter et al., 2013; Rattan et al., 2019). Particulate P is less bioavailable than the P released during internal loading, thus watershed contributions to Buffalo Pound, though likely nutrient-rich, are potentially less important to cyanobacteria in the immediate term than internal sources. It is also unclear if and how mid-summer blooms are able to use nutrients delivered during snowmelt given the potential for uptake and transformation within tributaries or the upstream segments of Buffalo Pound Lake itself (e.g., Terry et al., 2022) in early spring prior to cyanobacterial blooms. Therefore, climatic conditions favourable to internal loading (e.g., extended periods of calm winds, hot summer days) may be more important to mid-summer blooms than watershed sources of P *within a year*; however, watershed nutrient sources should certainly not be discounted given internal P loading is derived from legacy watershed P stored within lake sediments (Orihel et al., 2017).

The influence of conductivity, a proxy for salinity, on cyanobacterial blooms is a less studied driver but possibly an important one in prairie systems where sulfate is the dominant anion and summer TN:TP is low (Marino et al., 1990; Joshi and Jackson, 2022). Low TN:TP is often associated with N-fixing cyanobacteria abundance (Higgins et al., 2018). Nitrogenase synthesis by N-fixing cyanobacteria requires micronutrients including Mo, typically in the form molybdate (MoO_4^{2-}) in oxygenated waters. Due to similar mass-to-charge ratios, sulfate (SO_4^{2-}) can compete for uptake sites with molybdate thereby restricting its use by cyanobacteria for nitrogen synthesis (Howarth and Cole, 1985; Cole et al., 1993; Marino et al., 2003). Experimental evidence suggests molybdate uptake rates are lower at higher sulfate concentrations, a phenomenon that has been observed across a range of sulfate concentrations (0.1 to 24 mM; Marino et al., 2003) which covers sulfate concentrations (up to 282 mg/L) in Buffalo Pound Lake. It is therefore possible that growth of cyanobacteria in the early years of our study (2014, 2015) was slowed by high sulfate concentrations disrupting Mo uptake and subsequently reduced the ability of blooms to accumulate the greater biomass seen in later years.

The conductivity and sulfate concentrations we observed are likely linked to watershed influence given the highest conductivity periods were associated with the lowest inflows from upstream Lake Diefenbaker and periods of high precipitation in the region (Water Security Agency, 2018; Terry et al., 2022). Of

particular concern to the BPWTP are the high concentrations of dissolved organic matter that occurred during years with higher than usual watershed contributions. High DOM is problematic for water treatment operators because DOM interacts with chlorine during the treatment process, producing toxic disinfection by-products such as trihalomethanes (THMs). THMs are known carcinogens and are thus strictly regulated (Health Canada, 2013). However, when large amounts of buoyant cyanobacteria are present, the water treatment plant must add an additional chlorination step at the lake pumping station (pre-chlorination) to reduce algae and prevent rising floc – aggregates of organic material resistant to coagulation, which accumulates on filtration media. Rising floc has the potential to reduce treatment capacity thus pre-chlorination is often required in the summer months, but it is difficult for BPWTP to meet regulated THM limits when DOM is high (BPWTP, 2015). Moreover, THMs can continuously form and may elevate up to 50% higher as treated water travels to Regina and Moose Jaw due to retention time and continuous exposure to free chlorine (B. Kardash, pers. comm.). DOM decreases with increasing river transit times (Hosen et al., 2021), and because Lake Diefenbaker is fed by the South Saskatchewan River, it is low in DOM relative to the surrounding landscape. Meanwhile, allochthonous DOM can increase rapidly during periods of high localized landscape flow (Raymond et al., 2016). This may explain why DOM became elevated so quickly in Buffalo Pound Lake when upstream flows of low-DOM water from Lake Diefenbaker were reduced given the extremely high DOM in wetlands of the prairies (M. Waiser, 2006), which can be important flow sources during wet periods (Nachshon et al., 2014).

We suspect that we observed a mechanistic shift from a “natural” (landscape runoff dominated years 2014 to 2015) to a “managed” (L. Diefenbaker dominated years 2016 to 2020) system, each with its own implications for water quality. The “natural” scenario, observed in 2014 and 2015, is problematic for the supply of treated drinking water due to DOM concentrations remaining high throughout the year (BPWTP, 2015). Moreover, repeated flocculation issues thought to be linked to extended thermal stratification developed in spring 2015, resulting in a critical water shortage due to restricted filtration capacity – a phenomenon that had never occurred in the plant’s operating history (BPWTP, 2015; CTV, 2015). A higher particulate load and earlier than usual algal biomass accumulation associated with watershed inflows and downstream backflows combined with periods of warm, calm weather and limited downstream outflow created the extraordinary circumstances. Our results indicated that water quality conditions related to this natural runoff scenario took multiple years to dissipate – a finding that was unexpected given how quickly conditions shifted in 2014. This event highlights the potential for unintended consequences of flow management changes and/or annual climatic variability to significantly impact the recovery of critical water resources for many years.

The “managed” scenario may create worse blooms in terms of magnitude – we observed higher phycocyanin in the managed flow years (2016 to 2020) – but this observation requires further study to understand why. We know the mid-

summer bloom in a managed flow year is typically composed of N-fixers and is associated with toxin production, particularly microcystins (Painter et al., 2022); however, we lack such temporally resolved knowledge of the cyanobacterial community in the earlier years. It is possible that the chemistry of water from upstream Lake Diefenbaker is more conducive to cyanobacterial proliferation. For example, Terry et al. (2022) modelled water quality under varying water transfer scenarios (from Lake Diefenbaker to Buffalo Pound Lake) and found that total N was reduced regardless of upstream flow rate once the watershed became drier after 2015 (i.e., once watershed contributions were reduced), which is typical of regional conditions (e.g., Painter et al., 2021). Therefore, catchment N and/or other constituents associated with landscape runoff that are more prevalent in wet years may reduce the magnitude of cyanobacterial blooms. However, wet years like 2014 are also likely to include more stochastic storm events (i.e., convective storms) due to regional precipitation recycling (Raddatz, 2000, Shook and Pomeroy, 2012), which have the potential to disrupt bloom formation due to high wind speeds which increase mixing and limit light due to sediment resuspension (Thayne et al., 2022) thus this hypothesis requires further investigation.

Anthropogenic pressures on Buffalo Pound Lake are accelerating and filling the knowledge gaps around water quality is critical: plans are underway to construct a new diversion canal for regional irrigation along the Qu’Appelle River, which will terminate in Buffalo Pound Lake (Diefenbaker Irrigation Projects, 2021). It is unclear how diversions from Lake Diefenbaker via the new canal will alter water quality in Buffalo Pound. Given demand on Lake Diefenbaker is already high and delivery of its source water from the glacier-fed South Saskatchewan River is increasingly precarious (North et al., 2015b), it remains to be seen if transfers at current water volumes to Buffalo Pound will be maintained. We caution that increased water residence time in Buffalo Pound Lake may result in conditions more similar to those observed in 2014 and 2015 when outflows were reduced due to limited downstream channel capacity and problematic DOM concentrations occurred. Moreover, the project is expected to result in the expansion of economically attractive irrigable crops (e.g., potatoes, soybeans) and associated regional industry; thus, there is potential for the return of irrigation and industrial (e.g., from potato washing facilities) waters to influence Buffalo Pound Lake’s water quality, for example by increasing nutrient concentrations (Cessna et al., 2001). We anticipate additional complexities in balancing the licensing of water use with the maintenance of adequate water quality for the treatment and distribution of drinking water from Buffalo Pound Lake – particularly in dry years when increased water withdrawals will be required to maintain crops.

In systems like Buffalo Pound where flow management has the potential to significantly alter water quality, there is a need for balance between the physical constraints on water volume transfers (Terry et al., 2022), needs for irrigation and industrial uses (Diefenbaker Irrigation Projects, 2021), maintenance of water quality for drinking water (BPWTP, 2015), and ecological integrity of the lake to support other ecosystem services (Schindler and Donahue, 2006). Knowledge

of environmental flow (e-flow) requirements is imperative to maintain current and projected water demand – including e-flows for the maintenance of water quality. The scenario presented here clearly demonstrates pitfalls of a quantity over quality approach to water allocation, and trade-offs in terms of different elements of water quality with low reservoir flow/high catchment contributions leading to problems (high conductivity, high DOM) distinct from high reservoir flow/lower catchment contributions (larger cyanobacterial blooms). Perhaps more concerning for water security, is the rapidity of change. Rapid changes in source water quality are challenging to adapt to, and here, the drastic shift in water quality (as DOM, conductivity) took several years to recover to baseline conditions. We have shown that Buffalo Pound Lake and reservoirs like it are weakly resilient to climate (and management) change. For now, the province of Saskatchewan is betting big on its ability to mitigate and adapt to climate change and going “all in” on flow management to meet water demands. We caution that water management decisions must include both quality and quantity if this bet is to pay off in the long-term.

ACKNOWLEDGEMENTS

Research funding for this work was generously provided by the Canada First Research Excellence Fund as part of the Global Water Futures Initiative FORM-BLOOM (Forecasting Tools and Mitigation Options for Diverse Bloom-Affected Lakes). Funding was also provided through an academic-industry partnership with the Buffalo Pound Water Treatment Plant supported by the Mitacs Accelerate program, and ongoing funding of the Buffalo Pound Water Treatment Plant for operations. We wish to sincerely thank Blair Kardash, Laboratory and Research Manager at the Buffalo Pound Water Treatment Plant for guidance, advice, cooperation and thoughtful insight about the lake and treatment processes. We also wish to thank Katy Nugent, Cameron Hoggarth, Jay Bauer and other members of the SaskWatChE Lab for buoy deployment, sensor maintenance, and data curation and we acknowledge support of the Canada Foundation for Innovation, and University of Saskatchewan for purchase of the buoy. Additionally, we thank John-Mark Davies from the Saskatchewan Water Security Agency (WSA) for sharing knowledge and insights from years of lake monitoring by WSA.

REFERENCES

- Ali, G., and C. English. 2019. "Phytoplankton blooms in Lake Winnipeg linked to selective water-gatekeeper connectivity." *Scientific reports* 9: 1-10.
- Anderson, D. "Saskatchewan's \$4 billion irrigation plan explained." *The Narwhal*. July 12, 2022. <https://thenarwhal.ca/saskatchewan-irrigation-project-explained/>
- Anderson, M. J. 2014. "Permutational multivariate analysis of variance (PERMANOVA)." In *Wiley statsref: statistics reference online*. John Wiley & Sons, Ltd.

- Bowman, J. S., and J. P. Sachs. 2008. "Chemical and physical properties of some saline lakes in Alberta and Saskatchewan." *Saline systems* 4: 1-17.
- Buffalo Pound Water. 2015. "Board Pound Water Annual Report: 2015". https://www.buffalopoundwtp.ca/images/docs/2015_buffalo_pound_annual_report.pdf
- Cessna, A. J., J. A. Elliott, L. Tollefson, and W. Nicholaichuk. 2001. "Herbicide and nutrient transport from an irrigation district into the South Saskatchewan River." *Journal of environmental quality* 30, no. 5: 1796-1807.
- Clarke, G. K. C., A. H. Jarosch, F. S. Anslow, V. Radić, and B. Menounos. 2015. "Projected deglaciation of western Canada in the twenty-first century." *Nature Geoscience* 8, no. 5: 372-377.
- Cole, J. J., J. M. Lane, R. Marino, and R. W. Howarth. 1993. "Molybdenum assimilation by cyanobacteria and phytoplankton in freshwater and salt water." *Limnology and oceanography* 38: 25-35.
- Corriveau, J., P. A. Chambers, and J. M. Culp. 2013. "Seasonal variation in nutrient export along streams in the northern Great Plains." *Water, Air, & Soil Pollution* 224, no. 7: 1-16.
- CTV. "Buffalo Pound Lake outflows increased as water issues linger." May 29, 2015
- <https://regina.ctvnews.ca/buffalo-pound-lake-outflows-increased-as-water-issues-linger-1.2397373>
- D'Silva, L. P. 2017. "Biological and physicochemical mechanisms affecting phosphorus and arsenic efflux from prairie reservoir sediment, Buffalo Pound Lake, SK, Canada." PhD diss., University of Saskatchewan.
- Glozier, N. E., J. A. Elliot, B. Holliday, J. Yarotski and B. Harker. 2006. "Water quality characteristics and trends in a small agricultural watershed: South Tobacco Creek, Manitoba, 1992–2001." Environment Canada, Ottawa, ON.
- ECCC. 2022. "Canadian Climate Normals 1981-2010 Station Data for Moose Jaw A." Environment and Climate Change Canada.
- ECCC. 2022b. "Historical Hydrometric Data: Elbow Diversion Canal at Drop Structure (05JG006)." Environment and Climate Change Canada.
- Esri Canada & Natural Earth Vector. 2020. "ArcGIS Online" via ArcGIS Pro 10.6, Environmental Systems Research Institute, Redlands, CA.
- Hayes, N. M., H. A. Haig, G. L. Simpson, and P. R. Leavitt. 2020. "Effects of lake warming on the seasonal risk of toxic cyanobacteria exposure." *Limnology and Oceanography Letters* 5, no. 6: 393-402.
- Health Canada. "Guidelines for Canadian Drinking Water Quality: Guideline Technical Document – Trihalomethanes." October 3, 2013.

<https://www.canada.ca/en/health-canada/services/publications/healthy-living/guidelines-canadian-drinking-water-quality-trihalomethanes.html>

Hervé, M. 2022. RVAideMemoire: testing and plotting procedures for biostatistics. R package version 0.9-81-2.

Higgins, S. N., M. J. Paterson, R. E. Hecky, D. W. Schindler, J. J. Venkiteswaran, and D. L. Findlay. 2018. "Biological nitrogen fixation prevents the response of a eutrophic lake to reduced loading of nitrogen: evidence from a 46-year whole-lake experiment." *Ecosystems* 21, no. 6: 1088-1100.

Hosen, J. D., G. H. Allen, G. Amatulli, S. Breitmeyer, M. J. Cohen, B. C. Crump, Y. Lu et al. 2021. "River network travel time is correlated with dissolved organic matter composition in rivers of the contiguous United States." *Hydrological Processes* 35, no. 5: e14124.

Howarth, R. W., and J. J. Cole. 1985. "Molybdenum availability, nitrogen limitation, and phytoplankton growth in natural waters." *Science* 229, no. 4714: 653-655.

Huisman, J., G. A. Codd, H. W. Paerl, B. W. Ibelings, J. M. H. Verspagen, and P. M. Visser. 2018. "Cyanobacterial blooms." *Nature Reviews Microbiology* 16, no. 8: 471-483.

Isles, P. D. F., and F. Pomati. 2021. "An operational framework for defining and forecasting phytoplankton blooms." *Frontiers in Ecology and the Environment* 19, no. 8: 443-450.

Joshi, S., and L. Jackson. 2022. "How Might Changing Climate Limit Cyanobacteria Growth in Shallow Prairie Lakes? An Empirical Space-For-Time Evaluation of the Potential Role of Increasing Sulfate." *Advances in Environmental and Engineering Research* 3.

Kassambara, A. and F. Mundt. 2020. factoextra: Extract and visualize the results of multivariate data analyses. R package version 1.0. 7.

Koiter, A. J., D. A. Lobb, P. N. Owens, E. L. Petticrew, K. H. D. Tiessen, and S. Li. 2013. "Investigating the role of connectivity and scale in assessing the sources of sediment in an agricultural watershed in the Canadian prairies using sediment source fingerprinting." *Journal of Soils and Sediments* 13, no. 10: 1676-1691.

Diefenbaker Expansion Irrigation Projects. 2021. "Lake Diefenbaker Irrigation Projects." www.diefenbakerirrigation.ca

Marino, R., R. W. Howarth, J. Shamess, and E. Prepas. 1990. "Molybdenum and sulfate as controls on the abundance of nitrogen-fixing cyanobacteria in saline lakes in Alberta." *Limnology and Oceanography* 35, no. 2: 245-259.

Marino, R., R. W. Howarth, F. Chan, J. J. Cole, and G. E. Likens. 2003. "Sulfate inhibition of molybdenum-dependent nitrogen fixation by planktonic

- cyanobacteria under sea water conditions: a non-reversible effect.” In *Aquatic Biodiversity*, pp. 277-293. Springer, Dordrecht.
- Nachshon, U., A. Ireson, G. Van Der Kamp, S. R. Davies, and H. S. Wheeler. 2014. ”Impacts of climate variability on wetland salinization in the North American prairies.” *Hydrology and Earth System Sciences* 18, no. 4: 1251-1263.
- Natural Resources Canada. 2019. ”Topographic Data of Canada: CanVec Series.” <https://open.canada.ca/data/en/dataset/8ba2aa2a-7bb9-4448-b4d7-f164409fe056>
- North, R. L., J. Johansson, D. M. Vandergucht, L. E. Doig, K. Liber, K. Lindenschmidt, H. Baulch, and J.J. Hudson. 2015. ”Evidence for internal phosphorus loading in a large prairie reservoir (Lake Diefenbaker, Saskatchewan).” *Journal of Great Lakes Research* 41: 91-99.
- North, R. L., J. Davies, L. E. Doig, K. Lindenschmidt, and J. J. Hudson. 2015b. ”Lake Diefenbaker: the prairie jewel.” *Journal of Great Lakes Research* 41: 1-7.
- Oksanen, J., F. G. Blanchet, M. Friendly, R. Kindt, P. Legendre, D. McGlinn, P. R. Minchin, R. B. O’Hara, G. L. Simpson, and P. Solymos. 2020. Vegan: community ecology package. R package version 2.5-7.
- Orihel, D. M., D. W. Schindler, N. C. Ballard, M. D. Graham, D. W. O’Connell, L. R. Wilson, and R. D. Vinebrooke. 2015. ”The “nutrient pump:” Iron-poor sediments fuel low nitrogen-to-phosphorus ratios and cyanobacterial blooms in polymictic lakes.” *Limnology and Oceanography* 60, no. 3: 856-871.
- 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, no. 12: 2005-2029.
- Painter, K. J., R. B. Brua, G. Koehler, J. Spoelstra, and A. G. Yates. 2021. ”Contribution of nitrogen sources to streams in mixed-use catchments varies seasonally in a cold temperate region.” *Science of The Total Environment* 764: 142824.
- Painter, K. J., J. J. Venkiteswaran, D. F. Simon, S. Vo Duy, S. Sauvé, and H. M. Baulch. 2022. ”Early and late cyanobacterial bloomers in a shallow, eutrophic lake.” *Environmental Science: Processes & Impacts* 24, 1212-1227.
- Pasztaleniec, Agnieszka, Andrzej Hutorowicz, and Agnieszka Napiórkowska-Krzebietke. 2020. ”Rapid monitoring of cyanobacteria in lakes—a case study in the Wel River catchment, Poland.” *Limnological Review* 20: 41-49.
- Pedersen, E. J., D. L. Miller, G. L. Simpson, and N. Ross. 2019. ”Hierarchical generalized additive models in ecology: an introduction with mgcv.” *PeerJ* 7: e6876.
- Persaud, A. D., A. M. Paterson, P. J. Dillon, J. G. Winter, M. Palmer, and K. M. Somers. ”Forecasting cyanobacteria dominance in Canadian temperate

- lakes." 2015. *Journal of environmental management* 151: 343-352.
- Pick, F. R. "Blooming algae: a Canadian perspective on the rise of toxic cyanobacteria." 2016. *Canadian Journal of Fisheries and Aquatic Sciences* 73, no. 7: 1149-1158.
- Raddatz, R. L. 2000. "Summer rainfall recycling for an agricultural region of the Canadian prairies." *Canadian Journal of Soil Science* 80, no. 2: 367-373.
- Rattan, K. J., E. A. Blukacz-Richards, A. G. Yates, J. M. Culp, and P. A. Chambers. 2019. "Hydrological variability affects particulate nitrogen and phosphorus in streams of the Northern Great Plains." *Journal of Hydrology: Regional Studies* 21: 110-125.
- Saskatchewan. "News Release: Funding for the Western Development Museum." November 26, 2002. <https://www.saskatchewan.ca/government/news-and-media/2002/november/26/funding-for-the-western-development-museum>
- Saskatchewan. "News Release: Saskatchewan Announces \$4 Billion Irrigation Project at Lake Diefenbaker." July 2, 2020. <https://www.saskatchewan.ca/government/news-and-media/2020/july/02/irrigation-project>
- Saskatchewan (2022). "Agriculture and Agri-Value." Accessed August 23, 2022. <https://www.saskatchewan.ca/business/investment-and-economic-development/key-economic-sectors/agriculture-and-agri-value>
- Schindler, D. W., and W. F. Donahue. 2006. "An impending water crisis in Canada's western prairie provinces." *Proceedings of the National Academy of Sciences* 103, no. 19: 7210-7216.
- Schindler, D. W., R. E. Hecky, and G. K. McCullough. 2012. "The rapid eutrophication of Lake Winnipeg: Greening under global change." *Journal of Great Lakes Research* 38: 6-13.
- Shook, K. R., and J. W. Pomeroy. 2011. "Memory effects of depressional storage in Northern Prairie hydrology." *Hydrological Processes* 25, no. 25: 3890-3898.
- Shook, K., and J. Pomeroy. 2012. "Changes in the hydrological character of rainfall on the Canadian prairies." *Hydrological Processes* 26, no. 12: 1752-1766.
- Simpson, G. L. 2018. "Modelling palaeoecological time series using generalised additive models." *Frontiers in Ecology and Evolution* 6: 149.
- Simpson, G. L. 2022. 'gratia': Graceful 'ggplot'-Based Graphics and Other Functions for GAMs Fitted Using 'mgcv'. R Package v. 0.7.3.
- Smayda, T. J. 1997. "What is a bloom? A commentary." *Limnology and Oceanography* 42, no. 5part2: 1132-1136.
- Stonechild, Blair. 2006. "Aboriginal peoples of Saskatchewan." In *The encyclopedia of Saskatchewan: A living legacy* pp. 11-36. Canadian Plains Research Centre, University of Regina.

- Taranu, Z. E., D. Köster, R. I. Hall, T. Charette, F. Forrest, L. C. Cwynar, and I. Gregory-Eaves. 2010. "Contrasting responses of dimictic and polymictic lakes to environmental change: a spatial and temporal study." *Aquatic sciences* 72: 97-115.
- Terry, J., J. Davies, and K. Lindenschmidt. 2022. "Buffalo Pound Lake—modelling water resource management scenarios of a large multi-purpose prairie reservoir." *Water* 14, no. 4: 584.
- Thayne, M. W., B. M. Kraemer, J. P. Mesman, B. W. Ibelings, and R. Adrian. 2022. "Antecedent lake conditions shape resistance and resilience of a shallow lake ecosystem following extreme wind storms." *Limnology and Oceanography* 67: S101-S120.
- Vogt, R. J., S. Sharma, and P. R. Leavitt. 2018. "Direct and interactive effects of climate, meteorology, river hydrology, and lake characteristics on water quality in productive lakes of the Canadian Prairies." *Canadian Journal of Fisheries and Aquatic Sciences* 75: 47-59.
- Waiser, M. J. 2006. "Relationship between hydrological characteristics and dissolved organic carbon concentration and mass in northern prairie wetlands using a conservative tracer approach." *Journal of Geophysical Research: Biogeosciences* 111, no. G2.
- Waiser, B. 2006. "Our Shared Destiny? Saskatchewan in 1905 and 2005." *Acadiensis* 35, no. 2: 157-162.
- Water Security Agency. 2018. "Qu'Appelle Nutrient Mass Balance 2013-2016." Water Quality and Habitat Assessment Services Report WQ201812-01.
- Wheater, H., and P. Gober. 2013. "Water security in the Canadian Prairies: science and management challenges." *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences* 371, no. 2002: 20120409.
- Winslow, L., J. Read, R. Woolway, J. Brentrup, T. Leach, J. Zwart, S. Albers, and D. Collinge. 2019. 'rLakeAnalyzer': Lake Physics Tools.
- Wood, S. N. 2003. "Thin plate regression splines." *Journal of the Royal Statistical Society: Series B (Statistical Methodology)* 65: 95-114.