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Paleolimnological assessment of past hydro-ecological variation at a shallow hardwater lake in the Athabasca Oil Sands Region before potential onset of industrial development

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ABSTRACT

Study region: McClelland Lake, Athabasca Oil Sands Region
Study focus: Effective environmental monitoring requires knowledge of inherent natural variation. In the absence of pre-development monitoring of aquatic ecosystems, paleolimnological approaches have been championed as a scientifically rigorous method to define pre-development conditions. Motivated by regulatory processes and absence of pre-development data, we conducted a comprehensive paleolimnological study at McClelland Lake to determine an appropriate timeframe for defining natural ranges of variation (NRVs) in hydroecological variables before potential onset of mining within its catchment.
New hydrological insights for the region: During the past ~325 years, five distinctive intervals of hydroecological conditions were identified. The first phase (ca. 1695–1750) coincided with the Little Ice Age (LIA), when arid conditions supported lake levels 2.6–3.5 m below present. Phase II (ca. 1750–1840) encompassed subsequent warming, lake-level rise to 1.2–2.6 m below present and increased aquatic productivity. Phase III included frequent natural disturbance by wildfires (ca. 1840–1900). During Phase IV (ca. 1900–1970), the lake deepened and algal communities diversified. Phase V (post–1970) captured influence of regional industrial development, climate warming and lake-level decline, and wildfires. We propose quantitative definitions of NRVs for McClelland Lake be derived from paleolimnological indicators since 1750, which provide a conservative and relevant range of hydroecological conditions, and explore merits and drawbacks of shorter-duration NRV definition for monitoring change.

1. Introduction

Environmental monitoring programs are designed to detect if, and when, deleterious changes occur to aquatic ecosystems because

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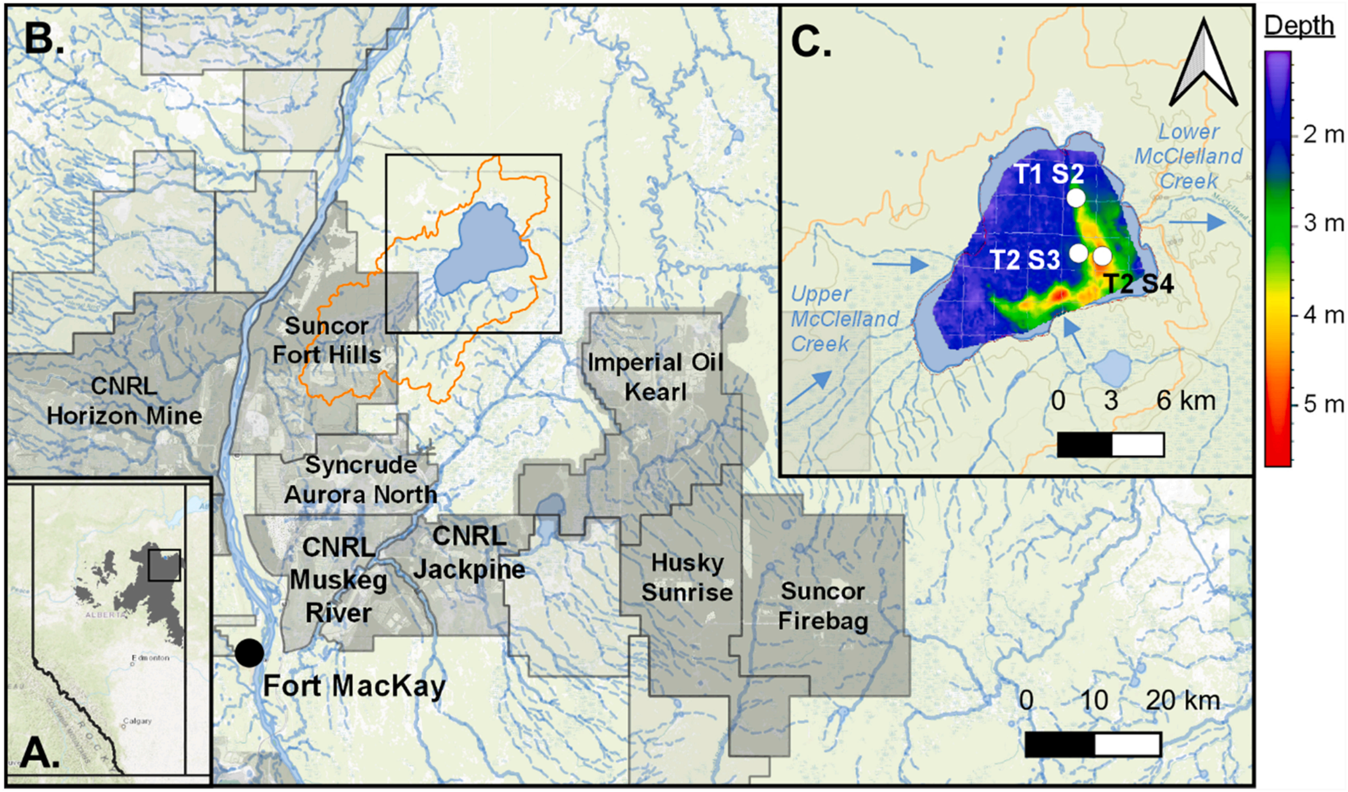

Fig. 1. (A) Location of McClelland Lake within the oil sands deposits (dark grey) of northern Alberta; (B) The McClelland Lake watershed (orange outline), with local active oil sands projects (medium grey fill) and inactive/prospective projects (light grey outline) indicated; and (C) McClelland Lake, including lake depth (colour gradient), inflows to the west and south, and the outflow to the east (blue lines). Locations of the three sediment coring sites reported in this paper are indicated with white circles. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

of human activities. Effective monitoring frameworks for major industrial projects hinge on comparing measurements on samples collected during and after the project has begun to pre-development conditions to detect departures from 'natural' or reference conditions (Arciszewski et al., 2017b). However, long-term monitoring programs often are not implemented until after a development has begun or a potential anthropogenic effect has been observed in the environment. In these situations, the characterization of 'pre-development' conditions of aquatic ecosystems is poor or missing, which challenges the capability to detect industry-driven change (Lindenmayer and Likens, 2010; Smol, 2008). The use of reference sites, i.e., other ecosystems assumed to be similar to the hypothetical pre-development or 'normal' conditions, can strengthen monitoring programs, but can be problematic as reference sites are assumed to mimic the focal ecosystem through time, including interannual natural variability; yet this assumption may not always hold true (see Arciszewski and Munkittrick, 2015). In Canada, current regulatory requirements frequently require or necessitate the collection of pre-development data. Such processes include the Impact Assessment process under the Canadian Impact Assessment Act (Impact Assessment Act IAA, 2019) and Environmental Assessment Program in Alberta (Government of Alberta, 2013). However, this is often limited to short-term assessments which may not adequately characterize the natural variability of the ecosystem(s) in question.

In the Athabasca Oil Sands Region (AOSR), systematic aquatic ecosystem monitoring began almost 30 years after initial industrial exploitation of the abundant bituminous sand deposits (Dowdeswell et al., 2010; Humphries, 2008). In the absence of sufficient pre-development monitoring data, paleolimnology (the scientific analysis of sediment cores to reconstruct past changes in environmental conditions of aquatic basins) has been championed as a source of hydroecological and/or geochemical information to characterize the natural range of variation for aquatic ecosystems and contaminant deposition in the AOSR (Arciszewski et al., 2017a; Dowdeswell et al., 2010; Klemt et al., 2021; Mills et al., 2017; Munkittrick and Arciszewski, 2017; Somers et al., 2018). Lake sediments typically accrete vertically, preserving an archive of components of the lake ecosystem, as well as contributions from its catchment and regional airmasses. Sediments deeper in this natural archive reflect older, pre-development periods of ecosystem history. Stratigraphic analysis of lake sediment cores allows for inferences to be made about past lake and environmental conditions, and qualitative to quantitative reconstructions of environmental change, and natural ranges of variability can be defined to establish baseline conditions before potential impact from industrial activities. Contemporary monitoring programs can integrate paleolimnological reconstructions to enhance their ability to detect changes during and after development, thereby supporting ecosystem protection.

Information preserved in lake sediment cores has been critical to characterize the pre-development conditions of the AOSR and to define the area enriched by contaminants of concern emitted from oil sands industrial development, which is currently considered to be within about 50 km of Fort MacKay, AB (e.g., Cooke et al., 2017; Hazewinkel et al., 2008; Jautzy et al., 2013; Klemt et al., 2020, 2021; Kurek et al., 2013; Summers et al., 2016; Wiklund et al., 2012). These studies agree well with spatial analysis of snowpack samples throughout the AOSR (Kelly et al., 2009, 2010; Cho et al., 2014; Kirk et al., 2014; Manzano et al., 2016). Other paleolimnological studies have focused on environmental and ecological conditions in lakes, regional hydroclimatology, and changes due to climatic trends (e.g., Curtis et al., 2010; Summers et al., 2016, 2019). These studies have been instrumental in quantifying and expanding our understanding of the magnitude and extent of industrial effects within the AOSR, as well as elucidating aspects of the pre-development history of the region. Despite common advocacy, paleolimnological data have yet to be used proactively by industry in the AOSR to quantify the range of natural variation for use by monitoring programs to improve ability to detect if, and when, environmental change has occurred as a result of industrial development.

Ongoing resource development interests within the northern AOSR prompted a study of the McClelland Lake watershed, located approximately 34 km NNE of Fort MacKay, AB (Fig. 1). The watershed is located within the surface mineable region of the AOSR and includes a wetland complex dominated by a large alkaline patterned fen, several sinkhole lakes, old-growth pine forests, and the Fort Hills landform, a dissected kame that borders McClelland Lake. Several of these features have been identified as Alberta Environmentally Significant Areas (ESA nos. 633, 638, 679). McClelland Lake is used as a staging area for numerous waterfowl species and as nesting habitat for Bald Eagle (*Haliaeetus leucocephalus*) and Sandhill Crane (*Antigone canadensis*), two sensitive species of concern (Sweetgrass Consultants Ltd, 1997; Fiera Biological Consulting, 2009). Other features near the lake are also significant: the patterned fen, located on the western shore of the lake, is one of the largest in Alberta; the Fort Hills kame and numerous sinkholes lakes, located to the south and throughout the watershed, respectively, are both rare landforms in the boreal region and the lakes may be vulnerable to water table alteration. The Fort McMurray-Athabasca Oil Sands Subregional Integrated Resource Plan (Alberta Environment, 1996, amended 2002) has restricted surface access for resource development within the eastern portion of the patterned fen as well as McClelland Lake. While development is permissible in the western portion of the fen, numerous guidelines in the Resource Plan require the maintenance of 'natural' hydrological conditions that include seasonal variability in the wetland complex, sinkhole lakes, and upland tributary streams of McClelland Lake (Alberta Environment, 1996).

The McClelland Lake-wetland complex is located on the periphery of the area of measurable effects from industrial AOSR activities, including atmospheric deposition of organic contaminants and metals (Cho et al., 2014; Kelly et al., 2009, 2010; Kirk et al., 2014), and as a result has received little scientific study. The paucity of environmental monitoring data for McClelland Lake renders that potential disturbance to the lake from proposed future mining activities will be difficult to identify. Thus, the objective of this study was to determine an appropriate timeframe for defining the natural range of past hydroecological conditions in McClelland Lake, using a comprehensive paleolimnological approach, before the potential onset of mining within its catchment.

2. Materials and methods

2.1. Study site

McClelland Lake (Fig. 1) is a large (surface area = 30.5 km^2, catchment area = 198 km^2), shallow (Z_{max} = 5.5 m), alkaline (mean pH = 8.60; mean total alkalinity as $CaCO_3$ = 142 mg/L) lake located in the AOSR, approximately 50 km north of the centre of oil sands industrial activity. The lake is located within the Firebag River watershed and is intermittently fed by runoff from surrounding streams, including Upper McClelland Creek, the large (9 km^2) adjacent patterned fen and smaller peatlands; the fen and peatlands are the primary sources of runoff to the lake. McClelland Lake is a seasonally closed basin, intermittently drained by Lower McClelland Creek as lake levels fluctuate throughout the year. The water balance of the lake is thought to comprise inputs from precipitation and surface and shallow groundwater inflows along the western patterned fen and losses due to evaporation and diffuse groundwater outflow along the eastern edge of the lake. Lake water oxygen isotope values, measured on several occasions from August 2017 – August 2019, narrowly range from − 10.5 to − 8.7‰ and closely correspond to an estimate of the terminal basin isotopic steady-state value where evaporation is equal to inflow (δ_{SSL}: −9.18‰) for nearby lakes in the Peace-Athabasca Delta (Remmer et al., 2018). Several oil sands leases and/or developments are located nearby or within the catchment of the lake.

The hydrogeological setting has been studied extensively to support development of resources in the area (e.g., True North, 2001). The lake and wetland complex are situated in relatively flat topography (<5 m relief) and are underlain by a Quaternary sand aquifer (10–20 m) and low permeability clay till. The patterned fen has a maximum peat thickness of about 8 m and slopes gently towards the lake. The orientation of strings and flarks within the fen is consistent with groundwater flow towards the lake (Quinton and Roulet, 1998). The patterned fen and lake complex are bounded by sandy upland areas of the Fort Hills kame to the north and south, and hydraulic head distributions indicate groundwater flow within the sand aquifer from the south towards the wetland complex. However, the dilute geochemical composition of the peat groundwaters along the boundary of the fen and lake itself (lake water specific conductance = 242 μS/cm) do not indicate significant contributions from the underlying sand aquifer.

2.2. Sediment core collection

This study reports analyses of sediment cores collected from three sites within the deep-water basin of the lake (3.3 – 5.5 m; Fig. 1). Cores were collected during September 2–4, 2018, using a Uwitec gravity corer (Uwitec; Mondsee, Austria). Two cores were collected from each site, for a total of six sediment cores. Sectioning was completed at 0.5-cm increments using a vertical “push rod type” extruder (Telford et al., 2021). Sediment samples were collected in WhirlPak® bags and stored in the dark at 4 °C during shipping and until analysis.

2.3. Sediment core analyses

Preliminary analysis of sediment composition by loss-on-ignition (LOI) was completed on contiguous samples from all cores collected. Based on these results, including stratigraphic agreement of paired cores and downcore changes in composition, sediment cores were selected for additional analyses (detailed below; Table 1). The sediment cores presented in this study showed similar stratigraphic variability (Fig. S1), implying a unified preserved lake history in the sediment records.

2.3.1. Sediment composition

LOI analysis was used to determine stratigraphic variation in water, organic matter, carbonate, and residual mineral matter content of each sediment core (Smith, 2003). Results from LOI analysis of each core were used to align core-pairs from each sampling location, align chronologies of sediment cores among sites, and select cores for subsequent analysis (Fig. S1; Table 1).

Inorganic sediment particle size distributions were characterized using a Horiba Partica LA-950V2 Laser Particle Size analyzer at Wilfrid Laurier University. Sediments were digested using 5% sodium hypochlorite solution for 24 h at room temperature to remove organic matter (Mikutta et al., 2005), and rinsed with deionized water until sodium hypochlorite residues were removed. Digested sediment samples were dispersed in 0.1% sodium hexametaphosphate solution during analysis. Particle-size classes were defined

Table 1
Distribution of sediment core analyses.

	T1 S2		T2 S3		T2 S4
	C1	C2	C1	C2	C1
^{210}Pb dating	–	x	x	–	x
Age-depth modelling	x	x	x	x	x
Inorganic particle size	–	x	–	x	x
Pigments	x	–	–	x	–
Diatoms	–	–	–	x	–
Organic $\delta^{13}C$ & $\delta^{15}N$	x	–	–	x	–
Cellulose $\delta^{18}O$	–	–	–	x	–
Carbonate $\delta^{18}O$	–	–	x	–	–
Macrocharcoal	–	–	–	–	x

following Blott and Pye (2001).

2.3.2. Radiometric dating & age-depth modelling

Radioisotope activities of ^{210}Pb, ^{226}Ra (as weighted mean ^{214}Bi and ^{214}Pb activities), and ^{137}Cs were measured to quantify age-depth relations in the sediment cores during the past ca. 150 years. Sediments were freeze-dried, packed into 8-mL polypropylene tubes, and sealed with a TFA Silicone Septum (Supelco™) and 1 cm^3 epoxy resin (Devcon® product No. 14310). Samples were allowed to equilibrate for at least 21 days prior to analysis. Measurements of radioisotope activities were collected using an Ortec co-axial HPGe Digital Gamma Ray Spectrometer interfaced with Maestro 32 software at the University of Waterloo. Chronologies based on ^{210}Pb activity were developed using the Constant Rate of Supply (CRS) model (Appleby, 2001; Sanchez-Cabeza and Ruiz-Fernández, 2012).

Analysis of stratigraphic trends in several cores using the CRS-modelled age-depth relationships revealed several conspicuous layers of sediments deposited throughout McClelland Lake, indicating key correlated events in the lake's history. These marker sediment layers contained distinct LOI and particle size characteristics (Figs. S1, S5), which were used in a Bayesian age-depth modelling framework (Blaauw and Christen, 2019) to develop more consistent age-depth relations among sediment cores including beyond the range of the CRS models. Markov Chain Monte Carlo model convergence, an indication that the Markov chain had reached a stationary distribution, was determined using the Gelman and Rubin Reduction Factor (Brooks and Gelman, 1997).

2.3.3. Phototrophic community

Abundance and composition of past communities of phototrophs were reconstructed from analysis of photosynthetic pigments, diatom frustules, and chrysophyte stomatocysts preserved in the sediments. Pigments were extracted following established methods. Briefly, freeze-dried sediments were soaked in a solution of acetone, methanol, and water (80:15:5 by volume) for 24 h at − 20 °C and filtered through 0.22-μm filters. The filtrate was dried under inert (N_2) gas and the residue was stored at − 20 °C until analysis. Dried pigments were re-eluted in 500 μL of injection solution, using Sudan II as an internal standard (Leavitt and Findlay, 1994), and were injected into a Waters HPLC reverse-phase system fitted with a Symmetry C18 column. The system was run following methods by Mantoura and Llewellyn (1983), as modified by Leavitt et al. (1989). Pigments were identified based on their retention times and elution sequence, and comparison of their spectra to known standards (Jeffrey et al., 1997; Roy et al., 2011). Identified pigments were assigned to specific groups of phototrophs, including algae and bacteria (Ringelberg, 1980; Jeffrey et al., 1997; Steinman et al., 1998; Pandolfini et al., 2000; Buchaca and Catalan, 2007; Romero-Viana et al., 2010; Roy et al., 2011). HPLC analysis was conducted at the University of Waterloo.

Microscope slides used to analyze diatoms and stomatocysts were prepared from wet sediment subsamples digested in a 10% HCl solution followed by a 30% H_2O_2 solution, consistent with methods reported by Renberg (1990) and Hall and Smol (1996). A known number of microspheres were added to the resulting cleaned diatom slurries for each sample to determine diatom valve and chrysophyte stomatocyst concentrations (Battarbee and Keen, 1982). Diatom slurries were then dispensed onto circular coverslips, air dried, and mounted onto slides using Naphrax™. Diatom slides were analyzed using a Zeiss Axioskop II Plus oil immersion compound light microscope. Diatom taxonomy followed Krammer and Lange-Bertalot (1986–1991) and Lavoie et al. (2008). Ecological and habitat preferences for diatom taxa followed established literature studies in northern Canada (e.g., Moser et al., 2004; Rühland et al., 2002; Summers et al., 2019). Diatom and chrysophytes stomatocyst enumeration were conducted at the University of Waterloo.

2.3.4. Stable isotope geochemistry

2.3.4.1. Organic carbon & nitrogen stable isotope analysis. Organic carbon (C) and nitrogen (N) elemental content and stable isotope ratios were determined on two sediment components: fine-fraction organic materials and coarse macrophyte remains. Subsamples of wet sediment were treated with 10% HCl at 80 °C for 2 h to remove carbonates and subsequently washed with deionized water until neutral. Samples were then freeze-dried and were sieved into coarse (≥ 250 μm) and fine fractions (< 250 μm).

Plant fragments and seeds from the coarse fraction (≥ 250 μm) were hand-picked and described qualitatively. Seeds were identified using published taxonomic guides and reference collections (Martin and Barkley, 1961; Brayshaw, 2000; Myrbo et al., 2011). One sample of small, shiny, lanceolate seeds from T2 S2 C1 (73.5 cm) was submitted for identification to Paleoscapes Archaeobotancial Services Team LLC. (PAST) in Bailey, Colorado, USA. Fragments were examined under a Bausch and Lomb Stereozoom microscope at magnifications of 10–70x.

Plant fragments from the coarse fraction and sediment from the fine fraction were analyzed for organic C and N elemental and isotope composition using a continuous flow isotope ratio mass spectrometer (CF-IRMS) at the University of Waterloo Environmental Isotope Laboratory. Stable C and N isotope ratios are reported as $\delta^{13}C_{org}$ (‰) units relative to the Vienna-Peedee Belemnite (VPDB) standard and $\delta^{15}N$ (‰) units relative to atmospheric N (AIR), respectively. Analytical uncertainty was ± 0.2‰ and ± 0.3‰ for $\delta^{13}C_{org}$ and $\delta^{15}N$, respectively.

2.3.4.2. Cellulose oxygen isotope analysis. Cellulose oxygen-isotope composition was determined following methods detailed in (Wolfe et al., 2001) using the acid-washed fine fraction (<250 μm) of the sediment subsamples, as described above. Solvent extraction, bleaching and alkaline hydrolysis were used to remove non-cellulose organic constituents, and hydroxylamine leaching was used to remove iron and manganese oxyhydroxides. The purified cellulose samples were freeze-dried, weighed, and analyzed using a CF-IRMS at the University of Waterloo Environmental Isotope Laboratory. Cellulose oxygen isotope results were used to infer lake water oxygen isotope composition ($\delta^{18}O_{lw}$) using a cellulose-water oxygen isotope fractionation factor of 1.028 (Wolfe et al., 2001; Savage et al., 2021), and are expressed as δ-values (‰) relative to the Vienna Standard Mean Ocean Water (VSMOW) standard and normalized to $\delta^{18}O_{SLAP}$. Analytical uncertainty was ± 0.4‰ for $\delta^{18}O_{cell}$.

2.3.4.3. Carbonate oxygen isotope analysis. Carbonates in freeze-dried sediment samples were analyzed using a Delta V Advantage mass spectrometer and a GasBench II peripheral following established methods (Spötl and Vennemann, 2003; Skrzypek and Paul, 2006). Briefly, solid carbonate samples were added to a vial, flushed with helium, and then acidified. After a minimum digestion period of 18 h at room temperature, a CTC Analytics autosampler transferred evolved headspace gas to the GasBench II, where multiple injections of the sample carbon (as CO_2) were measured versus a pure CO_2 monitoring gas. In-house standards, established by analyses of NBS18, NBS19, and LSVEC, were analyzed as samples to permit results reported as $\delta^{18}O_{carb}$ values (‰) relative to the VPDB standard with analytical uncertainty of ± 0.3‰.

2.3.4.4. Isotope-inferred lake-water temperature. Lake water temperature was reconstructed at 5-year increments from depth-averaged

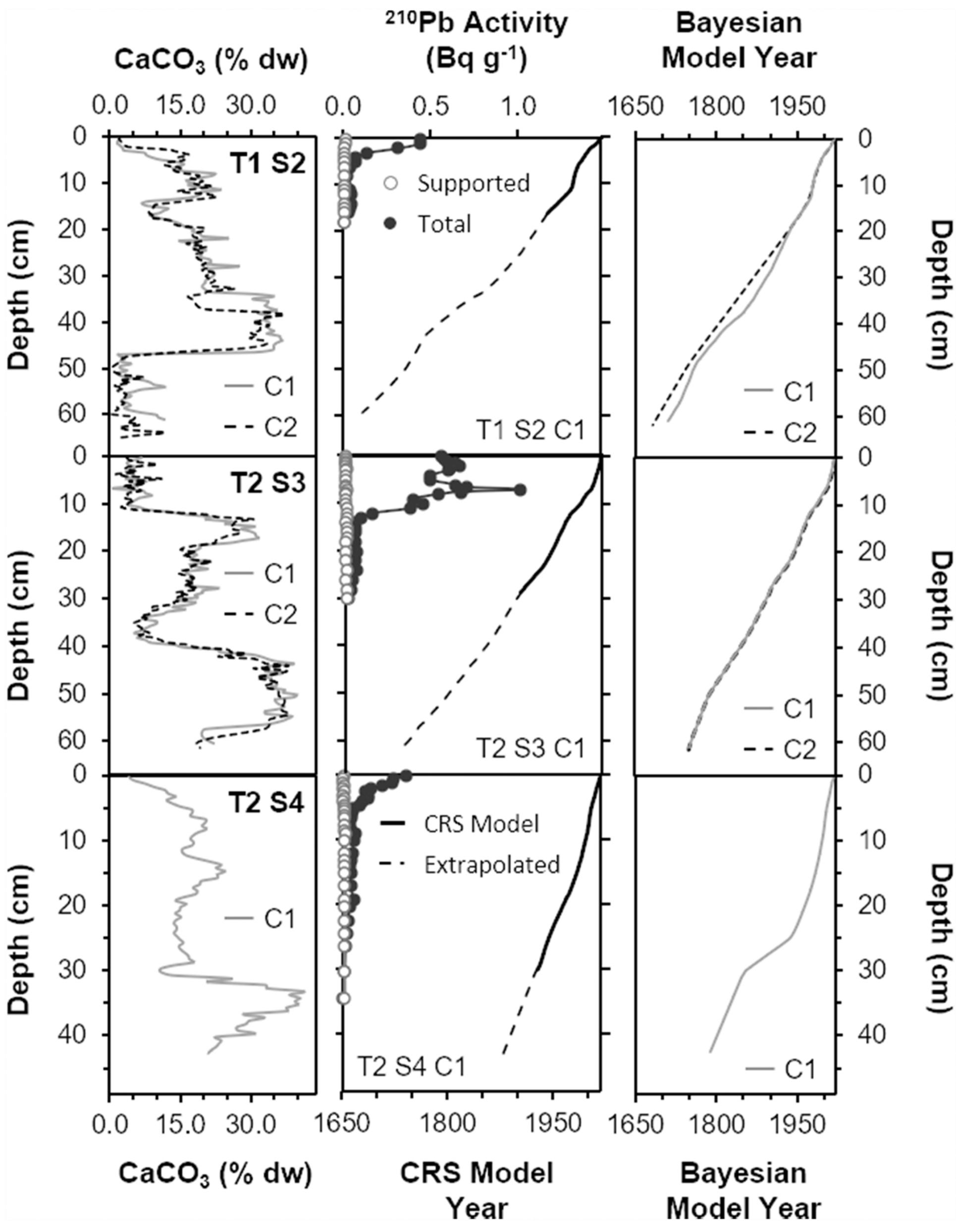

Fig. 2. Wiggle-matched carbonate content (left), total ^{210}Pb activities (center, dark grey circles) and supported ^{210}Pb activities (center, open light grey circles), CRS model ages (center, black line) and extrapolated age estimates (center, black dashed line), and Bayesian age-depth models (right) for the three coring sites used in this study.

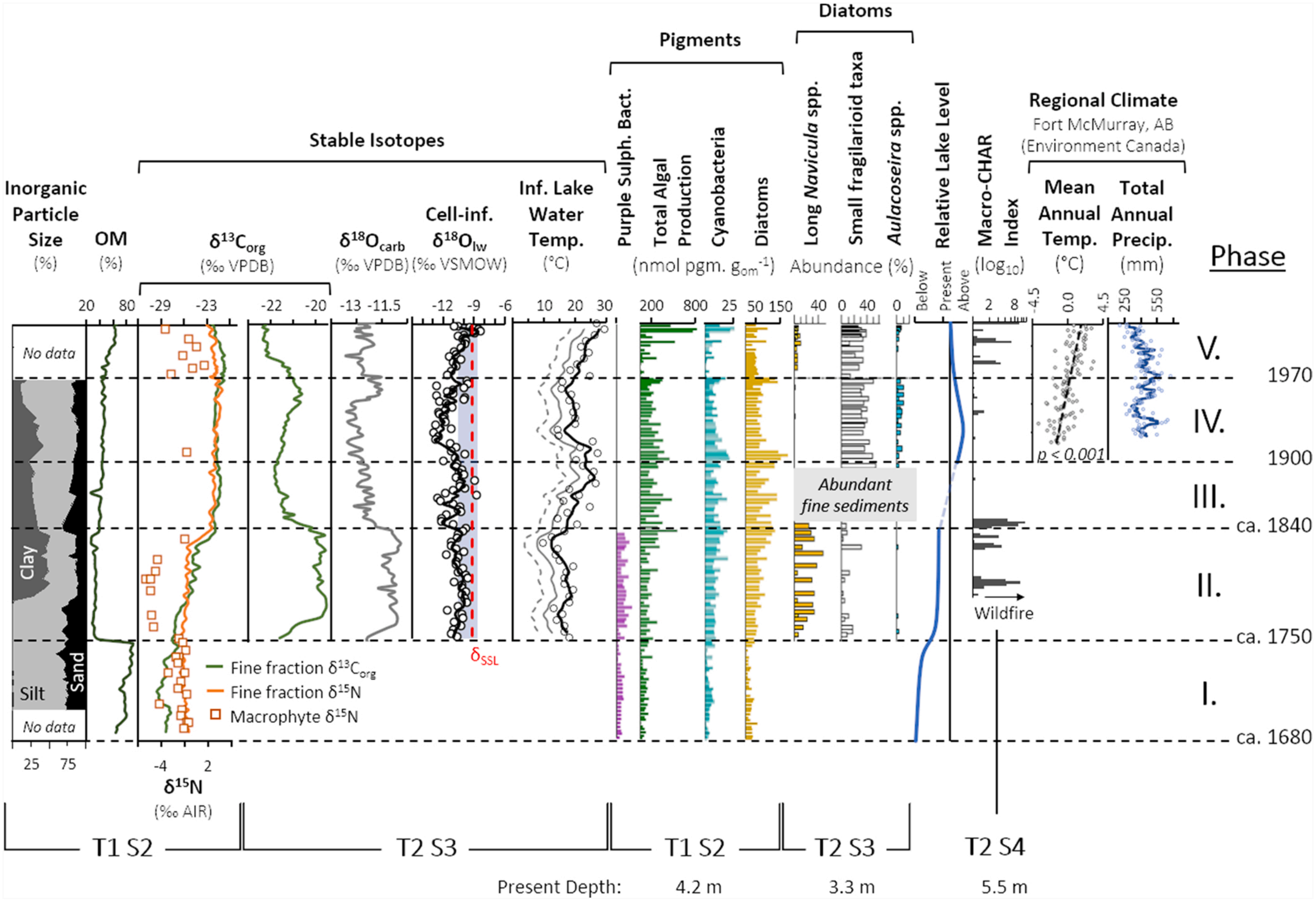

Fig. 3. Summary of the paleolimnological reconstruction at McClelland Lake, AB, highlighting stratigraphic profiles of key variables measured in indicated sediment cores as well as external climatic data (Environment Canada, Fort McMurray, AB, stations 3062693, 3062695, 3062696). Nitrogen isotope ratios ($\delta^{15}N$) are presented for the sediment fine fraction (< 250 μm; orange solid line) and coarse macrophyte fragments (orange squares). Organic carbon isotope ratios ($\delta^{13}C_{org}$) for the sediment fine fraction organic matter are shown for two cores using green solid lines. Three-point running mean cellulose-inferred $\delta^{18}O_{lw}$ is shown using a black solid line while individual values are shown using open circles. Superimposed on this plot is the range of measured lakewater $\delta^{18}O$ values (vertical grey-blue bar), and the estimated contemporary terminal basin isotopic steady-state value (δ_{SSL}: vertical red dashed line) from Remmer et al. (2018). Inferred lake-water temperature was calculated under three scenarios: 1) following Leng and Marshall (2004) and the three-point running mean using a black line; 2) under the assumption carbonates precipitated in more isotopically depleted water (grey solid line); and 3) assuming carbonates were affected by kinetic fractionation in more isotopically depleted water (grey dashed line; Rozanski et al., 2010). Relative lake level was semi-quantitatively inferred based on macrophyte remains, diatom community composition, stable isotope ratios, and inorganic particle size distributions. Significant rise of regional mean annual temperature is shown using a black dashed line, with individual data points shown with open circles. Total annual precipitation is shown using open blue circles and the three-point running mean is shown using a blue solid line. The present water depth of each coring site is indicated along the bottom of the figure. Five hydroecological phases were delineated through qualitative inspection of stratigraphic variations and numerical analyses of all measured sedimentary variables separately for all cores used in this study. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

$\delta^{18}O_{carb}$ and $\delta^{18}O_{lw}$ values, using the calcite-water equilibrium equation (Kim and O'Neil, 1997; Leng and Marshall, 2004):

$$T_{{}^\circ C} = 13.8 - 4.58\left(\delta^{18}O_{carb} - \delta^{18}O_{lw}\right) + 0.08\left(\delta^{18}O_{carb} - \delta^{18}O_{lw}\right)^2$$

This calculation assumes endogenic carbonate and aquatic cellulose are produced in the lake at the same time during the open-water season, and that carbonate precipitation occurs under equilibrium conditions. To account for the possibility that seasonal precipitation of carbonate occurred in early ice-free season waters that had undergone less evaporation than during subsequent photosynthesis of aquatic cellulose, an estimated − 1.0‰ correction was applied to $\delta^{18}O_{lw}$. We also imposed an estimated correction of − 4.2 °C to address the possibility of rapid carbonate precipitation and associated kinetic fractionation effects (Rozanski et al., 2010). Lake water temperatures were re-calculated using the seasonal offset correction and both corrections combined. We regard the 'raw' temperature reconstruction as an estimate of the maximum possible value, whereas the 'combined correction' temperature reconstruction reflects an estimate of the minimum possible value.

2.3.5. Macrocharcoal analysis

Samples of sediment were freeze-dried and passed through a 250-μm sieve. The coarse portion retained on the sieve was hand-picked and charcoal fragments were enumerated using a dissecting microscope. Charcoal fragments were identified using published charcoal morphology (Enache and Cumming, 2006; Jensen et al., 2007; Mustaphi and Pisaric, 2014). Macro-charcoal accumulation rates (mCHAR: # macro-charcoal pieces cm^{-2} yr^{-1}) were calculated using sedimentation rates determined by age-depth modelling. Macro-charcoal enumeration was conducted at the University of Waterloo.

In order to identify periods of past wildfires, mCHAR values were decomposed using published approaches (Clark and Royall, 1996; Long et al., 1998; Higuera et al., 2010). Macro-charcoal accumulation rates were interpolated into 2-year intervals, representing the median age-span of analyzed sediment sections. Background charcoal concentrations were estimated by fitting a robust locally weighted regression to log(x + 1)-transformed interpolated mCHAR values using a 30-year smoothing window (Cleveland, 1979; Higuera et al., 2010). A transform-index detrending model was chosen to develop an interpolated mCHAR index using estimated 'background' and total mCHAR values (Higuera et al., 2010). The transformation index was calculated by dividing log (x + 1)-transformed interpolated mCHAR values by estimated log(x + 1)-transformed background mCHAR values. A threshold value of 1.4 for the mCHAR index was selected by comparing mCHAR values and the associated error in the age-depth model with published wildfire records for Alberta (Government of Alberta, 2017; Higuera et al., 2010). mCHAR index values greater than this threshold were used to infer localized wildfire events.

2.3.6. Numerical analyses and data visualization

Numerical analyses and data visualization were conducted using R Statistical Language version 3.6.0 (R Core Team, 2019), C2 data analysis version 1.7.7 (Juggins, 2014), and QGIS version 3.4.2-Maderia (QGIS Development Team, 2019). Core R packages were used in addition to 'analogue' (Simpson and Oksanen, 2020), 'rbacon' (Blaauw and Christen, 2019), and 'rioja' (Juggins, 2020) packages.

3. Results and interpretation

3.1. Sediment core chronologies

Radiometric analysis was used to develop chronologies for one sediment core from each of the three sites (T1 S2 C2, T2 S3 C1, and T2 S4 C1; Fig. 2). Total ^{210}Pb activity declined with depth in all cores, reaching 'background', or supported, ^{210}Pb activity between 18.0 and 30.0 cm depth (Fig. 2). Stratigraphic profiles of ^{137}Cs activity suggested cesium mobility in all cores, and therefore it could not be used to confirm CRS modelled dates. Bayesian model results produced mean core-bottom ages of 1682 for T1 S2 C2, 1731 for T2 S3 C1, and 1792 for T2 S4 C1 (Fig. 2). Core-bottom ages for paired cores, aligned using wiggle matching of LOI profiles, were 1711 for T1 S2 C1 and 1749 for T2 S3 C2.

3.2. Hydroecological reconstruction

Select physical, geochemical and biological variables measured in cores collected from the three sample locations are presented in Fig. 3 to highlight key findings about variation in hydroecological conditions during the past ca. 325 years at McClelland Lake. Key variables to inform our interpretations include: inorganic particle size; sediment organic matter content; organic carbon and nitrogen isotope composition of the sediment fine fraction (< 250 μm) and coarse macrophyte remains ($\delta^{13}C_{org}$ and $\delta^{15}N$, respectively); cellulose-inferred lake water $\delta^{18}O$ values ($\delta^{18}O_{lw}$) relative to the range of contemporary measured lake water $\delta^{18}O$ values; carbonate oxygen isotope composition ($\delta^{18}O_{carb}$); lake water temperature inferred from $\delta^{18}O_{lw}$ and $\delta^{18}O_{carb}$; indicator pigments for total algal production (as the sum concentration of chlorophyll-*a* and derivatives) and abundance of purple sulphur bacteria (as the sum concentration of okenone, rhodopin, and rhodovibrin), benthic cyanobacteria (as scytonemin), and diatoms (as diatoxanthin); relative abundances of select diatom taxa (long *Navicula* species, small fragilaroid taxa, *Aulacoseira* species, and other centric diatom taxa); and the macrocharcoal index. Relative lake level is semi-quantitatively inferred based on macrophyte remains, diatom community composition, stable isotope compositions, and inorganic particle size distributions. Also included are mean annual air temperature and annual precipitation data for the past 100 years measured at Fort McMurray (Alberta) by Environment and Climate Change Canada. Five hydroecological phases were delineated through qualitative inspection of stratigraphic variations and numerical analyses of all measured sedimentary variables separately for all cores used in this study, and used as a descriptive tool to reconstruct the environmental history of McClelland Lake in coherent time periods. Detailed stratigraphic profiles of loss-on-ignition, elemental C:N ratios, stable isotope ratios, pigment concentrations, and diatom community composition in select cores are included in the Supplemental material (Figs. S1–S5, Table S1).

3.2.1. Phase I (ca. 1695 – 1750)

The earliest phase (ca. 1695 – 1750; Fig. 3) corresponds with a peak of cold, arid conditions of the Little Ice Age (LIA; Luckman, 2000). Only the sediment cores collected from site T1 S2 included this phase. Sediments from this phase were organic-rich ($\geq$ 80%) and peaty. The minor ($<$ 20%) inorganic fraction consisted mainly of silt with some sand and rare gravel, which is coarser than inorganic sediment deposited during the subsequent phases when organic content became substantially lower. Relatively coarse inorganic sediments suggest frequent contributions or in-washing from the catchment (Håkanson and Jansson, 1983), as soils in the McClelland Lake watershed are silt and sand-rich (Natural Regions Committee, 2006). Both coarse and fine sediment fractions had similarly low $\delta^{15}N$ values, ranging from − 4.3 to − 0.5‰ (Fig. 3), as well as relatively low $\delta^{13}C_{org}$ values (range: −29.6 to −23.5‰; Fig. S2) and C:N ratios (range: 11.99–13.26; Fig. S2). Low $\delta^{13}C_{org}$ and C:N ratios are generally indicative of organic matter sourced from aquatic primary production (LaZerte and Szalados, 1982; Leng et al., 2005; Meyers, 1994). Low $\delta^{15}N$ values have been associated with macrophyte production (Leng et al., 2005) and atmospheric nitrogen fixation by cyanobacteria (Gu, 2009; Peterson and Fry, 1987; Talbot, 2001). Conspicuous organic fragments and seeds of *Najas flexilis* (slender naiad, a submergent aquatic plant), *Nuphar variegata* (yellow pondlily, a floating-leaved aquatic plant), and *Potamogeton friesii* (Fries' pondweed, a submergent aquatic plant) were observed during core sectioning and sample processing. Together, this suggests that habitat in McClelland Lake was dominated by submergent, floating-leaved and emergent macrophytes. The presence of *N. variagata* seeds suggests that lake levels were lower than present at site T1 S2 (present-day water depth: 4.2 m), as this species is only found in shallow water typically $<$ 2 m deep (Dieffenbacher-Krall and Halteman, 2000; Nichols, 1999). Following the study by Dieffenbacher-Krall and Halteman (2000), who used seeds found in lake sediments as a tool to infer lake depth, we infer that lake levels were 2.0 – 2.5 m lower than present day. During this phase, McClelland Lake was likely much smaller than present and restricted to the present-day boomerang-shaped deep-water region at the east end of the present-day lake (Fig. 4). Algal communities present were likely diverse but consisted mainly of benthic and epiphytic taxa, and those tolerant of higher UV exposure due to lower lake levels, such as scytonemin-producing cyanobacteria (Fig. 3). Presence of the pigments okenone, rhodopin, and rhodovibrin, produced by strictly anaerobic purple sulphur bacteria (family Chromatiaceae, Madigan and Jung, 1995), identifies that lake-bottom waters were anoxic, at least seasonally. Long duration of ice cover and relatively small lake volume likely fostered seasonal anoxia/hypoxia as a consequence of organic matter decomposition.

3.2.2. Phase II (ca. 1750 – 1840)

The second phase (ca. 1750 – 1840; Fig. 3) includes the end of the LIA (Luckman, 2000), when climate warmed after a preceding colder period. Sediment organic matter content decreased sharply from approximately 75% in Phase I to approximately 25% at the onset of Phase II. Despite this change, macrophyte remains and coarse organic fragments remained relatively abundant. A sharp increase in sediment clay content (from <1 to ~50%) occurred after ca. 1770, identifying increased supply of fine-grained inorganic

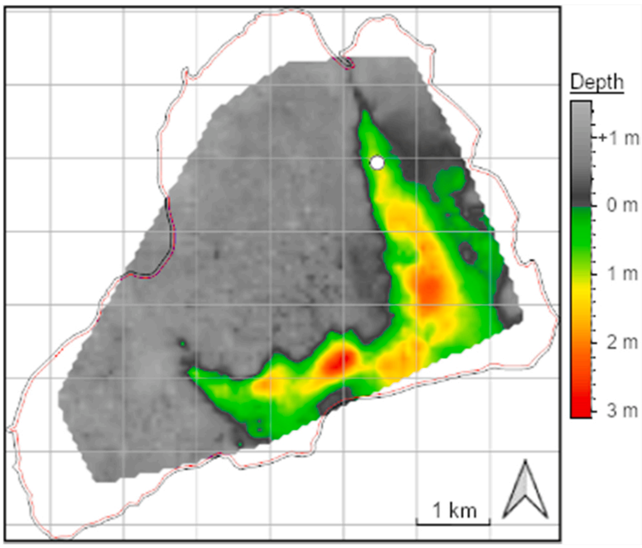

Fig. 4. Bathymetric map of McClelland Lake showing reconstructed low water levels during Phase I (ca. 1695 – 1750), as inferred by macrophyte remains and stable isotope measurements. Sediment core location T1 S2 is indicated using the white circle, on which this paleo-lake level reconstruction is mainly based. Grey areas indicate presently submerged regions of the lake that were likely exposed during this period. White areas within the lake's perimeter indicate regions for which bathymetric data were not available. Graticule is composed of 1 × 1 km squares.

sediment to the lake and/or redistribution of fine mineral sediment within the lake (Håkanson and Jansson, 1983). The increase in fine-grained inorganic sediments and marked decline of organic matter content (Fig. 3) suggest potential suppression of algal primary production in the water column and/or a decline in organic matter preservation. While remains of *N. variegata* were no longer found among the macrophyte fragments of this phase, seeds of *P. friesii* were still present. Given *N. variegata* requires shallow-water habitats but *Potamogeton* can grow in deeper water (Dieffenbacher-Krall and Halteman, 2000), we infer that lake levels likely increased during this period to approximately 0.5 – 1.0 m lower than present (i.e., maximum depth was ~4.0 – 4.5 m). Water balance remained relatively stable and strongly influenced by evaporation during this phase, indicated by relatively consistent values of cellulose-inferred $\delta^{18}O_{lw}$ (mean ± SD: −10.6 ± 0.6‰) that are slightly lower than the estimated modern isotope value where evaporation is equal to inflow for lakes in the nearby Peace-Athabasca Delta (δ_{SSL} = −9.18‰; Remmer et al., 2018). Carbonate $\delta^{18}O$ values were relatively high, likely driven by both evaporation and relatively low isotope-inferred water temperatures (~4–19 °C).

Higher inferred water level and warming climate coincided with increased algal production and a proliferation and diversification of the benthic and epiphytic diatom community, as indicated by increased concentrations of algal pigments (Figs. 3, S4), relatively high concentrations of diatom frustules compared to later phases (Fig. S3), and high $\delta^{13}C_{org}$ values (Fig. 3). The dominant diatom taxa are those that occupy mainly benthic and epiphytic habitats, including *Cymbellafalsa diluviana*, long *Navicula* species, and *Amphora* species (Fig. S3). *Fragilaria capucina* and *F. tenera* were also relatively abundant during this phase (Fig. S3). However, these taxa are relatively adaptable and can occupy both planktonic and periphytic habitats (Rühland and Smol, 2002, 2005), potentially indicating a modest expansion of planktonic habitat in the lake as water levels increased. The composition of the diatom community indicates limnological conditions during this phase were likely slightly more acidic than at present with higher water-column concentrations of DOC and total nitrogen (Rühland and Smol, 2002, 2005; Moser et al., 2004). Increased abundance of pigments from purple sulfur bacteria (Fig. 3) indicates continued seasonal anoxia, likely driven by higher lake primary production. Macrocharcoal analysis detected 2–3 episodes of substantial local wildfire, but these episodes resulted in few if any discernable changes in the sedimentary variables analyzed.

3.2.3. Phase III (ca. 1840 – 1900)

The third phase (ca. 1840 – 1900; Fig. 3) was defined by notable change in several paleolimnological variables. Marked declines in $\delta^{13}C_{org}$ and $\delta^{18}O_{carb}$ were observed, as well as greater variation of cellulose-inferred $\delta^{18}O_{lw}$ values, which suggest a period of increased hydrologic variability. Carbonate $\delta^{18}O$ values declined, likely in response to a rise in isotope-inferred water temperature (8–26 °C) relative to Phase II as the climate warmed following the end of the LIA. Inorganic sediment grain size, sediment organic matter content, and $\delta^{15}N$ did not vary greatly during this phase. Increasing concentrations of several algal pigments (Fig. S3) suggests a shift in algal community composition driven by steady increases in algal production by several algal groups (Leng et al., 2005), likely in response to increasing water temperatures and associated longer growth season duration (Fig. 3). The disappearance of purple sulfur bacterial pigments (Fig. 3) supports increasing lake levels, greater wind-driven water column mixing, and/or longer ice-free season, which would have reduced the incidence and duration of lake bottom anoxia. Abundant fine sediments obscured the few diatom frustules present in sediments during this phase, making quantitative diatom enumeration impossible. The few identifiable diatom remains visible were from heavily silicified taxa. Thus, variation in lake depth, temperature, pH, zooplankton grazing, and/or post-depositional bioturbation could have resulted in poorer preservation of diatom frustules (Battarbee et al., 2001; Hassan et al., 2018; Ryves et al., 2003, 2006, 2013). Increases in macrocharcoal index values ca. 1840 indicate a period (~10 years) of extensive, frequent wildfire activity around the lake or possibly one major wildfire event. Fire could be a potential driver of the variable hydrological and limnological conditions in McClelland Lake at the onset of this phase.

While the inferred conditions during Phase III appear well supported by the data, we cannot exclude the possibility that other interpretations of the hydroecological conditions are possible. Given the complex morphometry of the lake (i.e., extensive shallow margin with small deep-water basin) and underlying karst bedrock porous for groundwater movement, it is possible that decline of lake levels could have resulted in values measured in the paleolimnological variables during the second half of Phase III. For example, rise of cellulose-inferred $\delta^{18}O_{lw}$ values to those approaching present-day δ_{SSL}, combined with rising isotope-inferred water temperature could suggest increased evaporative drawdown of lake levels. Decline of water level could lead to transport of fine sediment from the littoral zones and to the deep-water basin and result in the abundant fine sediments observed on diatom slides. Resultant increase in influence of groundwater flux, undersaturated in silica, on water balance of a reduced lake that occupied only the deep-water basin, could account for dissolution of lightly silicified diatom frustules (Rippey, 1983; Ryves et al., 2006, 2009). Despite uncertainty of interpretations for the second half of Phase III, inference of increased water depth during the first half of Phase III is well supported by sharp decline of cellulose-inferred $\delta^{18}O_{lw}$ values indicative of positive water balance (Fig. 3), marked rise in inorganic sediment particle size (Fig. S5) and increase of sediment on diatom slides (Figs. 3, S3) via transport to the coring site by input waters, and loss of pigments from purple sulfur bacteria indicating loss of deep-water anoxia (Figs. 3, S4).

3.2.4. Phase IV (ca. 1900 – 1970)

The fourth phase (ca. 1900 – 1970; Fig. 3) was characterized by rising $\delta^{13}C_{org}$ values (core T2 S3; Fig. 3), sharp increases and high concentrations of several algal pigments (Fig. S4), a shift to dominance of epiphytic and fragilarioid diatom taxa, and the appearance of planktonic *Aulacoseira* taxa (Figs. 3, S3). Relatively low cellulose-inferred $\delta^{18}O_{lw}$ values (<−12‰) during ca. 1910–1950 indicates reduced influence of evaporation and strongly positive lake water balance, and rising values after ~1920 are indicative of a shift towards increasing importance of evaporation. Carbonate $\delta^{18}O$ values continue to decline during most of this phase, mainly in response to shifting hydrological conditions given isotope-inferred water temperatures (8–25 °C) are similar to Phase III. An abrupt increase in $\delta^{18}O_{carb}$ values between ca. 1950 and 1970 appears to be in response to an increase in evaporation.

As influence of evaporation increased during the latter portion of Phase IV, increases in algal production were observed, indicated by increase in concentrations of several pigments and rise of $\delta^{13}C_{org}$ values (Fig. 3), and increase in concentrations of diatoms and chrysophyte stomatocysts (Fig. S3). Peak algal production during this phase, ca. 1940–1965, was coincident with the appearance of other planktonic diatoms, including *Cyclotella* spp. and *Tabellaria flocculosa*. Their presence suggests deeper water levels in McClelland Lake during this phase and stronger stratification of the water column that persisted until the end of this phase (Rühland and Smol, 2002; Moser et al., 2004). The diatom community during this phase differed markedly from previous phases and was composed mainly of small fragilarioid and achnanthoid taxa (Figs. 3, S3). These taxa are tolerant of lower light availability and are associated with abundant submergent macrophyte growth (as many achnanthoid and some fragilarioid species are epiphytic), more alkaline conditions, and higher DIC and lower DOC concentrations (Rühland and Smol, 2002, 2005; Moser et al., 2004). Several small wildfires, reconstructed using macrocharcoal in the sediment record, occurred during the ca. 1950s and 1960s (Fig. 3), in agreement with the historical wildfire record (Government of Alberta, 2017). Wildfire activity, as well as changing lake level, likely resulted in more turbid waters, shifting the diatom community from dominance by large benthic species during the first two phases of lake history to small, disturbance-tolerant benthic species during Phase IV (Figs. 3, S3).

3.2.5. Phase V (ca. 1970 – 2018)

The fifth phase (ca. 1970 – 2018; Fig. 3) is marked by rising sediment organic matter content, cellulose-inferred $\delta^{18}O_{lw}$ (Fig. 3), relative abundance of benthic diatom taxa, and diatom:chrysophyte ratio (Fig. S3), and by declining $\delta^{13}C_{org}$ values (Fig. 3) and concentration of pigments produced by green algae and cryptophytes (Fig. S4). During this phase, cellulose-inferred $\delta^{18}O_{lw}$ values increase steadily from − 11.0 to − 9.2‰ and approach contemporary measured lake water $\delta^{18}O$ values (Fig. 3), indicating continued increase in importance of evaporation, which aligns with declining total annual precipitation and warming air temperature in the meteorological record. A brief excursion to the highest cellulose-inferred $\delta^{18}O_{lw}$ values in the mid-2010s may reflect increasingly arid conditions. Increasing isotope-inferred water temperature (11–28 °C), relative to Phase IV, also corresponds with rising mean annual air temperature.

Increasing concentration of algal pigments and diatom concentrations (Figs. 3, S3, S4) indicate increasing algal production during this phase, perhaps in response to climate warming and associated longer growth season duration (Fig. 3). Changes in pigment concentrations (Figs. 3, S4) indicate that increasing total algal production is associated with rise in chlorophyte, cyanobacteria, and diatom biomass. A sharp increase in diatom:chrysophyte ratio (Fig. S3) suggests possible nutrient enrichment favouring the production of diatoms over chrysophytes (Smol, 1985). Warmer, drier conditions may be slowly lowering lake level, leading to increased relative abundance of benthic diatoms such as *Cymbellafalsa diluviana* and long *Navicula* spp. which were previously abundant when low inferred lake levels occurred during Phase II. Recent (post-2000) increases in centric, planktonic diatoms suggest a longer ice-free season and greater availability of planktonic habitat (Fig. 3; Rühland and Smol, 2005). Similar changes in diatom community composition have been reconstructed at lakes throughout the AOSR (Libera et al., 2020; Summers et al., 2017, 2019). Several wildfires occurred during the 1980s and 2000s, with macrocharcoal-reconstructed events agreeing with the documented wildfire record (Government of Alberta, 2017).

4. Discussion

4.1. Summary of the McClelland Lake multi-proxy stratigraphic record

In this study, we used multiple lake sediment cores to reconstruct the past ca. 325 years of environmental history of McClelland Lake from analysis of several key paleolimnological variables. Results were used to identify five distinctive phases, characterized by differing climatic, hydrological, and ecological conditions. The first phase (ca. 1695 – 1750) included the Little Ice Age (LIA) when lake depth in McClelland Lake was inferred to be 2 m or less (i.e., 2.0 – 2.5 m lower than at present; Fig. 4), and primary production was dominated by macrophytes and benthic and epiphytic diatom taxa. Phase II (ca. 1750 – 1840) was a period of inferred lake expansion due to rising water levels and was accompanied by a proliferation of primary production as the climate warmed following the end of the LIA. A period of inferred hydrologic variability followed during Phase III (ca. 1840 – 1900), marked by increase of isotope-inferred water temperature and poor diatom frustules preservation. Phase IV (ca. 1900 – 1970) was characterized by a return to relative hydrological stability and increased lake levels, with high concentrations of algal pigments and a diatom community dominated by small, epiphytic taxa. Phase V (ca. 1970 – 2018) was marked by increases in several variables that indicated rising isotope-inferred water temperatures and primary production, lower lake levels, and relatively frequent wildfires. Meteorological records during Phase V align with the water temperature and lake-level reconstructions. The paleolimnological record provides a detailed account of past hydrological and ecological conditions at McClelland Lake. The largely coherent reconstruction is indicative of the strength of our extensive multi-proxy approach, which furnishes opportunity to consider an appropriate interval of time to define the natural range of variability (NRV).

4.2. Temporal bounds for defining natural ranges of variability (NRVs)

The term ‘natural range of variability (NRV)’ is a concept used to describe the range of environmental conditions prior to potentially deleterious human activities. Also referred to as the ‘normal range’ or ‘baseline conditions’ in the literature, this concept aims to define relevant environmental conditions prior to onset of anthropogenic activity or those conditions representative of acceptable, ‘normal’, or unimpacted ecosystems (Kilgour et al., 2015). Typically, the natural range of variability is characterized prior to development through monitoring and the systematically collected data are considered relevant and representative of the natural range, despite issues with duration and frequency of pre-development monitoring and potential for effects due to timing resulting in poor representation of the true range of natural variability (Schmitt and Osenberg, 1996). In this study, very few contemporary monitoring data are available for McClelland Lake, and so a paleolimnological approach was identified as the preferred method to reconstruct pre-development conditions. As a result, we are faced with the opposite problem: an abundance of pre-development data characterizing several phases of lake history. Given the abundance of data and extensive amount of time the stratigraphic record comprises, the relevance and representativeness of the reconstructed data and the timeframe for defining NRVs for McClelland Lake require evaluation.

Setting the timeframe for defining NRVs for McClelland Lake requires careful consideration of the modern environmental setting for the lake and the goals for a monitoring program, including anticipation of potential future impacts from development activities within the catchment, the expected project life span, and the value system(s) of the multiple stakeholders involved in the future of McClelland Lake. While the explicit definition of NRVs for McClelland Lake, and the development and application of methods for their quantification, are outside the scope of this paper, below we discuss the merits and potential drawbacks of various temporal bounds that could be used to define NRVs for McClelland Lake based on scientific understanding. Indeed, this critical decision is needed to inform definition and quantification of NRVs. Ultimately, to develop an effective monitoring program capable of detecting changes attributable to local development, guidance on NRV definition and quantification should be provided by all stakeholders and founded on rigorous scientific monitoring frameworks.

Several studies have demonstrated effects of anthropogenic climate change on lakes in northern Alberta, and that hydroecological changes are already underway (Kurek et al., 2013; Summers et al., 2016, 2017). Meteorological data for the AOSR indicate warming air temperatures and declining annual precipitation since about 1970. Our paleolimnological results indicate the hydroecological responses to these climatic changes at McClelland Lake include increasing primary production, shifts in algal community composition, warmer isotope-inferred water temperatures and rising influence of evaporation on water balance. With continued climate change expected in the AOSR, the time period used to define NRVs should capture climatic and environmental conditions and trends expected in the future. Phase I (ca. 1680–1750) occurred during the LIA, a period of cooler and drier climate (Luckman, 2000), when the lake was significantly smaller in area than at present. Peat-rich sediments accumulated at the bottom of the deep-water basin of McClelland Lake at this time, which are absent during the subsequent four phases. The smaller lake area, lower water levels, and very different sediment characteristics during Phase I suggest that this phase may not be relevant to the modern day. At the nearby Peace-Athabasca Delta, paleolimnological evidence identified near-complete desiccation of a perched basin during the LIA, hydrological conditions that are also outside the range of variability since that time (Wolfe et al., 2005). Although lower water levels in lakes and ponds are anticipated for this region based on climate change models (Thompson et al., 2017), we suggest the analogous conditions during Phase I are unlikely to be environmentally relevant to the current lake-state. We consider these conditions to be outside the 'limits of acceptable change' (Gell, 2017) for defining NRVs upon which change induced by future potential mining development should be evaluated.

Paleolimnological data at McClelland Lake during subsequent Phases II through V reveal marked variation in lake levels (from lower to higher than at present) and water balance, and associated changes in biotic communities, sediment composition, nutrient cycling, and aquatic productivity. These data appear well suited for defining NRVs capable of capturing a range of potential future ecosystem states due to the potentially combined effects of mining development within the lake's catchment and regional climate change – including increase and decrease of water inputs relative to losses. Despite somewhat cooler conditions, reconstructed hydroecological conditions at McClelland Lake during Phase II (ca. 1750–1840) have the potential to resemble anticipated lower lake and pond levels caused by ongoing climate change (Thompson et al., 2017). This phase included demonstrably lower water levels in McClelland Lake, while the lake maintained a similar area to present. The inclusion of Phase II in defining NRVs would thus likely accommodate analogous future conditions in McClelland Lake due to processes unrelated to mining development within the catchment, such as climate-induced lower water levels. The observed variability reconstructed during Phase III, including potential influence of one or more wildfires, is highly likely to occur in the future of the lake, as wildfire frequency and extent are expected to increase with continued climate change (De Groot et al., 2013). Phase IV encompasses the most recent conditions in the lake that are similar to present, but still prior to industrial development in the AOSR. Finally, Phase V includes the current state of the lake and includes effects of recent regional climate change.

Although we recommend that 1750 to present is the relevant time period for defining NRVs for McClelland Lake, based on paleolimnological data spanning Phases II through V, arguments can be made to include or exclude some portions of these temporal bounds. The two most recent phases (Phases IV and V) span just over a century and capture a period of rising air and isotope-inferred water temperatures, declining precipitation, and increasing influence of evaporation on lake water balance at McClelland Lake. Hydroecological conditions during these phases likely represent the narrowest definition of the range of natural variability for McClelland Lake that are most similar to present and near-future conditions. Thus, Phases IV and V may be sufficient for quantitative definition of NRVs for the lake, upon which a monitoring program can be based with a sound scientific basis. The NRVs will not capture a broader range of natural variation experienced by the lake during preceding phases, but this approach would result in conservative NRVs ensuring strong protection for McClelland Lake and greater likelihood of change detection. This could be a preferred and beneficial approach for early detection and warning of changes in the lake, which could prompt enhanced surveillance of lake conditions, and possibly additional studies. On the other hand, narrower temporal definition of NRVs comes with increased likelihood of detecting a false positive or committing type I error (i.e., detection of hydroecological changes at McClelland Lake that are not a result of mining development, but instead are a result of other factors such as regional climate change). Investigation of false positives would result in increased expenditure of time and resources for developers who are not responsible for these changes.

Conversely, inclusion of older Phases II and III that highlight the dynamic nature of McClelland Lake and past responses to differing climatic conditions and disturbance events, including wildfires, will lead to broader definition of NRVs and are more likely to encompass hydroecological changes that have potential to occur with ongoing climate change. A broader definition may more accurately reflect the "true" nature of variability in McClelland Lake, from an environmental perspective, as it more fulsomely acknowledges the pre-development history of the lake. However, a broad definition would be less sensitive to departures from the range of natural variability and provide protection for developers in avoiding unnecessary studies. Broader definition would, however, increase the likelihood of false negatives or committing type II error (i.e., not detecting changes caused by development). To minimize the likelihood of detection errors and help discern effects due to local mining development versus climate change, monitoring programs could utilize one or more reference sites. This would afford the lake greater protection while balancing the cost to developers implementing monitoring programs by incorporating a regional hydroecological definition of natural variability (Arciszewski and Munkittrick, 2015; Hanson, 2011; Lindenmayer and Likens, 2010).

Ultimately, decisions to include or exclude phases of the natural environmental history of McClelland Lake should be made collaboratively by all stakeholders involved in any development project(s) and they should be founded on solid scientific principles, stakeholder priorities, and risk-management frameworks. Careful attention should be paid to balancing potential risk of harm to the environment with risk of harm to the industrial development, such that the likelihood of false negatives is equal to that of false positives, as has been done for federal Environmental Effects Monitoring (EEM) programs in Canada (Environment Canada ECCC, 2012). Part of managing these risks is understanding the implications of various temporal bounds for defining the range of natural variability for McClelland Lake, which can be informed using the paleolimnological reconstruction presented herein. Once the temporal bounds have been established by stakeholders, the range of natural variability can be quantified. While beyond the scope of this study, various perspectives, methods, and issues in quantitatively defining natural ranges of variability have been discussed in the literature (e.g., Barrett et al., 2015; Kilgour et al., 1998, Kilgour et al., 2015; Munkittrick and Arciszewski, 2017).

5. Conclusion

Paleolimnological analyses of multiple sediment cores have revealed temporally dynamic hydroecological conditions at McClelland Lake during the past ca. 325 years. Water levels, water balance, water clarity, primary production, algal community composition and abundance, sediment properties and nutrient cycling have responded to past climatic variations and disturbance events. Here, we use these data to suggest that pre-impact baselines, or natural ranges of variation (NRVs), be defined using Phases II through IV of the five-phase history which span the past ~270 years (since ca. 1750) of the ca. 325-year long sediment record obtained in this study. Phase II (ca. 1750–1840) captured lake conditions during the end of the cool, arid Little Ice Age, which may not provide good analogues to future projected climatic conditions, but it captured a period of lower lake levels and inferred seasonal anoxia which could be possible under future climate change scenarios. Ultimately, the timeframe used to bound definitions of NRVs should be decided collaboratively by all stakeholders and based on scientific rigour, stakeholder priorities, and risk management frameworks. Approaches and formulae for quantitative definition of NRVs for paleolimnological variables, as well as monitoring programs, can then be developed after consensus has been reached on relevant timeframes for defining natural variation. We echo previous authors in championing the use of paleolimnology as a valuable approach for characterizing baseline ecosystem condition and we further support the incorporation of paleolimnology during the early design stages of monitoring programs at McClelland Lake and lakes elsewhere.

CRediT authorship contribution statement

N.A. Zabel: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Visualization, Writing – original draft, Writing – review & editing; **A.M. Soliguin**: Investigation, Writing – original draft; **J.A. Wiklund**: Formal analysis, Investigationn; **S.J. Birks**: Conceptualization, Funding acquisition, Project administration, Formal analysis, Writing – review & editing; **J.J. Gibson**: Conceptualization, Funding acquisition, Project administration, Formal analysis, Writing – review & editing; **X. Fan**: Funding acquisition, Project administration, Writing – review & editing; **B.B. Wolfe**: Conceptualization, Funding acquisition, Methodology, Project administration, Resources, Supervision, Writing – review & editing; **R.I. Hall**: Conceptualization, Funding acquisition, Methodology, Project administration, Resources, Supervision, Writing – review & editing.

Declaration of Competing Interest

None.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.ejrh.2021.100977.

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