Received: 24 October 2023 | Revised: 11 April 2024 | Accepted: 1 May 2024

DOI: 10.1111/fwb.14260

ORIGINAL ARTICLE

Freshwater Biology WILEY

Impacts of hydrologic management on the eutrophication of shallow lakes in an intensive agricultural landscape (Saskatchewan, Canada)

Cale A. C. Gushulak[1,2] | **Amir M. Chegoonian**[1,3] | **Jared Wolfe**[4] | **Kristen Gray**[1] | **Stefano Mezzini**[1] | **Bjoern Wissel**[2] | **Brenda Hann**[5] | **Helen M. Baulch**[4,6] | **Kerri Finlay**[1,2] | **Peter R. Leavitt**[1,2]

[1]Department of Biology, University of Regina, Regina, Saskatchewan, Canada
[2]Institute of Environmental Change and Society, University of Regina, Regina, Saskatchewan, Canada
[3]Department of Geography and Environmental Management, University of Waterloo, Waterloo, Ontario, Canada
[4]Global Institute for Water Security, University of Saskatchewan, Saskatoon, Saskatchewan, Canada
[5]Department of Biological Sciences, University of Manitoba, Winnipeg, Manitoba, Canada
[6]School of Environment and Sustainability, University of Saskatchewan, Saskatoon, Saskatchewan, Canada

Correspondence
Cale A. C. Gushulak, Department of Biology, University of Regina, Regina, Saskatchewan, S4S 0A2, Canada.
Email: cale.gushulak@uregina.ca; calegushulak@gmail.com

Present address
Jared Wolfe, Natural Resources Canada, Ottawa, Ontario, Canada
Stefano Mezzini, Irving K. Barber Faculty of Science, University of British Columbia, Kelowna, British Columbia, Canada
Bjoern Wissel, Laboratoire d'écologie Des hydrosystèmes Nautrels et anthropisés (LEHNA), Claude Bernard University Lyon 1, Villeurbanne, France

Funding information
Natural Sciences and Engineering Research Council of Canada

Abstract

1. Hydrologic management of shallow lakes is often undertaken to prevent fluctuations in lake level, and to ensure sufficient water volume for economic, domestic, and recreational uses, but there is inconsistent evidence of whether lake-level stabilisation through hydrological management promotes or hinders eutrophication.
2. Here we used multi-proxy paleolimnological assessments of water quality (sedimentary carbon, nitrogen, total phosphorus, fossil pigments), and zooplankton community ecology (fossil Cladocera assemblages), combined with Landsat-derived estimates of lake surface area in two shallow eutrophic lakes, in the Prairies of southern Saskatchewan, Canada, to quantify how 8 decades of contrasting hydrological management strategies (continuous or intermittent) affect primary production and phytoplankton composition.
3. Analysis revealed that irregular hydrological management of Pelican Lake led to sharp increases in primary production concomitant with lake-level decline. In contrast, continuously managed Buffalo Pound Lake, a drinking water reservoir for regional cities, exhibited slow, persistent eutrophication over decades despite active regulation of water levels. In both lakes, strong correlations of $\delta^{15}N$ values with pigments from diazotrophic cyanobacteria (canthaxanthin) showed that N_2-fixation increased during eutrophication irrespective of the timing of change. Finally, variation in fossil cladoceran density and composition reflected changes in pelagic and littoral habitats (e.g., reduced macrophyte cover) due to changes in both lake level and water quality.
4. Basin comparison shows that while hydrologic management can moderate water quality degradation due to lake-level change, it does not prevent eutrophication when nutrient influx remains high.

5. Given that regional water availability is forecast to decline in coming decades, we anticipate that continued hydrological management will be unavoidable and will be unable to improve water quality unless nutrient influx is also controlled.

KEYWORDS
Buffalo Pound Lake, lake level, Paleolimnology, Pelican Lake, remote sensing

1 | INTRODUCTION

Declines in water quality and increased frequency of algal and cyanobacterial blooms in global lakes and reservoirs is a persistent environmental challenge (Huisman et al., 2018; Paerl & Huisman, 2009; Taranu et al., 2015). Long-term eutrophication of inland waters has been attributed to increased influxes of phosphorus (P) and nitrogen (N), as well as impacts of climate change due to rising water temperatures, decreased periods of ice-cover (longer growth seasons), and intensified thermal stratification (Jeppesen et al., 2014; Meerhoff et al., 2022). In the northern Great Plains, many shallow waterbodies exhibit substantial increases in both primary production and toxic cyanobacterial blooms due to nutrient inputs from over a century of intensive agriculture (Hall, Leavitt, Quinlan, et al., 1999; Maheaux et al., 2016; Tse et al., 2015), urban pollution (Bunting et al., 2016; Leavitt et al., 2006), and hydrologic fluctuations (e.g., lake level, residence time, flow regimes) related to climate change (Bjorndahl et al., 2022; Gibson et al., 2016; Mosley, 2015).

Lakes in the northern Great Plains of North America are susceptible to hydroclimatic stress due to complex and variable interactions between the Arctic, Pacific, and Gulf air masses (Bryson & Hare, 1974), teleconnections between major climate systems (e.g., Pacific Decadal Oscillation, El Niño-southern Oscillation), and irregular variation in jet-stream position (Gurrapu et al., 2016; Hurrell, 1995; Michels et al., 2007; Shabbar, 2006). Such climate forcing results in high seasonal and inter-annual variability in regional precipitation, run-off, and lake level (Haig et al., 2020; Pham et al., 2009; van der Kamp et al., 2008) that can affect primary production over seasonal-to-millennial time scales (Bjorndahl et al., 2022; Michels et al., 2007; Mosley, 2015; Vinebrooke et al., 1998). In principle, increased evaporative forcing can reduce lake level, increase nutrient concentrations, alter light penetration, and favour elevated primary production that can lead to deep-water anoxia, elevated sediment nutrient release (both N and P), and sustained eutrophication (da Costa et al., 2016; Jane et al., 2021; Özen et al., 2010; Qin et al., 2006; Søndergaard et al., 2013; Vinebrooke et al., 1998). However, hydroclimatic changes that cause declines in lake level may also be associated with alterations in terrestrial nutrient transport and influx (da Costa et al., 2016; Mosley, 2015; Özen et al., 2010) leading to biogeochemical changes that favour blooms of N_2-fixing cyanobacteria depending on whether N or P is more strongly affected by variation in internal and external nutrient loading (Joshi & Jackson, 2022; Shayo & Limbu, 2018). Thus, while large changes in water level are characteristic of lakes in the northern Great Plains (Bjorndahl et al., 2022; van der Kamp et al., 2008), it is unclear whether lake-level change, as well as its direct management through damming and water conveyance, has a consistent effect on lake eutrophication.

Long-term variation in climate processes has resulted in severe droughts in the Canadian Prairies, including multi-year events in the 1930s, 1980s, and 2000s (Bonsal et al., 2013; Nkemdirim & Weber, 1999). Reduced annual precipitation, particularly in winter, can result in water-level declines of 2–10 m in prairie lakes (van der Kamp et al., 2008). Despite recent increases in annual precipitation, general circulation and regionally downscaled models suggest that the Prairies will remain susceptible to droughts (Asong et al., 2016; Dibike et al., 2012) due to strengthened evaporation and reduced spring run-off (Dumanski et al., 2015; Pham et al., 2009; Tanzeeba & Gan, 2012). With further declines in winter precipitation (Akinremi et al., 1999; Asong et al., 2016), regional lake levels may begin to decline outside periods of drought (Dibike et al., 2017) and intensify water quality degradation (Bjorndahl et al., 2022; Vinebrooke et al., 1998). This sub-humid climate, combined with increasing demand for water for agricultural, industrial, and domestic purposes (Gan, 2000; Wheater & Gober, 2013), has resulted in hydrological management of many rivers and lakes in the region to address perceived water insecurity (Hassanzadeh et al., 2019).

The hydrology of rivers and lakes in the Canadian Prairies has been managed for over 100 years. For example, outflows of lakes in the Qu'Appelle River catchment of southern Saskatchewan were initially impounded in the late 1800s to suppress high variability in water levels and maintain river flow for year-round water conveyance between ecosystems. Many Qu'Appelle lakes are now supplemented by water diverted from Lake Diefenbaker (Figure 1), a headwater reservoir completed in 1967 by damming the South Saskatchewan and Qu'Appelle rivers (North et al., 2015; Smith & Kells, 1993). Since the 1960s, reservoir water has been diverted into shallow Buffalo Pound Lake, the drinking water source for the cities of Regina and Moose Jaw, Saskatchewan, before it flows downstream through the rest of the catchment (Hall, Leavitt, Dixit, et al., 1999; Terry et al., 2022). Despite continuous lake-level management over the last 6 decades, and vested interest in maintaining good water quality in the lake, preliminary paleolimnological (Hall, Leavitt, Dixit, et al., 1999) and water-quality studies (Hayes et al., 2019; Hosseini et al., 2018; Terry et al., 2022) suggest that Buffalo Pound Lake has experienced substantial eutrophication and now exhibits regular blooms of toxic cyanobacteria (Hayes et al., 2019; Painter et al., 2022). Similar biogeochemical and ecological responses to concurrent changes in

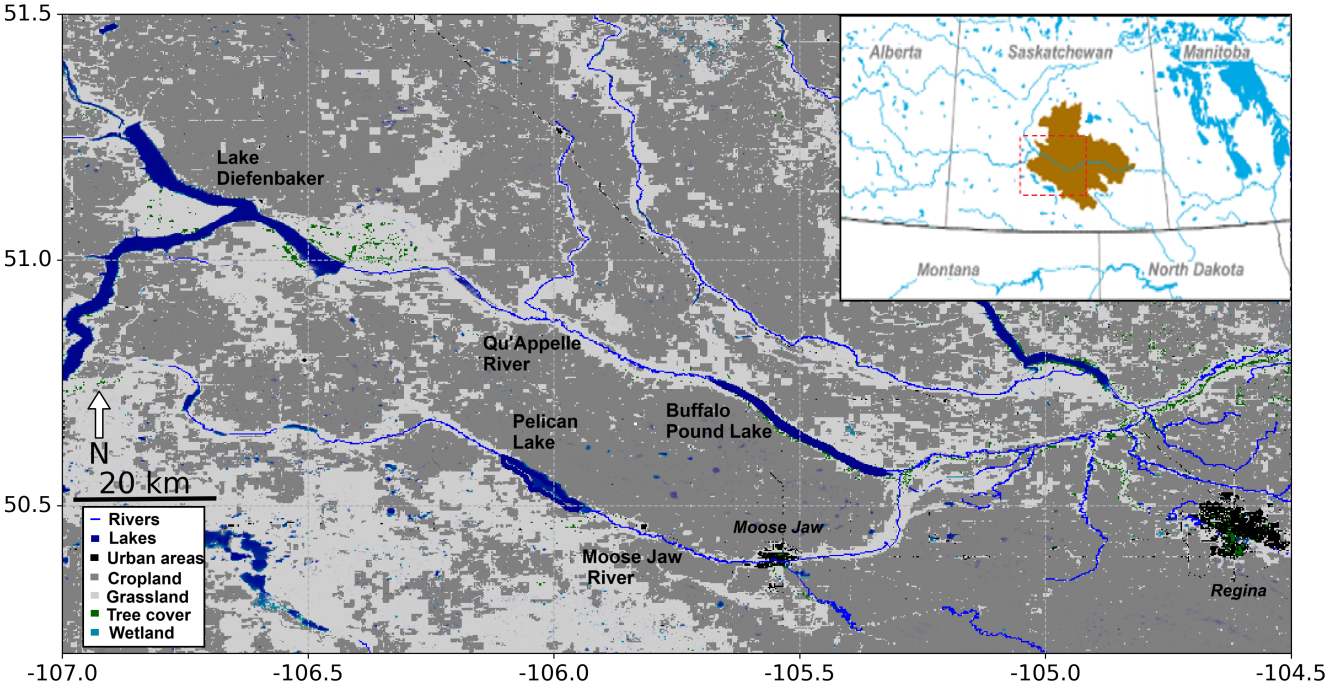

FIGURE 1 Location of Buffalo Pound Lake and Pelican Lake within the Qu'Appelle River Basin in southern Saskatchewan, Canada. Lake Diefenbaker, the source water to the Qu'Appelle River is also shown upstream of Buffalo Pound Lake. The urban centers of Moose Jaw and Regina are highlighted. Land-use data were collected from a global land-cover map generated from 10-m resolution Sentinel-2 data collected in 2021 (Zanaga et al., 2022). Inset denotes the study region within the Qu'Appelle River catchment (highlighted) of Saskatchewan.

climate, land use, and hydrological management have been observed elsewhere (Coops et al., 2003; Hambright et al., 2008); consequently, it remains uncertain whether intensive hydrological management has an overall beneficial or detrimental effect on the eutrophication of shallow lakes.

In this study, we present a multi-proxy, paleolimnological comparison of a pair of adjacent prairie lakes that either receive intensive and continuous (Buffalo Pound Lake) or irregular hydrological management (Pelican Lake) to evaluate how these contrasting regulatory strategies affect water quality and cyanobacterial abundance. The lakes are separated by *c.* 40 km (Figure 1) and experience common land-use and climate conditions (AAFC, 2022; ECCC, 2023a), but only Buffalo Pound Lake is subject to long-term stabilisation of lake levels (Hall, Leavitt, Dixit, et al., 1999; Terry et al., 2022). Here we compared historical time series of lake surface area derived from satellite images with fossil metrics of past primary production and community composition to quantify the decadal-scale effects of hydroclimate change and lake management on eutrophication. Based on previous studies of relatively unmanaged lakes (Bjorndahl et al., 2022; Vinebrooke et al., 1998), and 30-years of water quality trends in Qu'Appelle system lakes (Hall, Leavitt, Dixit, et al., 1999; Hall, Leavitt, Quinlan, et al., 1999; Vogt et al., 2018), we hypothesised that the long-term lake level management of Buffalo Pound Lake should buffer primary production responses to decadal variation in regional hydroclimate. Similarly, we predicted that irregularly managed Pelican Lake should display substantial water-level changes related to variation in hydroclimate, resulting in highly dynamic and concomitant changes in phytoplankton production and composition.

2 | METHODS

2.1 | Site descriptions

Buffalo Pound Lake (50°35.887′ N 105°24.525′ W) and Pelican Lake (50°30.541′ N 105°57.896′ W) are two shallow eutrophic lakes (Table 1) located within the headwaters of the Qu'Appelle River drainage basin, a *c.* 52,000 km^2 expanse of Prairie located in southern Saskatchewan, Canada (Figure 1). Land use in the catchments of both lakes is *c.* 65% agricultural, including both crop and livestock production (AAFC, 2022; Hall, Leavitt, Quinlan, et al., 1999; Pham et al., 2008). The climate of the region is cold continental, with mean annual temperatures of 4.2°C, mean summer temperatures of 18.5°C, and mean winter temperatures of −11.0°C in the 1981–2010 period (ECCC, 2023a; Canadian Climate Normals, Moose Jaw, SK). Mean annual temperatures have risen by *c.* 1°C since 1900, although annual means can vary by up to 6°C (Figure S1; ECCC, 2023b, Adjusted and homogenised Canadian climate data, Moose Jaw, SK). Current annual precipitation is 365 mm/year (ECCC, 2023a), with inter-annual differences of >300 mm (Figure S2). Despite this variability, overall trends in annual precipitation were generally stable to 1990, after which values increased substantially. This increase was driven by large increases in summer precipitation since *c.* 1990 (Figure S2), while winter precipitation has steadily decreased since the 1980s (Figure S2; ECCC, 2023b).

Buffalo Pound Lake serves as the drinking water reservoir for the cities of Regina and Moose Jaw, Saskatchewan, and has received considerable hydrological management to maintain stable

TABLE 1 Mean values of select physical and chemical parameters of Pelican and Buffalo Pound lakes over summer (June–August) 2015. Pelican Lake data provided by Saskatchewan Water Security Agency (WSA).

Variable	Mean
Pelican Lake	
Water depth	3.3 m
Secchi disc depth	0.5 m
Dissolved oxygen	8.50 mg/L
Conductance	3694 μS/cm
pH	8.67
Dissolved organic carbon	39.9 mg/L
Ammonia	0.04 mg/L
Nitrate	0.09 mg/L
Total nitrogen	3.3 mg/L
Dissolved orthophosphate	0.56 mg/L
Total phosphorus	0.82 mg/L
Chlorophyll a	43.52 μg/L
Buffalo Pound Lake	
Water depth	4.2 m
Secchi disc depth	1.1 m
Dissolved oxygen	7.64 mg/L
Conductance	882 μS/cm
pH	8.49
Dissolved organic carbon	8.2 mg/L
Ammonia	4.63 mg/L
Nitrate	0.11 mg/L
Total dissolved nitrogen	0.82 mg/L
Soluble reactive phosphorus	20.7 μg/L
Total dissolved phosphorus	31.8 μg/L
Chlorophyll a	31.96 μg/L

lake levels. The lake's outlet to the Qu'Appelle River was initially impounded in 1939 to stabilise water levels in response to drought, after which levels were raised by *c.* 2 m in 1952 by pumping water overland from the South Saskatchewan River (Hall, Leavitt, Dixit, et al., 1999). Since 1963, water has been diverted into Buffalo Pound Lake through the Upper Qu'Appelle River channel from mesotrophic Lake Diefenbaker (North et al., 2015). These modifications raised the maximum depth of Buffalo Pound Lake from *c.* 3 to *c.* 5 m, increased surface area from *c.* 5 to *c.* 28 km^2, lengthened water residence times, and reduced seasonal water-level fluctuations to *c.* 1 m in most years since the mid-1960s (Figures S3 and S4; Hall, Leavitt, Dixit, et al., 1999; Terry et al., 2022). The lake is currently held at a stable maximum depth of 5 m and has a mean depth of 3.8 m.

Pelican Lake is located *c.* 40 km southwest of Buffalo Pound Lake and is of similar surface area (28 km^2), but presently has a shallower maximum depth of 2.9 m. The lake drains into the Moose Jaw Creek which joins the Qu'Appelle River immediately downstream of Buffalo Pound Lake (Figure 1). Pelican Lake has experienced similar land-use and climate patterns to Buffalo Pound Lake (AAFC, 2022; ECCC, 2023a, 2023b), but has received irregular and lesser hydrological management. Water levels were originally managed by damming a series of channelled wetlands upstream of the lake in 1968 and 1974 to conserve bird habitat around the lake (IBA Canada, 2023). A third dam was built at the outlet of the lake in 1989 to sustain water levels following substantial declines that occurred during droughts in the late 1980s (IBA Canada, 2023; SIPA, 2008). Finally, Pelican Lake began to receive intermittent transfers of water from mesotrophic Lake Diefenbaker via upstream Thunder Creek in 1991 (SIPA, 2008). Water conveyance to Pelican Lake is minor relative to that of Buffalo Pound Lake (Figure S4; data from Saskatchewan Water Security Agency), with most of its hydrological inputs originating from nutrient-rich terrestrial run-off within the catchment. In addition, the hydrological management of Pelican Lake did not stabilise its water levels until the late 1990s, with water levels varying greatly prior to this time.

2.2 | Remote sensing

Time series of changes in lake area were retrieved from Landsat heritage images (Figures 2 and S5). The Multi-Spectral Scanner (MSS) onboard Landsat 1–3 was used to reconstruct lake surface area from 1972 to 1983. Meanwhile, the Thematic Mapper (TM) sensor onboard Landsat 4–5 was used from 1984 to 2011, and the Enhanced TM (ETM+) onboard Landsat 7 was employed for 2012–2021. For both lakes, clear sky level-1 images of the mentioned sensors captured from May to October (i.e., the open water season) were downloaded (441 images in total) from the U.S. Geological Survey (USGS, 2023). A threshold on radiance values in mid- and short-wave infrared bands of the Multi-Spectral Scanner (band 7; 0.8–1.1 μm) and TM/Enhanced TM sensors (band 5, 1.55–1.75 μm) were used to isolate lake surface from land cover. Lake areas for each image were then computed given the spatial resolution of the sensors and were summarised using a median operator to obtain mean water surface area during the ice-free period. Annual variation in lake surface area was estimated as the standard deviation of the lake area within any given year.

2.3 | Sediment coring

Sediment cores were collected from Buffalo Pound and Pelican lakes in summer 2014 using a Glew gravity corer (Glew, 1989) and were immediately sectioned on shore. The Buffalo Pound Lake core (50°35.887′ N 105°24.525′ W) was collected from *c.* 4.3 m lake depth, was 41 cm long, and was sectioned at 0.7-cm intervals based on known sedimentation rates of previously collected cores (Hall, Leavitt, Dixit, et al., 1999; Hall, Leavitt, Quinlan, et al., 1999). The 23-cm long Pelican Lake core (50°30.541′ N 105°57.896′ W) was obtained from *c.* 2.9 m lake depth, and sectioned at 0.5-cm intervals. Sediment samples were stored in the dark at 4°C at the University of Regina's Institute of Environmental Change and Society. Sediment

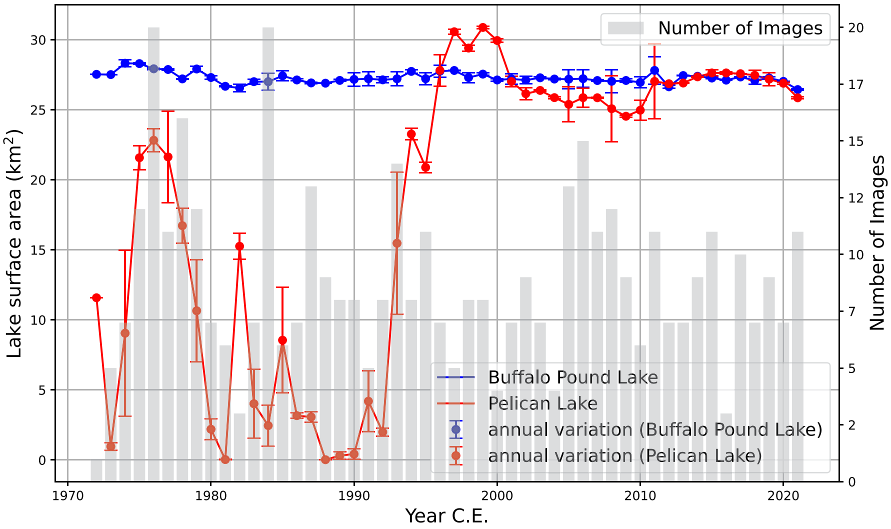

FIGURE 2 Changes in the surface area of Buffalo Pound and Pelican lakes as inferred by Landsat images from 1972 to 2021. Error bars represent the amount of surface area variation detected by the images during the ice-free season (May–October) of each year. Vertical grey bars indicate the number of images used to calculate the median lake surface areas for each year.

intervals were freeze-dried shortly after collection in preparation for laboratory analysis which occurred throughout late 2014 and 2015.

2.4 | Paleolimnological methods

Sediment chronology was estimated using activities of ^{210}Pb and ^{137}Cs radionuclides measured using low-background gamma spectrometry analysis on 15 evenly spaced sediment intervals from the Buffalo Pound and Pelican lakes cores (Appleby et al., 1986; Schelske et al., 1994). The sediment core age–depth relationships were initially determined using constant rate of supply (CRS) models (Binford, 1990; Schelske et al., 1994) based solely on ^{210}Pb activities (as ^{137}Cs activities were not conclusive), but were then refined using shape-constrained additive models (SCAMs) with monotone decreasing P-splines and generalised cross-validation smoothness parameter selection in the *scam* package v. 1.2–14 (Pya, 2023) in R v. 4.2.3 (R Core Team, 2023).

Sedimentary stable isotope values for total carbon (δ^{13}C) and nitrogen (δ^{15}N) were quantified following the methods of Savage et al. (2004) using a Thermo Scientific Delta V Plus isotope ratio mass spectrometer connected to a Costech ECS 4010 Elemental Analyser via a ConFlo IV interface. Isotopic ratios were expressed using the standard delta (δ) notation in ‰ relative to Vienna-Pee Dee Belemnite and atmospheric N_2 for C and N, respectively. Elemental composition of carbon and nitrogen were estimated as the percent (C%, N%) dry mass and were used to calculate C:N mass ratios.

Total phosphorus (TP) content of Buffalo Pound Lake and Pelican Lake sediments was quantified using the protocols of Hieltjes and Lijklema (1980) and Engstrom and Wright Jr. (1984). TP was measured as the combination of ortho-phosphate extracted by digestion of sediments with concentrated H_2SO_4, K_2SO_4, and HgO, and chemically exchangeable P that was extracted in a second aliquot of 1 M NH_4Cl. TP concentrations were then standardised to the dry mass of the sediments as mg P/g dry mass.

Photosynthetic pigments were quantified using high-performance liquid chromatography (HPLC) following Leavitt and Hodgson (2001). Pigments were extracted from dried sediments in an 80:15:5 (by volume) solution of HPLC-grade acetone, methanol, and water, and were then filtered through 0.22-μm-pore PTFE filters, dried under inert N_2 gas, and re-suspended in 500 μL injection solution. Pigment concentrations were measured with an Agilent model 1100 HPLC system equipped with a photodiode array detector and calibrated with authentic pigment standards and using Sudan II (Sigma-Aldrich) as an internal reference. Sedimentary pigments were expressed as nmol/g C as derived from previous C content analysis. Identification of fossil chlorophylls, derivatives, and carotenoid pigments was restricted to taxonomically diagnostic biomarkers including fucoxanthin (diatoms, chrysophytes, some dinoflagellates), diatoxanthin (mainly diatoms), alloxanthin (cryptophytes), pheophytin b (chlorophytes), echinenone (total cyanobacteria), canthaxanthin (Nostocales cyanobacteria), and aphanizophyll (N_2-fixing cyanobacteria). Concentrations of lutein (chlorophytes) and zeaxanthin (cyanobacteria) co-eluted and so were used as an indicator of bloom-forming taxa (Leavitt & Hodgson, 2001). Ubiquitous β-carotene and the chemically stable chlorophyll a derivative, pheophytin a, were used as estimates of total phototroph production (Leavitt & Hodgson, 2001).

Cladoceran microfossils were isolated from the Buffalo Pound and Pelican cores following the protocol of Hann (1989). Freeze-dried and tared sediment samples were soaked overnight in 10% KOH at room temperature and gently swirled before microfossils were concentrated onto a chemically inert screen (37-μm mesh). Slides for enumeration were prepared by dispersing 50 μL of an aqueous suspension of microfossils into glycerine jelly and covering with a coverslip. Cladoceran remains were counted until a minimum of 100 microfossils were enumerated. In rare cases, up to 10 slides were counted. Cladoceran microfossils were identified to the species level and verified using reference slides from the region (Suchy et al., 2010). Remains were tabulated as head shields, carapaces or *others* (mandibles, post-abdomens, claws, antennal segments), and the most numerous remains were used to calculate microfossil abundance. Historical changes in invertebrates were presented at the genus level, abundances were expressed as relative abundance (%), and as the concentrations were expressed as the number of individuals per mass of dry sediment (remains/g dry mass), a metric that is linearly correlated to invertebrate water column densities (Hann et al., 1994). Taxa that occurred in only one sample or were <5% of the fossil sum were summarised as being part of *other littoral taxa* or *other planktonic taxa* in the assemblage.

2.5 | Numerical analysis

Generalised additive models (GAMs) were used to fit temporal trends of $\delta^{13}C$ and $\delta^{15}N$, C:N mass ratios, and sedimentary content of C, N, and TP in the Buffalo Pound and Pelican lakes cores using the *mgcv* package v. 1.8–41 (Simpson, 2018; Wood, 2017, 2023) in R. Isotope compositions ($\delta^{13}C$, $\delta^{15}N$) were modelled using a Gaussian family of distributions, while strictly positive C:N and TP data used Gamma families. Lastly, C and N content used beta families, after scaling the response values to the interval (0, 1). The observations in all models were weighted using the temporal resolution of the sediment samples (Simpson, 2018). Time-series of pigment concentrations were modelled with GAMs as those described above, but a Tweedie distribution was used to allow for intervals with non-detections (since the Gamma family does not support zeroes; see Wood et al., 2016). A final GAM was fit to the ratio of *Daphnia* spp. and *Bosmina* spp. within Buffalo Pound and Pelican lakes to determine periods of change in Cladocera assemblages. In each GAM, change over time was estimated using a smooth effect of SCAM-inferred date with adaptive splines, and smoothness parameters were estimated using Restricted Maximum Likelihood (Wood, 2011; Simpson, 2018).

Cross correlation analysis was performed between $\delta^{15}N$ values and concentrations of potentially N_2-fixing cyanobacterial pigments (canthaxanthin, aphanizophyll) to investigate potential changes in N_2-fixation through time in both lakes and to evaluate the potential for temporal lags (i.e., number of observations) in the nitrogen cycling relationships within the lakes. These analyses were performed using the *ccf* function from the stats package in R (R Core Team, 2023). Additional Pearson coefficients between estimates of lake area and biogeochemical proxies (pigment concentrations, TP, C%, N%, C:N, $\delta^{13}C$, $\delta^{15}N$) were also calculated for both lakes to evaluate the presence of relationships between fluctuations in water level, biological production, and sediment geochemistry.

In all analyses, the *tidyr* v. 1.3.0 and *dplyr* v. 1.0.10 packages (Wickham, François, et al., 2023; Wickham, Vaughan, et al., 2023) were used for data wrangling, while plots were created using the *ggplot2* v. 3.4.1 and *cowplot* v. 1.1.1 packages (Wickham, Chang, et al., 2023; Wilke, 2020).

3 | RESULTS

3.1 | Lake area

Remote-sensing analysis revealed major changes in the surface area (and consequently depth) of Pelican Lake since 1972, but virtually no change in lake area in Buffalo Pound Lake over the same period (Figures 2 and S5). Pelican Lake exhibited an area of *c.* 1 km^2 in 1973, before lake area rose to *c.* 22 km^2 by 1977, followed by a decline to an area of <1 km^2 in 1981. The regional drought of the 1980s resulted in generally low lake areas in Pelican Lake until the early 1990s, although there were transient increases in area to *c.* 15 km^2 in 1982 and to *c.* 8 km^2 in 1985 (Figure 2). Land exposed during the droughts may have been used for agriculture (crop and range land), although at least some of the area would have been wetland during the drying phase (AAFC 2022; Hall, Leavitt, Quinlan, et al., 1999; IBA, 2023). Lake area rose rapidly after 1992 due to increases in regional precipitation and the conveyance of mesotrophic water from Lake Diefenbaker to Pelican Lake beginning in 1991. Pelican Lake reached a maximum surface area of *c.* 31 km^2 during the late 1990s before declining to its stable area of 25–28 km^2 since 2000 (Figure 2).

Despite the large-scale changes in lake area in Pelican Lake, the location from which the core was taken did not appear to be entirely dewatered at any point in the study interval, as noted by uninterrupted and consistent trends in biogeochemical proxies and sediment characteristics. Specifically, there was no evidence of periodic desiccation, including declines in pigment preservation, changes in ratios of labile to chemically stable compounds, rapid fluctuations in carbon content, or visible changes in sediment lithography.

3.2 | Chronology

The Buffalo Pound Lake sediment core displayed an exponential decline in ^{210}Pb activity with depth until background activity was reached at *c.* 20 cm (Figure S6). Sedimentary ^{210}Pb activities in the Pelican Lake core were much lower than those of Buffalo Pound Lake and did not display a pronounced exponential decline (Figure S6). However, background lead activities were eventually reached and the resultant CRS model age–depth extended to the late 1940s due to the low radioisotope activity (Figure 3). The CRS models of both lakes were refined using a SCAM model that extrapolated the age–depth relationship to *c.* 1900 for Pelican Lake, and improved age estimate error values in Buffalo Pound Lake. However, due to high uncertainty in the SCAM age–depth relationship of Pelican Lake, and the high error ranges of estimated dates in Buffalo Pound Lake below *c.* 17 cm despite SCAM refinement, subsequent analyses and interpretations of paleolimnological proxy data were limited those beginning at *c.* 1940 (Figure 3), the deepest accurately dated interval in both lakes. In recognition of these chronological issues, we used the age–depth models cautiously in interpreting paleolimnological time series and their response to environmental change.

3.3 | Sediment geochemistry

Fitted GAM trends of geochemical parameters revealed simple monotonic patterns in Buffalo Pound Lake since *c.* 1940, whereas contemporaneous trends were more dynamic in Pelican Lake (Figure 4). Sediment C and N content increased steadily in Buffalo

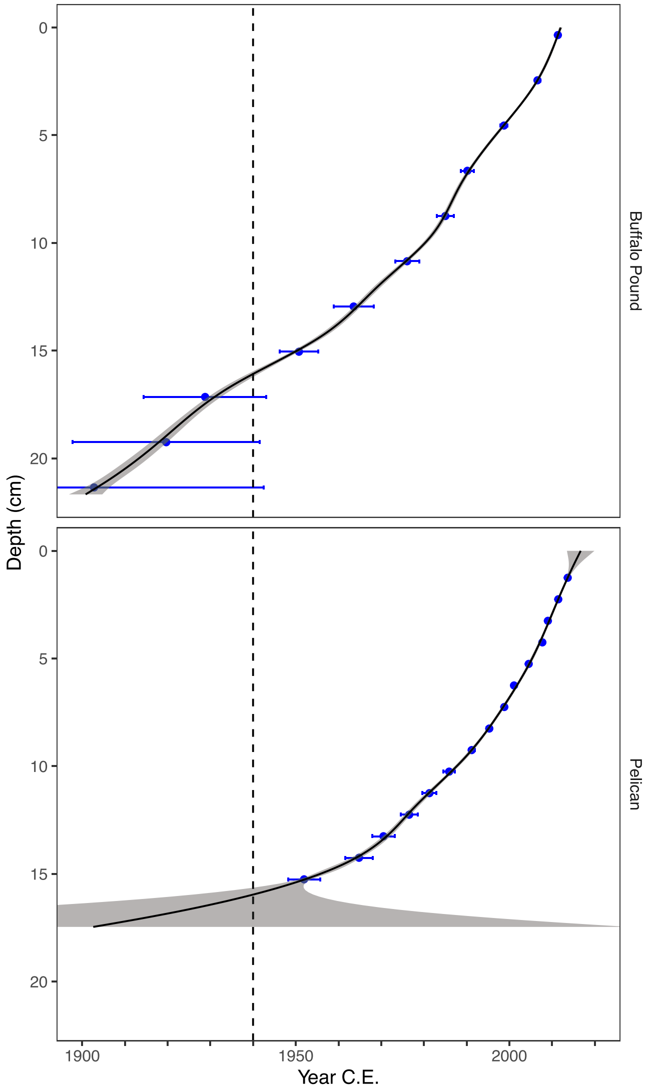

FIGURE 3 Age–depth relationships with inferred ages generated from constant rate of supply (CRS) and shape-constrained additive models (SCAMs) based on activities of ^{210}Pb from sediment cores from Buffalo Pound and Pelican lakes. Error bars are 1 *SE* of estimated ages based on ^{210}Pb activity, while the grey shading represents 95% confidence intervals. The dashed line marks 1940, which serves as the basal date for biogeochemical analyses.

Pound Lake and resulted in a generally stable C:N ratio of *c*. 11 until *c*. 2000, after which time values declined slightly due to relatively elevated N content (Figure 4). Whole-sediment δ^{13}C values showed a linear decrease from *c*. −22‰ to *c*. −25‰ over the sediment record (Figure 4). Similarly, δ^{15}N values declined steadily from *c*. 7‰ in the 1940s to *c*. 5‰ *c*. 2010 (Figure 4). Sedimentary TP concentrations in the Buffalo Pound Lake core increased slightly, rising from *c*. 1.25 to *c*. 1.5 mg P/g dry mass after about 1990 (Figure 4).

Geochemical proxies in Pelican Lake exhibited non-monotonic temporal trends. Carbon content rose from median values of *c*. 5% *c*. prior to 1950 to a maximum of *c*. 7% near 1980 before declining through the remainder of the core (Figure 4). In contrast, N content was generally stable from the late 1940s to the late 1990s, before declining through to the end of the study period (Figure 4). These trends resulted in elevated C:N ratio values compared to Buffalo Pound Lake that gently rose to maximal values of *c*. 16 during the 1980s before declining thereafter (Figure 4). Values of δ^{13}C were more enriched in Pelican Lake than in Buffalo Pound Lake with values of about −21‰ deep in the core but increasing irregularly to −19‰ by *c*. 2010 (Figure 4). Values of δ^{15}N were similarly enriched in Pelican Lake relative to Buffalo Pound, with values of *c*. 10‰ that declined slowly to *c*. 9‰ during the 1980s and then enriched to *c*. 11‰ during the 21st century (Figure 4). TP content in the sediments of Pelican Lake declined from *c*. 1.00 to 0.75 mg P/g dry mass between the late 1940s and the late 1980s, but then increased sharply to *c*. 1.25 mg P/g dry mass by about 2010 (Figure 4).

Correlation analysis revealed that C and N content (%) were both significantly ($p < 0.05$) and negatively correlated with the surface area of Pelican Lake, while δ^{15}N values were significantly positively correlated with lake area (Figure S7). However, correlations between C:N ratios and δ^{13}C values with lake area showed no significant relationship (both $p > 0.1$). Concentrations of TP were not significantly correlated with lake area in Pelican Lake ($p = 0.3$). In Buffalo Pound Lake, no geochemical parameters were significantly correlated with lake area (Figure S8).

3.4 | Fossil pigments

Fitted GAM trends of fossil pigment concentrations in Buffalo Pound Lake displayed consistent increases in all pigments from low values prior to *c*. 1960 to maximum values near 2010 (Figure 5). Historical trends in pigment concentrations were more complicated in Pelican Lake. Concentrations of siliceous algae (as fucoxanthin) declined throughout the sediment record while the primarily diatom pigment (diatoxanthin) increased substantially from the base of the core to maximum values during the 1970s, before declining to a transitory minimum *c*. 2000, and finally increasing in the most recent period (Figure 5). Concentrations of cryptophyte (alloxanthin) pigment showed little pronounced trend between *c*. 1940–1980 before declining progressively between about 1990 and 2010 (Figure 5). Concentrations of chlorophytes (pheophytin b) increased slightly from older sediments to a maximum near *c*. 1980, before declining after *c*. 1990 (Figure 5). Concentrations of lutein-zeaxanthin (bloom-forming taxa) rose steadily between the late 1940s to about 1980 and remained generally stable for the remainder of the sediment record (Figure 5). Concentrations of pigments representing total cyanobacteria (echinenone) as well as Nostocales cyanobacteria (canthaxanthin) rose from the base of the core to a maximum *c*. 1990 before declining afterwards (Figure 5), whereas concentrations of N_2-fixing cyanobacteria (aphanizophyll) were negligible in the Pelican Lake sediment core until *c*. 1970, after which concentrations increased and became more variable (Figure 5). Total phototroph production (as pheophytin a), increased between the late 1940s and about 1990 and declined thereafter, while concentrations of β-carotene displayed a similar trend but reach maximum values nearer to 1980 (Figure 5).

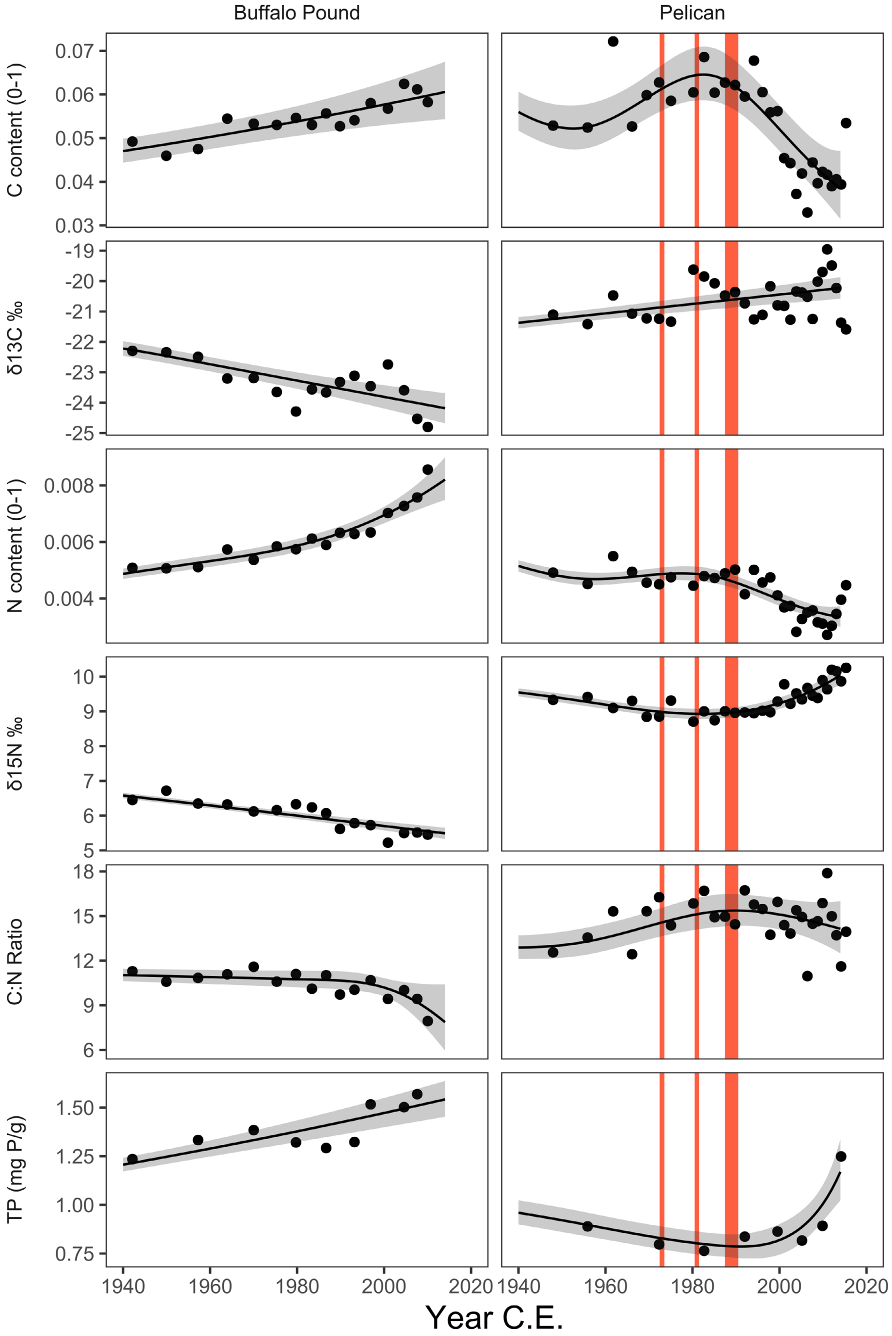

FIGURE 4 Carbon and nitrogen dry mass content (scaled 0-1.0; see methods), C:N mass ratios, $\delta^{13}C$ and $\delta^{15}N$ (‰) values, and total phosphorus concentrations (mg P/g dry mass) from Buffalo Pound and Pelican lake sediment cores plotted over time with fitted generalised additive model trends. In all plots the solid line is estimated mean, while the shading represents 95% confidence interval. Vertical bars in the Pelican Lake plots represent periods of lake surface areas <1 km^2: 1973, 1981, and 1988–1990 (see Figure 2). Uncertainty in the age–depth models (Figure 3) may mean that estimated dates may not precisely align with the actual date of low water-levels.

In general, Pelican Lake pigment concentrations were correlated negatively with lake surface area, although only the relationships with diatoms (diatoxanthin), cryptophytes (alloxanthin), bloom-forming taxa (lutein-zeaxanthin), and Nostocales (canthaxanthin) were statistically significant ($p < 0.05$; Figure S9). In Buffalo Pound Lake, correlations between lake surface area and pigment concentrations displayed consistent negative relationships, although but none were statistically significant save for that with Nostocales cyanobacteria (canthaxanthin; Figure S10).

Cross correlations between time-series of $\delta^{15}N$ values and concentrations of biomarkers from potentially N_2-fixing cyanobacteria (canthaxanthin, aphanizophyll) revealed similar, significant relationships in both Buffalo Pound and Pelican lakes (Figure 6). In Buffalo Pound Lake, concentrations of both canthaxanthin and aphanizophyll were correlated significantly and negatively with $\delta^{15}N$ values at lags of −3, although the greatest correlation occurred with no lag (Figure 6). In Pelican Lake, concentrations of canthaxanthin were also significantly correlated negatively with $\delta^{15}N$ values at −3 - 0 lag, whereas aphanizophyll levels were not correlated significantly with $\delta^{15}N$ values (Figure 6).

3.5 | Fossil Cladocera

Fossil Cladocera assemblages were composed predominantly of planktonic *Bosmina* spp. and secondarily littoral *Chydorus* spp.

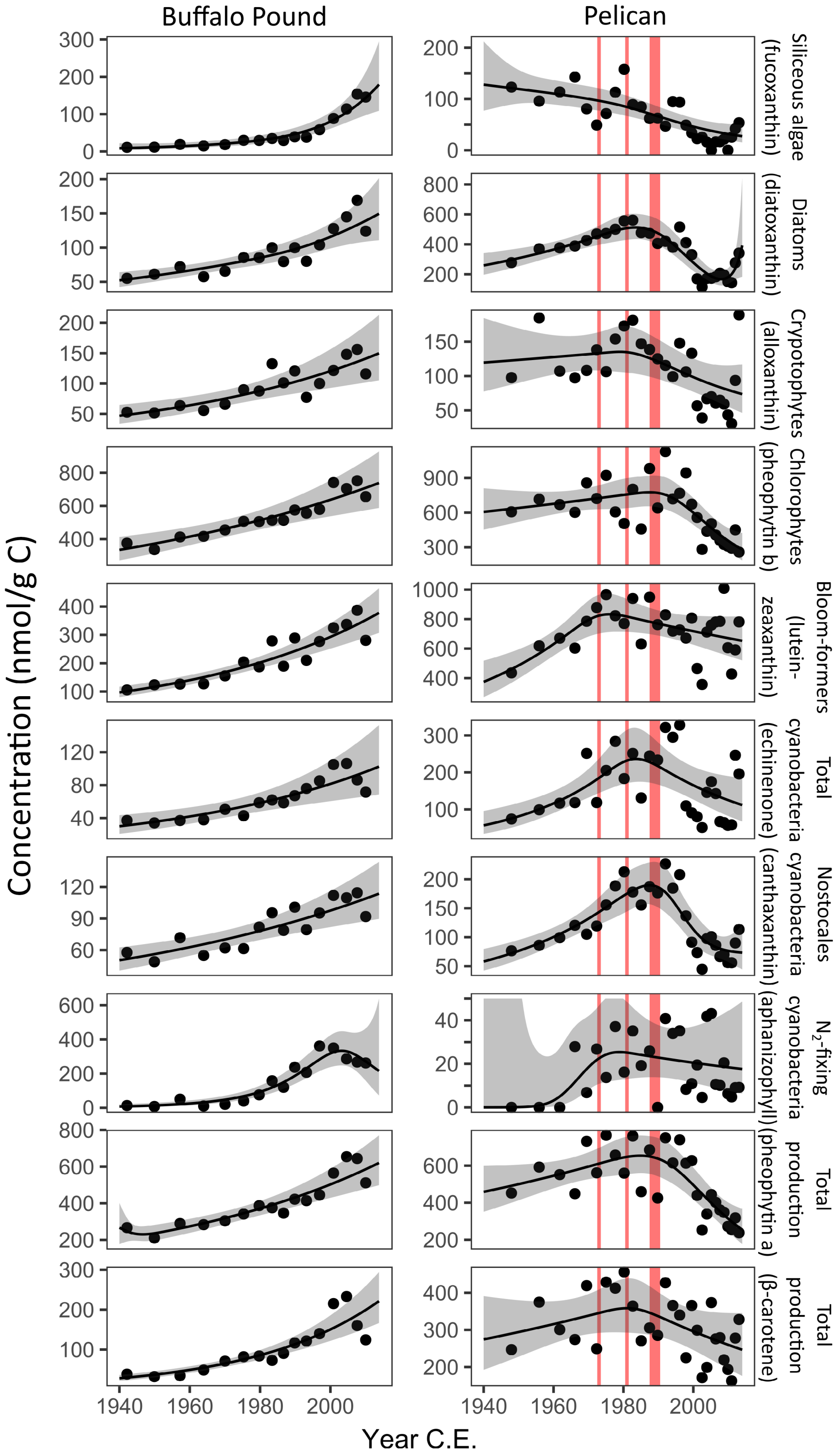

FIGURE 5 Concentrations of fossil pigments from sediment cores from Buffalo Pound and Pelican lakes over time. Pigment concentrations were fit with generalised additive models where the solid lines are estimated means and shading represents 95% confidence intervals. Vertical bars in the Pelican Lake plots represent periods of lake surface areas <1 km^2: 1973, 1981, and 1988–1990 (see Figure 2). Uncertainty in the age–depth models (Figure 3) means that estimated dates may not precisely align with the actual date of low water-levels.

remains throughout the Buffalo Pound Lake core, with greatest concentrations between the late 1940s and about 1960 (Figures S11 and S12). Sediments also contained low concentrations and relative abundances of pelagic *Daphnia* spp. Absolute concentrations of *Bosmina* spp. and *Chydorus* spp. declined over the first few decades of the cores to *c*. 1960, although their relative abundances remained high. Sediment assemblages recorded very minor increases in the abundances and concentrations of *Alona* spp. and *Daphnia* spp. during the interval between *c*. 1970 and about 2000 (Figures S11 and S12).

The invertebrate assemblage in Pelican Lake had higher overall concentrations of Cladocera than did that of Buffalo Pound Lake (Figure S13). There were elevated densities of remains from both *Chydorus* spp. and *Bosmina* spp., with lesser concentrations of fossils

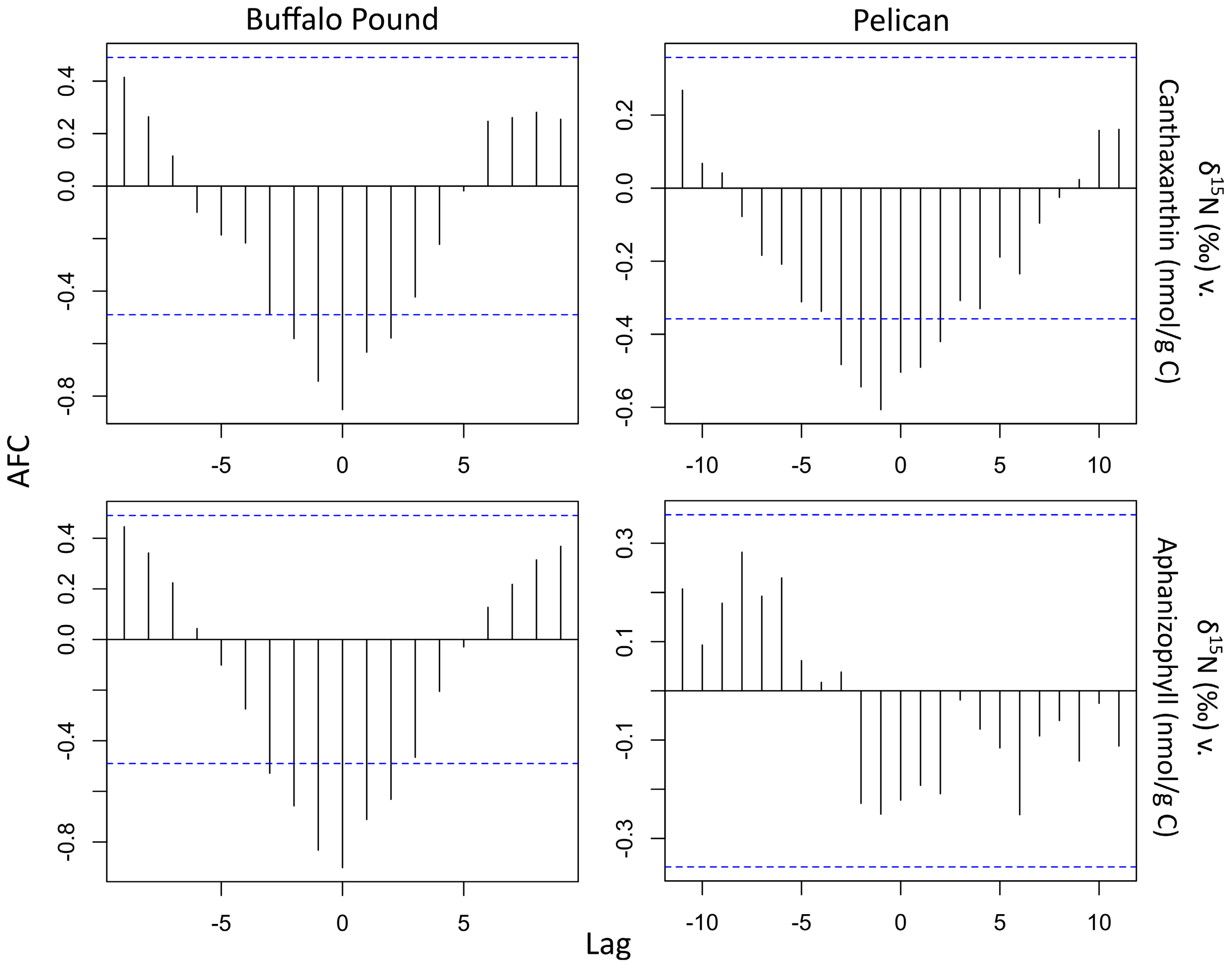

FIGURE 6 Cross correlation plots between $\delta^{N}15$ (‰) values and concentrations of pigments from possibly N_2-fixing cyanobacteria (canthaxanthin and aphanizophyll) in Buffalo Pound and Pelican lakes. Lag represents the number of observations away from the correlation. Dotted lines represent thresholds of significant correlation ($p < 0.05$). AFC represents estimated cross-correlation.

from *Daphnia* spp. during the interval from the late 1940s to about 1980 (Figure S13). In general, the relative abundance of *Bosmina* spp. was elevated in the deeper sediments from Pelican Lake, but declined after *c*. 1970, while the relative abundance of *Chydorus* spp. increased (Figures S13 and S14). *Daphnia* spp. abundance was relatively low throughout the sediment record of Pelican Lake (Figure S14). Concentrations of all cladoceran taxa declined after *c*. 1980, but remained composed mainly of *Chydorus* spp. and *Bosmina* spp. (Figure S13).

Fitted GAM trends of the *Daphnia*:*Bosmina* ratios showed somewhat different patterns between Buffalo Pound and Pelican lakes (Figure 7). In Buffalo Pound Lake, mean ratios were consistently low until *Daphnia* spp. increased *c*. 2000 and resulted in a progressive increase in ratio thereafter (Figure 7). In Pelican Lake, *Daphnia*:*Bosmina* ratios rose rapidly between the late 1940s and *c*. 2000, before plateauing through the remainder of the core due to declines in *Bosmina* spp. concentrations (Figure 7).

4 | DISCUSSION

Steady increases in C and N content, TP levels, and pigment concentrations suggest that Buffalo Pound Lake progressively eutrophied during the past 60–70 years, while less consistent trends over a similar period in Pelican Lake suggest that peak biological production in that system occurred during intervals of drought-induced water level decline. Despite differences in management strategies and eutrophication patterns, both lakes experienced marked increases in N_2-fixing and bloom-forming cyanobacteria with strong negative correlations between their pigments (canthaxanthin, aphanizophyll) and sedimentary $\delta^{15}N$ values signifying that diazotrophic taxa were important controls of overall N biogeochemistry (Hayes et al., 2019; Patoine et al., 2006). Similarly, while both lakes experienced general declines in planktonic Cladocera and exhibited high abundances of *Bosmina* spp. and *Chydorus* spp. taxa, differences in *Daphnia*:*Bosmina* ratios among sites suggest that basin-specific effects of lake-level

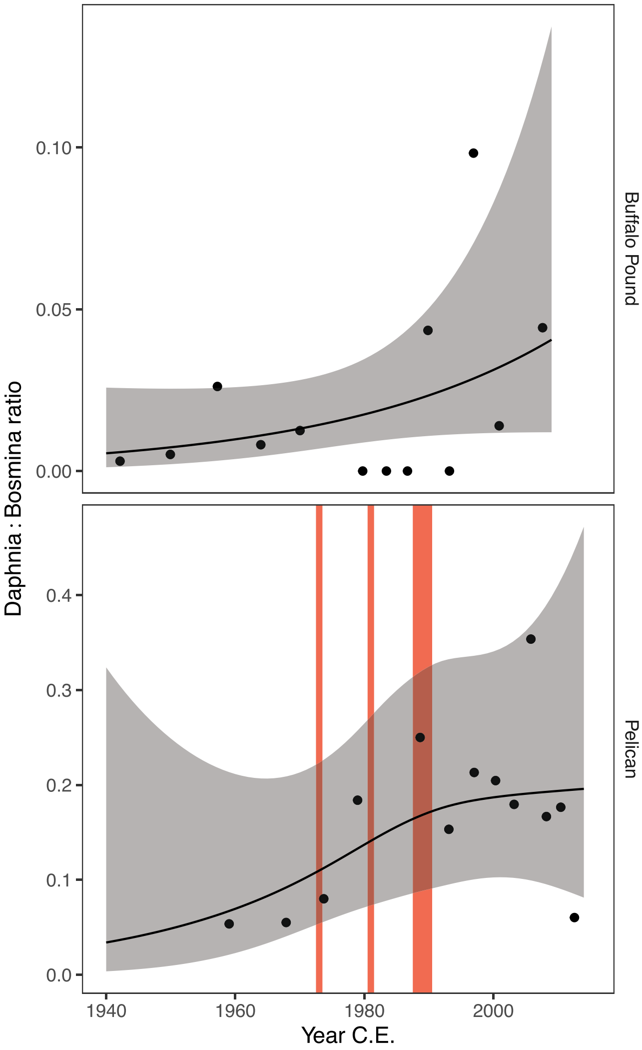

FIGURE 7 Ratio between *Daphnia* spp. and *Bosmina* spp. plotted by year from sediment cores of Buffalo Pound and Pelican lakes. Mean Cladocera ratios were estimated using generalised additive models. Solid lines represent the estimated mean ratio, while the shaded areas are 95% confidence intervals. Vertical bars in the Pelican Lake plot represent periods of lake surface area <1 km^2: 1973, 1981, and 1988–1990 (see Figure 2). Uncertainty in the age–depth models (Figure 3) means that estimated dates may not precisely align with the actual date of low water levels.

change and eutrophication may have affected in situ irradiance, macrophyte habitats, or predation regimes (Amsinck et al., 2003; Gal et al., 2013). Overall, eutrophication patterns of Buffalo Pound Lake were consistent with historical increases in agricultural development and nutrient influx from the catchment (Bunting et al., 2016; Hall, Leavitt, Dixit, et al., 1999; Hall, Leavitt, Quinlan, et al., 1999), whereas those in Pelican Lake appear to mainly reflect changes in basin hydrology prior to stabilisation of lake levels in the late 1990s (Bjorndahl et al., 2022; Vinebrooke et al., 1998). Together, these patterns suggest that the long-term hydrologic management of Buffalo Pound Lake has buffered phototroph response against the hydroclimatic variability that is clearly displayed in Pelican Lake. However, while hydrological management appeared to prevent abrupt eutrophication arising from pronounced basin dewatering, it did not prevent long-term eutrophication associated with the agricultural intensification seen since the mid-20th century in many prairie lakes (Bunting et al., 2016; Gushulak et al., 2024; Hall, Leavitt, Quinlan, et al., 1999).

4.1 | Hydrological management and eutrophication of shallow prairie lakes

The increases in fossil pigment concentrations and sedimentary C, N, and TP content, combined with declines in $\delta^{13}C$ values, are consistent with long-term effects of regional agriculture (Edlund et al., 2009; Hall, Leavitt, Dixit, et al., 1999; Hall, Leavitt, Quinlan, et al., 1999; Hecky & Hesslein, 1995), as well as findings from previous paleolimnological assessments of local (Hall, Leavitt, Dixit, et al., 1999; Hall, Leavitt, Quinlan, et al., 1999; Tse et al., 2015), regional (Dixit et al., 2000), and other eutrophied prairie lakes (Bunting et al., 2016; Maheaux et al., 2016; Pham et al., 2008). Despite experiencing similar local land use and climate histories, the differing trends in sediment geochemistry and pigment production in Pelican and Buffalo Pound lakes suggest that unique mechanisms also operate in individual lakes to regulate historical changes in primary production and water quality.

Maximum values of most pigments in Pelican Lake appear to have occurred during the 1980s, although estimates of exact year of each maximum were difficult to ascertain given the variability in underlying age–depth models. Fortunately, analysis of Landsat images revealed clearly contrasting intervals of extremes of lake area and inferred depth that could be compared with fossil trends in primary production to show that hydroclimatic changes were probably important drivers of lake eutrophication. Specifically, elevated pigment concentrations in Pelican Lake were generally concomitant with reductions in lake surface area and level arising from prolonged, multi-year, regional droughts due to variation in the Pacific/North American teleconnection pattern and the Pacific Decadal Oscillation (Gurrapu et al., 2016; Hurrell, 1995; Shabbar, 2006). Water-level declines, combined with influxes of agricultural nutrients via Thunder Creek, resulted in shallow nutrient-rich lake conditions ideal for eutrophication (da Costa et al., 2016; Özen et al., 2010; van der Kamp et al., 2008; Zhou et al., 2022). Reduced water levels in Pelican Lake may have influenced sediment release processes via impacts on internal nutrient loading and the resuspension of shallow sediments by wind, which may help sustain eutrophic conditions (Nürnberg & LaZerte, 2016; Søndergaard et al., 2013). The marked increase in diatoms (as diatoxanthin), rather than planktonic cryptophytes (alloxanthin), during periods of low lake levels also suggest that a shallower water column may have selectively favoured benthic production of epiphytic diatoms (Gushulak et al., 2021; Hofmann et al., 2020). Similarly, evaporative concentration of nutrients (da Costa et al., 2016; Özen et al., 2010) may have also promoted the growth of eutrophic chlorophytes (pheophytin b) and cyanobacteria (canthaxanthin; da Costa et al., 2016; Gushulak et al., 2021; Joshi

& Jackson, 2022) in both benthic and pelagic habitats (Gushulak et al., 2021).

Lake level changes in Pelican Lake may have also driven changes in some geochemical markers, although it is unclear whether these changes were the direct result of dewatering, altered nutrient loading, or elevated primary production. For example, sedimentary C and N content varied significantly with lake area, similar to the general relationships between lake area and pigment concentrations, and suggest that C and N content increased due to elevated phytoplankton deposition in the sediment (Bunting et al., 2016; Hall, Leavitt, Quinlan, et al., 1999) and more limited influx of allochthonous materials during droughts (Özen et al., 2010). However, as there was no relationship between lake area and $\delta^{13}C$ values, we infer that there were no marked changes in C source during periodic dewatering or reflooding intervals (Hecky & Hesslein, 1995), and rather hypothesise that variation in $\delta^{13}C$ values reflected increased primary production and deposition of depleted (organic) C. In some lakes, enriched $\delta^{13}C$ values can reflect elevated influx of terrestrial C (Gu et al., 2006; Woodward et al., 2012), as well as uptake of atmospheric CO_2 during photosynthesis (Tranvik et al., 2009), whereas in other basins, elevated production can increase deposition of depleted C sources due to use of respired CO_2 (−25 to −30‰) by phototrophs (Bunting et al., 2016; Gushulak et al., 2024). The absence of major changes in $\delta^{13}C$ values in Pelican Lake suggest that despite substantial variability in lake area, inferred basin depth, and primary production, roughly the same proportions of terrestrial and aquatic C were deposited in the sediments through time. In general, overall values of $\delta^{13}C$ of (*c.* −20‰) in Pelican Lake are consistent with mix of phytoplankton biomass (−25 to −30‰), terrestrial C sources (−15 to −20‰), and probably some degree of inorganic C source (e.g., carbonate in the substrate; −5 to −10‰) (Hecky & Hesslein, 1995; Valero-Garcés et al., 1997). Finally, lack of significant relationship between lake size and sedimentary TP concentration suggests that lake area and depth were not substantial controls of P biogeochemistry, although confirmation of this interpretation requires a more sophisticated analysis of sedimentary P fractions than was conducted herein (e.g., Bunting et al., 2016).

Substantial declines in concentrations of all cladoceran taxa, combined with increased relative abundance of macrophyte-associated *Chydorus* spp. after *c.* 1980, suggest the presence of major ecological changes in Pelican Lake during the past 5 decades. Sharp declines in lake area during the 1980s would have removed most of the pelagic habitat for Cladocera, while the elevated relative abundance of *Chydorus* spp. suggests an increase in relative importance of macrophyte beds during this interval (Amsinck et al., 2003; Cheng et al., 2020, 2021; Hann et al., 1994; Lauridsen et al., 1996). Further, because maximum Cladocera concentrations in Pelican Lake preceded the period of greatest phototrophic abundance, we suggest that changes in invertebrate concentrations did not arise from a bottom-up stimulation of secondary production, as would occur during an abrupt shift from macrophyte-rich to turbid states (Hobbs et al., 2012; Ramstack Hobbs et al., 2016), but instead probably reflect habitat restriction due to lake-level declines (Stephen et al., 1998). Consequently, we infer that the increase in the *Daphnia*:*Bosmina* ratio over the period of lake-level declines reflect the proportionately greater loss of *Bosmina* spp. due to this habitat transition, rather than an undocumented change in zooplanktivory by visually orienting fish (Leavitt et al., 1989). In contrast, increased *Daphnia*:*Bosmina* ratios in Buffalo Pound Lake around 2000 are probably due to long-term declines in macrophyte abundance, as inferred by reduced densities of littoral *Chydorus* spp. (Lauridsen et al., 1996), and/or declining lake transparency as the lake eutrophied (Estlander et al., 2009). Despite these changes, the cladoceran assemblage of Buffalo Pound Lake remained composed largely of *Bosmina* spp., suggesting an abundance of visually orienting fishes that prey selectively on the larger *Daphnia* (Leavitt et al., 1989; Amsinck et al., 2003).

While findings from Pelican Lake suggest that hydroclimate-driven declines in lake level can result in water quality degradation, biogeochemical trends in Buffalo Pound Lake show that effective hydrological management of the basin's water levels did not prevent eutrophication. In both lakes, eutrophication probably resulted from agricultural intensification in the respective catchments (Hall, Leavitt, Dixit, et al., 1999; Hall, Leavitt, Quinlan, et al., 1999; Hayes et al., 2019), while the shallow nature of both basins is expected to favour rapid warming of the water column, enhanced internal nutrient loading (Zhou et al., 2022), and elevated blooms of toxic cyanobacteria (Hayes et al., 2019; Swarbrick et al., 2019; Painter et al., 2022). Eutrophication of Pelican Lake also appeared to be particularly intense during periods diminished lake area, whereas lake level was not a significant factor controlling primary production in Buffalo Pound Lake (Hall, Leavitt, Dixit, et al., 1999; Hall, Leavitt, Quinlan, et al., 1999; Hayes et al., 2019). In contrast, nearby Lake Diefenbaker, although a comparatively young reservoir, has experienced much less eutrophication than Buffalo Pound during the past 60 years (Hall, Leavitt, Dixit, et al., 1999; Tse et al., 2015), probably due to the continuous inflow of mesotrophic waters, limited catchment run-off, and a much larger size (394 km^2) and depth (mean depth = 21.6 m; max depth = 59 m) (Tse et al., 2015; Vogt et al., 2018). Taken together, these observations imply that while hydrologic management of Buffalo Pound Lake has stabilised lake level and improved water security, current strategies have not prevented excessive growth of algae and toxic cyanobacteria in this drinking water reservoir. We suggest that catchment-scale hydrological and nutrient modelling may be needed to evaluate whether current water quality in Buffalo Pound Lake is significantly better than that which would have occurred in the absence of lake-level regulation.

4.2 | Nitrogen-fixation during eutrophication

Despite clear evidence of eutrophication in both basins, patterns of sedimentary $\delta^{15}N$ did not show pronounced enrichment typical in prairie lakes impacted by agricultural run-off or urban wastewater (Leavitt et al., 2006; Bunting et al., 2016). The lack of $\delta^{15}N$ enrichment in Buffalo Pound and Pelican lakes may reflect the observation that most water-column N is derived from lake sediments in

Qu'Appelle lakes (Patoine et al., 2006; Leavitt et al., 2006), despite strong positive correlations between N influx and planktonic primary production in decades of bioassay studies (Swarbrick et al., 2019). Instead, both lakes exhibited $\delta^{15}N$ depletion during periods of elevated densities of N_2-fixing cyanobacteria (as fossil canthaxanthin, aphanizophyll) consistent with control of planktonic N biogeochemistry by diazotrophic cyanobacteria (Patoine et al., 2006; Hayes et al., 2019). Specifically, analysis showed that sedimentary $\delta^{15}N$ and fossil cyanobacterial biomarkers exhibited consistent, substantial, and inverse correlations, particularly with canthaxanthin from Nostocales cyanobacteria. While speculative, we infer the negative correlation between aphanizophyll (N_2-fixing cyanobacteria) and $\delta^{15}N$ observed only in Buffalo Pound Lake may reflect either differences among basins in the species composition and pigmentation of the phytoplankton (Kehoe et al., 2015) or variability in preservation of aphanizophyll, an oxygen-rich carotenoid known to be more labile than canthaxanthin (Leavitt, 1993; Leavitt & Hodgson, 2001). Taken together, these patterns suggest that controls of N-cycling should be further investigated in response to changes in lake level and production in shallow prairie ecosystems (Beversdorf et al., 2013; Hayes et al., 2019).

This study shows that rates of N_2-fixation increased in both Buffalo Pound and Pelican lakes due to the eutrophication, irrespective of the precise degree of hydrological management. While concentrations of potentially N_2-fixing cyanobacteria declined (canthaxanthin) or plateaued (aphanizophyll) in Pelican Lake, canthaxanthin continues to increase in Buffalo Pound Lake, suggesting that blooms of N_2-fixing Nostocales cyanobacteria (e.g., *Dolichospermum*, *Aphanizomenon*) are likely to continue (Hayes et al., 2019; Painter et al., 2022). Because some species of these taxa are toxigenic, increases in these blooms could pose threats to human health (Beversdorf et al., 2013; Painter et al., 2022), particularly as the urban centres of Moose Jaw and Regina use Buffalo Pound Lake for their drinking water.

5 | CONCLUSIONS

Hydrologic management of Buffalo Pound Lake prevented climatically induced fluctuations in lake surface area and water level but did not prevent eutrophication of the basin. The eutrophication of Buffalo Pound Lake appears to be due to long-term inputs of terrestrial nutrients due to agricultural activity in the catchment (Hall, Leavitt, Dixit, et al., 1999; Hall, Leavitt, Quinlan, et al., 1999), as well as increased water residence times due to impoundment (Terry et al., 2022). In contrast, Pelican Lake experienced eutrophication associated with lake-level declines during periods of drought (Vinebrooke et al., 1998; Hall, Leavitt, Dixit, et al., 1999; Hall, Leavitt, Quinlan, et al., 1999; Tse et al., 2015; Bjorndahl et al., 2022), even though the relatively modest chronological models used herein prevented us precisely identifying all causal mechanisms (e.g., habitat changes, livestock, individual crops). Regardless, in both lakes, eutrophication was clearly associated with an increase in N_2-fixation by potentially toxic cyanobacteria, which pose substantial health risks to livestock and human populations (Patoine et al., 2006; Hayes et al., 2019).

This study showed that while hydrological management can buffer against the impacts of hydroclimate-induced eutrophication, naturally productive prairie lakes remain susceptible to cultural eutrophication, particularly in heavily agricultural catchments (Hambright et al., 2008; Gushulak et al., 2024). Further, with increased hydrologic stress expected across the Prairies due to atmospheric warming and declining snow accumulation (Asong et al., 2016; Johnson & Poiani, 2016), regional water bodies will probably experience future declines in lake level (van der Kamp et al., 2008). These changes may impair water quality in lakes lacking hydrological management (Bjorndahl et al., 2022), while intensively managed sites may require additional water resources to offset losses through evapotranspiration and the decline in water quality and security (Painter et al., 2022; Terry et al., 2022). Finally, this study highlights that improved management strategies beyond elevated water supply may be needed to protect drinking water in reservoirs located in continental interiors or agriculturally impacts regions.

AUTHOR CONTRIBUTIONS

Conceptualisation: C.G., J.W., H.B., K.F., P.R.L. Developing methods: H.B., P.R.L. Conducting the research: C.G., A.C., J.W., K.G., S.M., B.W., B.H. Data analysis: C.G., A.C., J.W., K.G., S.M. Data interpretation: C.G., J.W., K.G., P.R.L. Preparation of figure and tables: C.G., A.C., J.W., K.G., S.M. Writing: all authors.

ACKNOWLEDGEMENTS

The authors thank Zoraida Quiñones-Rivera for assistance with HPLC analysis and the University of Minnesota for sediment phosphorus analysis. This project was funded by Natural Science and Engineering Research Council (NSERC) of Canada, the Canadian Foundation for Innovation, Canada Research Chair program, and the Province of Saskatchewan. Data in this study was obtained from samples collected from part of Treaty 4 territory, the traditional territories of Nêhiyawak, Anihšināpēk, Dakota, Lakota, and Nakoda First Nations, and the homeland of the Métis Nation.

CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

DATA AVAILABILITY STATEMENT

Original data and code for analyses is available on GitHub at https://github.com/StefanoMezzini/buffalo-pelican.

ORCID

Cale A. C. Gushulak https://orcid.org/0000-0001-9780-3886
Amir M. Chegoonian https://orcid.org/0000-0003-0862-2763
Jared Wolfe https://orcid.org/0000-0001-7623-8929
Kristen Gray https://orcid.org/0000-0002-1962-189X
Stefano Mezzini https://orcid.org/0000-0001-8551-7436
Bjoern Wissel https://orcid.org/0000-0001-6312-149X

Brenda Hann https://orcid.org/0000-0002-5228-7846
Helen M. Baulch https://orcid.org/0000-0001-9018-4998
Kerri Finlay https://orcid.org/0000-0001-6835-8832
Peter R. Leavitt https://orcid.org/0000-0001-9805-9307

REFERENCES

AAFC (Agriculture and Agri-Food Canada). (2022). *Annual crop inventory*. https://open.canada.ca/data/en/dataset/ee44d2d4-f887-47d1-9a5b-4c9c7b43bb46

Akinremi, O. O., McGinn, S. M., & Cutforth, H. W. (1999). Precipitation trends on the Canadian Prairies. *Journal of Climate*, *12*, 2996–3003. https://doi.org/10.1175/1520-0442(1999)012<2996:PTOTCP>2.0.CO;2

Amsinck, S. L., Jeppesen, E., & Ryves, D. (2003). Cladoceran stratigraphy in two shallow brackish lakes with special reference to changes in salinity, macrophyte abundance and fish predation. *Journal of Paleolimnology*, *29*, 495–507. https://doi.org/10.1023/A:1024470329314

Appleby, P. G., Nolan, P. J., Gifford, D. W., Godfrey, M. J., Oldfield, F., Anderson, N. J., & Battarbee, R. W. (1986). 210Pb dating by low background gamma counting. *Hydrobiologia*, *143*, 21–27. https://doi.org/10.1007/BF00026640

Asong, Z. E., Khaliq, M. N., & Wheater, H. S. (2016). Projected changes in precipitation and temperature over the Canadian prairie provinces using the generalized linear model statistical downscaling approach. *Journal of Hydrology*, *539*, 429–446. https://doi.org/10.1016/j.jhydrol.2016.05.044

Beversdorf, L. J., Miller, T. R., & McMahon, K. D. (2013). The role of nitrogen fixation in cyanobacterial bloom toxicity in a temperate, Eutrophic Lake. *PLoS One*, *8*, e56103. https://doi.org/10.1371/journal.pone.0056103

Binford, M. W. (1990). Calculation and uncertainty analysis of 210 Pb dates for PIRLA project lake sediment cores. *Journal of Paleolimnology*, *3*, 253–267. https://doi.org/10.1007/BF00219461

Bjorndahl, J. A., Gushulak, C. A. C., Mezzini, S., Simpson, G. L., Haig, H. A., Leavitt, P. R., & Finlay, K. (2022). Abrupt changes in the physical and biological structure of endorheic upland lakes due to 8-m lake-level variation during the 20th century. *Limnology and Oceanography*, *67*, 1022–1039. https://doi.org/10.1002/lno.12054

Bonsal, B. R., Aider, R., Gachon, P., & Lapp, S. (2013). An assessment of Canadian prairie drought: Past, present, and future. *Climate Dynamics*, *41*, 501–516. https://doi.org/10.1007/s00382-012-1422-0

Bryson, R. A., & Hare, F. K. (1974). *Climates of North America*. Elsevier.

Bunting, L., Leavitt, P. R., Simpson, G. L., Wissel, B., Laird, K. R., Cumming, B. F., St.Amand, A., & Engstrom, D. R. (2016). Increased variability and sudden ecosystem state change in Lake Winnipeg, Canada, caused by 20th century agriculture. *Limnology and Oceanography*, *61*, 2090–2107. https://doi.org/10.1002/lno.10355

Cheng, L., Xue, B., Zawisza, E., Liu, J., Yao, S., & Li, S. (2021). Specific species response of Cladocera to the trophic and hydrological environments of lakes: A case study of a typical shallow mesotrophic lake. *Catena*, *207*, 105630. https://doi.org/10.1016/j.catena.2021.105630

Cheng, L., Xue, B., Zawisza, E., Yao, S., Liu, J., & Li, L. (2020). Effects of environmental change on subfossil Cladocera in the subtropical shallow freshwater east Taihu Lake, China. *Catena*, *188*, 104446. https://doi.org/10.1016/j.catena.2019.104446

Coops, H., Beklioglu, M., & Crisman, T. L. (2003). The role of water-level fluctuations in shallow lake ecosystems – Workshop conclusions. *Hydrobiologia*, *506*, 23–27. https://doi.org/10.1023/B:HYDR.0000008595.14393.77

da Costa, M. R. A., Attayde, J. L., & Becker, V. (2016). Effects of water level reduction on the dynamics of phytoplankton functional groups in tropical semi-arid shallow lakes. *Hydrobiologia*, *778*, 75–89. https://doi.org/10.1007/s10750-015-2593-6

Dibike, Y., Prowse, T., Bonsal, B., & O'Neil, H. (2017). Implications of future climate on water availability in the western Canadian river basins. *International Journal of Climatology*, *37*, 3247–3263. https://doi.org/10.1002/joc.4912

Dibike, Y., Prowse, T., Shrestha, R., & Ahmed, R. (2012). Observed trends and future projections of precipitation and air temperature in the Lake Winnipeg watershed. *Journal of Great Lakes Research*, *38*, 72–82. https://doi.org/10.1016/j.jglr.2011.04.005

Dixit, A. S., Hall, R. I., Leavitt, P. R., Quinlan, R., & Smol, J. P. (2000). Effects of sequential depositional basins on lake response to urban and agricultural pollution: A palaeoecological analysis of the Qu'Appelle Valley, Saskatchewan, Canada. *Freshwater Biology*, *43*, 319–337. https://doi.org/10.1046/j.1365-2427.2000.00516.x

Dumanski, S., Pomeroy, J. W., & Westbrook, C. J. (2015). Hydrological regime changes in a Canadian prairie basin. *Hydrological Processes*, *29*, 3893–3904. https://doi.org/10.1002/hyp.10567

Edlund, M. B., Engstrom, D. R., Triplett, L. D., Lafrancois, B. M., & Leavitt, P. R. (2009). Twentieth century eutrophication of the St. Croix River (Minnesota–Wisconsin, USA) reconstructed from the sediments of its natural impoundment. *Journal of Paleolimnology*, *41*, 641–657. https://doi.org/10.1007/s10933-008-9296-1

Engstrom, D. R., & Wright, H. E., Jr. (1984). Chemical stratigraphy of lake sediments as a record of environmental change. In E. Y. Haworth & J. W. G. Lund (Eds.), *Lake sediments and environmental history* (pp. 11–67). University of Minnesota Press.

Environment and Climate Change Canada (ECCC). (2023a). *Canadian climate normals 1981–2020 station data*. Moose Jaw, Saskatchewan. https://climate.weather.gc.ca/climate_normals/results_1981_2010_e.html?searchType=stnName&txtStationName=moose&searchMethod=contains&txtCentralLatMin=0&txtCentralLatSec=0&txtCentralLongMin=0&txtCentralLongSec=0&stnID=2967&dispBack=0

Environment and Climate Change Canada (ECCC). (2023b). *Homogenized Canadian climate station data*. Moose Jaw, Saskatchewan. https://www.canada.ca/en/environment-climate-change/services/climate-change/science-research-data/climate-trends-variability/adjusted-homogenized-canadian-data.html

Estlander, S., Nurminen, L., Olin, M., Vinni, M., & Horppila, J. (2009). Seasonal fluctuations in macrophyte cover and water transparency of four brown-water lakes: Implications for crustacean zooplankton in littoral and pelagic habitats. *Hydrobiologia*, *620*, 109–120. https://doi.org/10.1007/s10750-008-9621-8

Gal, G., Skerjanec, M., & Atanasova, N. (2013). Fluctuations in water level and the dynamics of zooplankton: A data-driven modelling approach. *Freshwater Biology*, *58*, 800–816. https://doi.org/10.1111/fwb.12087

Gan, T. Y. (2000). Reducing vulnerability of water resources of Canadian prairies to potential droughts and possible climatic warming. *Water Resources Management*, *14*, 111–135. https://doi.org/10.1023/A:1008195827031

Gibson, J. J., Birks, S. J., Yi, Y., Moncur, M. C., & McEachern, P. M. (2016). Stable isotope mass balance of fifty lakes in central Alberta: Assessing the role of water balance parameters in determining trophic status and lake level. *Journal of Hydrology: Regional Studies*, *6*, 13–25. https://doi.org/10.1016/j.ejrh.2016.01.034

Glew, J. R. (1989). A new trigger mechanism for sediment samplers. *Journal of Paleolimnology*, *2*, 241–243. https://doi.org/10.1007/BF00195474

Gu, B., Chapman, A. D., & Schelske, C. L. (2006). Factors controlling seasonal variations in stable isotope composition of particulate organic matter in a softwater eutrophic lake. *Limnology and*

Oceanography, *51*, 2837–2848. https://doi.org/10.4319/lo.2006.51.6.2837

Gurrapu, S., St-Jacques, J.-M., Sauchyn, D. J., & Hodder, K. R. (2016). The influence of the Pacific decadal oscillation on annual floods in the Rivers of Western Canada. *JAWRA Journal of the American Water Resources Association*, *52*, 1031–1045. https://doi.org/10.1111/1752-1688.12433

Gushulak, C. A. C., Haig, H. A., Kingsbury, M. V., Wissel, B., Cumming, B. F., & Leavitt, P. R. (2021). Effects of spatial variation in benthic phototrophs along a depth gradient on assessments of whole-lake processes. *Freshwater Biology*, *66*, 2118–2132. https://doi.org/10.1111/fwb.13820

Gushulak, C. A. C., Mezzini, S., Moir, K. E., Simpson, G. L., Bunting, L., Wissel, B., Engstrom, D. R., Laird, K. R., St.Amand, A., Cumming, B. F., & Leavitt, P. R. (2024). Differential effects of a century of land use change on eutrophication of adjacent large prairie lakes: Lakes Manitoba and Winnipeg, Canada. *Freshwater Biology*, *69*, 47–63. https://doi.org/10.1111/fwb.14192

Haig, H. A., Hayes, N. M., Simpson, G. L., Yi, Y., Wissel, B., Hodder, K. R., & Leavitt, P. R. (2020). Comparison of isotopic mass balance and instrumental techniques as estimates of basin hydrology in seven connected lakes over 12 years. *Journal of Hydrology X*, *6*, 100046. https://doi.org/10.1016/j.hydroa.2019.100046

Hall, R. I., Leavitt, P. R., Dixit, A. S., Quinlan, R., & Smol, J. P. (1999). Limnological succession in reservoirs: A paleolimnological comparison of two methods of reservoir formation. *Canadian Journal of Fisheries and Aquatic Sciences*, *56*, 1109–1121. https://doi.org/10.1139/f99-047

Hall, R. I., Leavitt, P. R., Quinlan, R., Dixit, A. S., & Smol, J. P. (1999). Effects of agriculture, urbanization, and climate on water quality in the northern Great Plains. *Limnology and Oceanography*, *44*, 739–756. https://doi.org/10.4319/lo.1999.44.3_part_2.0739

Hambright, K. D., Zohary, T., Eckert, W., Schwartz, S. S., Schelske, C. L., Laird, K. R., & Leavitt, P. R. (2008). Exploitation and destabilization of a warm, freshwater ecosystem through engineered hydrological change. *Ecological Applications*, *18*, 1591–1603. https://doi.org/10.1890/07-0931.1

Hann, B. J. (1989). *Methods in quaternary ecology #6*. Cladocera. Geoscience Canada.

Hann, B. J., Leavitt, P. R., & Chang, P. S. S. (1994). Cladocera community response to experimental eutrophication in Lake 227 as recorded in laminated sediments. *Canadian Journal of Fisheries and Aquatic Sciences*, *51*, 2312–2321. https://doi.org/10.1139/f94-234

Hassanzadeh, E., Strickert, G., Morales-Marin, L., Noble, B., Baulch, H., Shupena-Soulodre, E., & Lindenschmidt, K.-E. (2019). A framework for engaging stakeholders in water quality modeling and management: Application to the Qu'Appelle River basin, Canada. *Journal of Environmental Management*, *231*, 1117–1126. https://doi.org/10.1016/j.jenvman.2018.11.016

Hayes, N. M., Patoine, A., Haig, H. A., Simpson, G. L., Swarbrick, V. J., Wiik, E., & Leavitt, P. R. (2019). Spatial and temporal variation in nitrogen fixation and its importance to phytoplankton in phosphorus-rich lakes. *Freshwater Biology*, *64*, 269–283. https://doi.org/10.1111/fwb.13214

Hecky, R. E., & Hesslein, R. H. (1995). Contributions of benthic algae to Lake food webs as revealed by stable isotope analysis. *Journal of the North American Benthological Society*, *14*, 631–653. https://doi.org/10.2307/1467546

Hieltjes, A. H. M., & Lijklema, L. (1980). Fractionation of inorganic phosphates in calcareous sediments. *Journal of Environmental Quality*, *9*, 405–407. https://doi.org/10.2134/jeq1980.00472425000900030015x

Hobbs, W. O., Hobbs, J. M. R., LaFrançois, T., Zimmer, K. D., Theissen, K. M., Edlund, M. B., Michelutti, N., Butler, M. G., Hanson, M. A., & Carlson, T. J. (2012). A 200-year perspective on alternative stable state theory and lake management from a biomanipulated shallow lake. *Ecological Applications*, *22*, 1483–1496. https://doi.org/10.1890/11-1485.1

Hofmann, A. M., Geist, J., Nowotny, L., & Raeder, U. (2020). Depth-distribution of lake benthic diatom assemblages in relation to light availability and substrate: Implications for paleolimnological studies. *Journal of Paleolimnology*, *64*, 315–334. https://doi.org/10.1007/s10933-020-00139-9

Hosseini, N., Akomeah, E., Davis, J.-M., Baulch, H., & Lindenschmidt, K.-E. (2018). Water quality modeling of a prairie river-lake system. *Environmental Science and Pollution Research*, *25*, 31190–31204. https://doi.org/10.1007/s11356-018-3055-2

Huisman, J., Codd, G. A., Paerl, H. W., Ibelings, B. W., Verspagen, J. M. H., & Visser, P. M. (2018). Cyanobacterial blooms. *Nature Reviews Microbiology*, *16*, 471–483. https://doi.org/10.1038/s41579-018-0040-1

Hurrell, J. W. (1995). Decadal trends in the North Atlantic oscillation: Regional temperatures and precipitation. *Science*, *269*, 676–679. https://doi.org/10.1126/science.269.5224.676

IBA (Important Bird Areas) Canada. (2023). Site summary: Pelican Lake Moose Jaw, Saskatchewan. https://www.ibacanada.com/site.jsp?siteID=SK059 (June 2023)

Jane, S. F., Hansen, G. J. A., Kraemer, B. M., Leavitt, P. R., Mincer, J. L., North, R. L., Pilla, R. M., Stetler, J. T., Williamson, C. E., Woolway, R. I., Arvola, L., Chandra, S., DeGasperi, C. L., Diemer, L., Dunalska, J., Erina, O., Flaim, G., Grossart, H. P., Hambright, K. D., … Rose, K. C. (2021). Widespread deoxygenation of temperate lakes. *Nature*, *594*, 66–70. https://doi.org/10.1038/s41586-021-03550-y

Jeppesen, E., Meerhoff, M., Davidson, T. A., Trolle, D., Sondergaard, M., Lauridsen, T., Beklioğlu, M., Brucet, S., Volta, P., González-Bergonzoni, I., & Nielsen, A. (2014). Climate change impacts on lakes: An integrated ecological perspective based on a multi-faceted approach, with special focus on shallow lakes. *Journal of Limnology*, *73*, 88–111. https://doi.org/10.4081/jlimnol.2014.844

Johnson, W. C., & Poiani, K. A. (2016). Climate change effects on prairie pothole wetlands: Findings from a twenty-five year numerical modeling project. *Wetlands*, *36*, 273–285. https://doi.org/10.1007/s13157-016-0790-3

Joshi, S., & Jackson, L. (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*, 1–35. https://doi.org/10.21926/aeer.2201007

Kehoe, M., Chun, K. P., & Baulch, H. M. (2015). Who smells? Forecasting taste and odor in a drinking water reservoir. *Environmental Science and Technology*, *49*, 10984–10992. https://doi.org/10.1021/acs.est.5b00979

Lauridsen, T., Pedersen, L. J., Jeppesen, E., & Sønergaard, M. (1996). The importance of macrophyte bed size for cladoceran composition and horizontal migration in a shallow lake. *Journal of Plankton Research*, *18*, 2283–2294. https://doi.org/10.1093/plankt/18.12.2283

Leavitt, P. R. (1993). A review of factors that regulate carotenoid and chlorophyll deposition and fossil pigment abundance. *Journal of Paleolimnology*, *9*, 109–127. https://doi.org/10.1007/BF00677513

Leavitt, P. R., Brock, C. S., Ebel, C., & Patoine, A. (2006). Landscape-scale effects of urban nitrogen on a chain of freshwater lakes in central North America. *Limnology and Oceanography*, *51*, 2262–2277. https://doi.org/10.4319/lo.2006.51.5.2262

Leavitt, P. R., Carpenter, S. R., & Kitchell, J. F. (1989). Whole-lake experiments: The annual record of fossil pigments and zooplankton. *Limnology and Oceanography*, *34*, 700–717. https://doi.org/10.4319/lo.1989.34.4.0700

Leavitt, P. R., & Hodgson, D. A. (2001). Sedimentary pigments. In J. P. Smol, H. J. B. Birks, W. M. Last, R. S. Bradley, & K. Alverson (Eds.), *Tracking environmental change using Lake sediments: Terrestrial, algal, and siliceous indicators* (pp. 295–325). Springer.

Maheaux, H., Leavitt, P. R., & Jackson, L. J. (2016). Asynchronous onset of eutrophication among shallow prairie lakes of the northern Great Plains, Alberta, Canada. *Global Change Biology*, *22*, 271–283. https://doi.org/10.1111/gcb.13076

Meerhoff, M., Audet, J., Davidson, T. A., De Meester, L., Hilt, S., Kosten, S., Liu, Z., Mazzeo, N., Paerl, H., Scheffer, M., & Jeppesen, E. (2022). Feedback between climate change and eutrophication: Revisiting the allied attack concept and how to strike back. *Inland Waters*, *12*, 187–204. https://doi.org/10.1080/20442041.2022.2029317

Michels, A., Laird, K. R., Wilson, S. E., Thomson, D., Leavitt, P. R., Oglesby, R. J., & Cumming, B. F. (2007). Multidecadal to millennial-scale shifts in drought conditions on the Canadian prairies over the past six millennia: Implications for future drought assessment. *Global Change Biology*, *13*, 1295–1307. https://doi.org/10.1111/j.1365-2486.2007.01367.x

Mosley, L. M. (2015). Drought impacts on the water quality of freshwater systems; review and integration. *Earth-Science Reviews*, *140*, 203–214. https://doi.org/10.1016/j.earscirev.2014.11.010

Nkemdirim, L., & Weber, L. (1999). Comparison between the droughts of the 1930s and the 1980s in the southern Prairies of Canada. *Journal of Climate*, *12*, 2434–2450. https://doi.org/10.1175/1520-0442(1999)012<2434:CBTDOT>2.0.CO;2

North, R. L., Davies, J.-M., Doig, L. E., Lindenschmidt, K.-E., & Hudson, J. J. (2015). Lake Diefenbaker: The prairie jewel. *Journal of Great Lakes Research*, *41*, 1–7. https://doi.org/10.1016/j.jglr.2015.10.003

Nürnberg, G. K., & LaZerte, B. D. (2016). More than 20 years of estimated internal phosphorus loading in polymictic, eutrophic Lake Winnipeg, Manitoba. *Journal of Great Lakes Research*, *42*, 18–27. https://doi.org/10.1016/j.jglr.2015.11.003

Özen, A., Karapınar, B., Kucuk, İ., Jeppesen, E., & Beklioglu, M. (2010). Drought-induced changes in nutrient concentrations and retention in two shallow Mediterranean lakes subjected to different degrees of management. *Hydrobiologia*, *646*, 61–72. https://doi.org/10.1007/s10750-010-0179-x

Paerl, H. W., & Huisman, J. (2009). Climate change: A catalyst for global expansion of harmful cyanobacterial blooms. *Environmental Microbiology Reports*, *1*, 27–37. https://doi.org/10.1111/j.1758-2229.2008.00004.x

Painter, K. J., Venkiteswaran, J. J., Simon, D. F., Duy, S. V., Sauvé, S., & Baulch, H. M. (2022). Early and late cyanobacterial bloomers in a shallow, eutrophic lake. *Environmental Science: Processes & Impacts*, *24*, 1212–1227. https://doi.org/10.1039/D2EM00078D

Patoine, A., Graham, M. D., & Leavitt, P. R. (2006). Spatial variation of nitrogen fixation in lakes of the northern Great Plains. *Limnology and Oceanography*, *51*, 1665–1677. https://doi.org/10.4319/lo.2006.51.4.1665

Pham, S. V., Leavitt, P. R., McGowan, S., & Peres-Neto, P. (2008). Spatial variability of climate and land-use effects on lakes of the northern Great Plains. *Limnology and Oceanography*, *53*, 728–742. https://doi.org/10.4319/lo.2008.53.2.0728

Pham, S. V., Leavitt, P. R., McGowan, S., Wissel, B., & Wassenaar, L. I. (2009). Spatial and temporal variability of prairie lake hydrology as revealed using stable isotopes of hydrogen and oxygen. *Limnology and Oceanography*, *54*, 101–118. https://doi.org/10.4319/lo.2009.54.1.0101

Pya, N. (2023). *scam: Shape constrained additive models*. R Package Version 1.2-14. https://cran.r-project.org/web/packages/scam/index.html

Qin, B., Zhu, G., Zhang, L., Luo, L., Gao, G., & Gu, B. (2006). Estimation of internal nutrient release in large shallow Lake Taihu, China. *Science in China Series D*, *49*, 38–50. https://doi.org/10.1007/s11430-006-8104-x

R Core Team. (2023). *R: A language for statistical computing. Version 4.2.3*. The R Foundation for Statistical Computing. https://www.r-project.org/

Ramstack Hobbs, J. M., Hobbs, W. O., Edlund, M. B., Zimmer, K. D., Theissen, K. M., Hoidal, N., Domine, L. M., Hanson, M. A., Herwig, B. R., & Cotner, J. B. (2016). The legacy of large regime shifts in shallow lakes. *Ecological Applications*, *26*, 2662–2676. https://doi.org/10.1002/eap.1382

Savage, C., Leavitt, P. R., & Elmgren, R. (2004). Distribution and retention of effluent nitrogen in surface sediments of a coastal bay. *Limnology and Oceanography*, *49*, 1503–1511. https://doi.org/10.4319/lo.2004.49.5.1503

Schelske, C. L., Peplow, A., Brenner, M., & Spencer, C. N. (1994). Low-background gamma counting: Applications for210Pb dating of sediments. *Journal of Paleolimnology*, *10*, 115–128. https://doi.org/10.1007/BF00682508

Shabbar, A. (2006). The impact of El Niño-southern oscillation on the Canadian climate. *Advances in Geosciences*, *6*, 149–153. https://doi.org/10.5194/adgeo-6-149-2006

Shayo, S., & Limbu, S. M. (2018). Nutrient release from sediments and biological nitrogen fixation: Advancing our understanding of eutrophication sources in Lake Victoria, Tanzania. *Lakes & Reservoirs: Science, Policy and Management for Sustainable Use*, *23*, 312–323. https://doi.org/10.1111/lre.12242

Simpson, G. L. (2018). Modelling Palaeoecological time series using generalised additive models. *Frontiers in Ecology and Evolution*, *6*, 149. https://doi.org/10.3389/fevo.2018.00149

SIPA (Saskatchewan Irrigation Projects Association). (2008). *A time to irrigate: Volume 1. The economic, social, and environmental benefits of expanding irrigation in the Lake Diefenbaker region*. Clifton Associates LTD.

Smith, C. D., & Kells, J. A. (1993). Development and impacts of Lake Diefenbaker: A multi-purpose project. *Canadian Water Resources Journal*, *18*, 503–512. https://doi.org/10.4296/cwrj1804503

Søndergaard, M., Bjerring, R., & Jeppesen, E. (2013). Persistent internal phosphorus loading during summer in shallow eutrophic lakes. *Hydrobiologia*, *710*, 95–107. https://doi.org/10.1007/s10750-012-1091-3

Stephen, D. B., & Phillips, A. G. (1998). The relative importance of top-down and bottom-up control of phytoplankton in a shallow macrophyte-dominated lake. *Freshwater Biology*, *39*, 699–713. https://doi.org/10.1046/j.1365-2427.1998.00321.x

Suchy, K. D., Salki, A., & Hann, B. J. (2010). Investigating the invasion of the nonindigenous zooplankter, *Eubosmina coregoni*, in Lake Winnipeg, Manitoba, Canada. *Journal of Great Lakes Research*, *36*, 159–166. https://doi.org/10.1016/j.jglr.2009.12.004

Swarbrick, V. J., Simpson, G. L., Glibert, P. M., & Leavitt, P. R. (2019). Differential stimulation and suppression of phytoplankton growth by ammonium enrichment in eutrophic hardwater lakes over 16 years. *Limnology and Oceanography*, *64*, S130–S149. https://doi.org/10.1002/lno.11093

Tanzeeba, S., & Gan, T. Y. (2012). Potential impact of climate change on the water availability of South Saskatchewan River basin. *Climatic Change*, *112*, 355–386. https://doi.org/10.1007/s10584-011-0221-7

Taranu, Z. E., Gregory-Eaves, I., Leavitt, P. R., Bunting, L., Buchaca, T., Catalan, J., Domaizon, I., Guilizzoni, P., Lami, A., McGowan, S., & Moorhouse, H. (2015). Acceleration of cyanobacterial dominance in north temperate-subarctic lakes during the Anthropocene. *Ecology Letters*, *18*, 375–384. https://doi.org/10.1111/ele.12420

Terry, J., Davies, J.-M., & Lindenschmidt, K.-E. (2022). Buffalo Pound Lake—Modelling water resource management scenarios of a large multi-purpose prairie reservoir. *Water*, *14*, 584. https://doi.org/10.3390/w14040584

Tranvik, L. J., Downing, J. A., Cotner, J. B., Loiselle, S. A., Striegl, R. G., Ballatore, T. J., Dillon, P., Finlay, K., Fortino, K., Knoll, L. B., Kortelainen, P. L., Kutser, T., Larsen, S., Laurion, I., Leech, D. M., McCallister, S. L., McKnight, D. M., Melack, J. M., Overholt, E., … Weyhenmeyer, G. A. (2009). Lakes and reservoirs as regulators of carbon cycling and climate. *Limnology and Oceanography*, *54*, 2298–2314. https://doi.org/10.4319/lo.2009.54.6_part_2.2298

Tse, T. J., Doig, L. E., Leavitt, P. R., Quiñones-Rivera, Z. J., Codling, G., Lucas, B. T., Liber, K., Giesy, J. P., Wheater, H., & Jones, P. D. (2015). Long-term spatial trends in sedimentary algal pigments in a narrow river-valley reservoir, Lake Diefenbaker, Canada. *Journal of Great Lakes Research*, *41*, 56–66. https://doi.org/10.1016/j.jglr.2015.08.002

USGS (U.S. Geological Survey). (2023). *Landsat collection-2 Level-1*. https://earthexplorer.usgs.gov/

Valero-Garcés, B. L., Laird, K. R., Fritz, S. C., Kelts, K., Ito, E., & Grimm, E. C. (1997). Holocene climate in the northern Great Plains inferred from sediment stratigraphy, stable isotopes, carbonate geochemistry, diatoms, and pollen at moon Lake, North Dakota. *Quaternary Research*, *48*, 359–369. https://doi.org/10.1006/qres.1997.1930

van der Kamp, G., Keir, D., & Evans, M. S. (2008). Long-term water level changes in closed-basin lakes of the Canadian Prairies. *Canadian Water Resources Journal*, *33*, 23–38. https://doi.org/10.4296/cwrj3301023

Vinebrooke, R. D., Hall, R. I., Leavitt, P. R., & Cumming, B. F. (1998). Fossil pigments as indicators of phototrophic response to salinity and climatic change in lakes of western Canada. *Canadian Journal of Fisheries and Aquatic Sciences*, *55*, 668–681. https://doi.org/10.1139/f97-263

Vogt, R. J., Sharma, S., & Leavitt, P. R. (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. https://doi.org/10.1139/cjfas-2016-0520

Wheater, H., & Gober, P. (2013). Water security in the Canadian Prairies: Science and management challenges. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, *371*, 20120409. https://doi.org/10.1098/rsta.2012.0409

Wickham, H., Chang, W., Henry, L., Takahashi, K., Wilke, C., Woo, K., Yutani, H., Dunnington, D., & van den Brand, T. (2023). *ggplot2: Create elegant data Visualisations using the grammar of graphics*. R Package Version 3.4.2. https://cran.r-project.org/web/packages/ggplot2/index.html

Wickham, H., François, R., Henry, L., Müller, K., Vaughan, D., & Posit Software, PBC. (2023). *dplyr: A grammar of data manipulation*. R Package Version 1.1.2. https://cran.r-project.org/web/packages/dplyr/index.html

Wickham, H., Vaughan, D., Girlich, M., Ushey, K., & Posit Software, PBC. (2023). *tidyr: Tidy messy data*. R Package Version 1.3.0. https://cran.r-project.org/web/packages/tidyr/index.html

Wilke, C. O. (2020). *cowplot: Streamlined Plot Theme and Plot Annotations for "ggplot2."* R Package Version 1.1.1. https://cran.r-project.org/web/packages/cowplot/cowplot.pdf

Wood, S. (2023). *mgcv: Mixed GAM computation vehicle with automatic smoothness estimation*. R Package Version 1.9-0. https://cran.r-project.org/web/packages/mgcv/index.html

Wood, S. N. (2011). Fast stable restricted maximum likelihood and marginal likelihood estimation of semiparametric generalized linear models. *Journal of the Royal Statistical Society, Series B: Statistical Methodology*, *73*, 3–36. https://doi.org/10.1111/j.1467-9868.2010.00749.x

Wood, S. N. (2017). *Generalized additive models: An introduction with R*. CRC Press.

Wood, S. N., Pya, N., & Säfken, B. (2016). Smoothing parameter and model selection for general smooth models. *Journal of the American Statistical Association*, *111*, 1548–1563. https://doi.org/10.1080/01621459.2016.1180986

Woodward, C. A., Potito, A. P., & Beilman, D. W. (2012). Carbon and nitrogen stable isotope ratios in surface sediments from lakes of western Ireland: Implications for inferring past lake productivity and nitrogen loading. *Journal of Paleolimnology*, *47*, 167–184. https://doi.org/10.1007/s10933-011-9568-z

Zanaga, D., Van De Kerchove, R., Daems, D., De Keersmaecker, W., Brockmann, C., Kirches, G., Wevers, J., Cartus, O., Santoro, M., Fritz, S., Lesiv, M., Herold, M., Tsendbazar, N. E., Xu, P., Ramoino, F., & Arino, O. (2022). *ESA world cover 10 m 2021 v200*. https://doi.org/10.5281/zenodo.7254221

Zhou, J., Leavitt, P. R., Zhang, Y., & Qin, B. (2022). Anthropogenic eutrophication of shallow lakes: Is it occasional? *Water Research*, *221*, 118728. https://doi.org/10.1016/j.watres.2022.118728

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

How to cite this article: Gushulak, C. A. C., Chegoonian, A. M., Wolfe, J., Gray, K., Mezzini, S., Wissel, B., Hann, B., Baulch, H. M., Finlay, K., & Leavitt, P. R. (2024). Impacts of hydrologic management on the eutrophication of shallow lakes in an intensive agricultural landscape (Saskatchewan, Canada). *Freshwater Biology*, *69*, 984–1000. https://doi.org/10.1111/fwb.14260