

Development and application of aquatic ecosystem monitoring approaches for the Peace-
Athabasca Delta and Whooping Crane Nesting Region of Wood Buffalo National Park

by

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Author's Declaration

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Statement of Contributions

Laura Neary was the sole author of Chapters 1 and 5, which were written under the supervision of Professors Roland Hall and Brent Wolfe and were not intended for publication. This thesis consists in part of three manuscripts (Chapters 2-4) written for publication as peer-reviewed scientific journal papers. Exceptions to sole authorship of material are as follows:

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Abstract

Shallow waterbodies are abundant in northern regions where their small water volume and large surface area-to-volume ratio make them vulnerable to effects of climate change, natural resource development and long-distance transport of contaminants. Canada's largest national park, Wood Buffalo National Park, spans the NWT-Alberta border and contains two Ramsar Wetlands of International Importance: The Peace-Athabasca Delta (PAD) and the Whooping Crane Nesting Region (WCNR; hereafter referred to as the Whooping Crane Summer Range (WCSR)). These aquatic landscapes play vital roles in supporting biodiversity and traditional livelihoods of Indigenous Peoples of the region, and in providing habitat for the threatened wood bison (*Bison bison athabascaae*) and the endangered whooping crane (*Grus americana*). They also underpin the park's designation as an UNESCO World Heritage Site. Despite these recognitions, increasing concern over aquatic ecosystem degradation due to climate change and industrial development prompted UNESCO to evaluate the park's status in 2017 and whether it should be downgraded to 'World Heritage in Danger'. In response, Parks Canada developed an Action Plan in 2019, which was supported by federal investment of \$87 million towards its implementation. Key actions include the development and implementation of an integrated aquatic ecosystem monitoring program for the PAD and WCSR to assess how aquatic ecosystems in these landscapes respond to multiple potential stressors – a need that has long been recognized.

Monitoring aquatic ecosystems in remote, hydrologically diverse landscapes poses significant challenges, particularly in balancing the need for comprehensive spatial and temporal data with logistical and financial constraints. Effective sampling methodologies that can track changes across a range of timescales (e.g., from hours to years) at a landscape scale can help overcome these barriers and generate critical knowledge to support aquatic ecosystem monitoring programs. At the PAD and WCSR, hydrological processes strongly influence the physical, chemical, and biological conditions of waterbodies. Thus, tracking these processes across space and time must serve as the cornerstone of an aquatic ecosystem monitoring program designed to address longstanding concerns about reduced

freshwater availability due to hydroelectric regulation of the Peace River by the WAC Bennett Dam and potential contamination from upstream Alberta Oil Sands operations.

This thesis reports results obtained during 2015-2022 using integrated, multi-faceted field-sampling approaches to investigate the responses of shallow waterbodies in two ecologically significant landscapes to anthropogenic stressors. Given the foundational role of hydrology in shaping these aquatic ecosystems, a multi-method approach was used to characterize influential hydrological processes, including episodic river flooding at the PAD, groundwater discharge in the WCSR, as well as precipitation and evaporation. The sampling spanned from continuous hourly water depth measurements to systematic seasonal (spring, summer, fall) measurement of water isotope tracers and water chemistry surveys at ≥ 60 waterbodies across both landscapes over multiple consecutive years. Additional approaches were employed to assess enrichment of nickel and vanadium, key oil sands indicators, at lakes across the PAD.

Key findings presented in this thesis are organized across three data chapters, each contributing advancements to understanding the hydrological processes shaping shallow waterbodies in the PAD and WCSR, as well as contaminant enrichment in fluvial-derived sediment to lakes in the PAD. In **Chapter 2** continuous water depth measurements were used to develop a new lake classification scheme for the PAD based on four distinct depth variation patterns associated with a singular, dominant hydrological process: 1) ‘Drawdown’ (≥ 15 cm decline) by evaporation and/or outflow after ice-jam floods, 2) ‘Stable’ lake levels (< 15 cm change) sustained by rainfall, 3) ‘Gradual Rise’ by inundation from the open-drainage network, and 4) ‘Rapid Rise’ by input of river floodwater. River flooding during the open-water season has not been well recognized as an important recharge mechanism for lakes in the PAD, yet it occurred extensively in the Athabasca sector in 2018 and 2019. Retrospective analysis of past peak summer levels of the Athabasca River revealed that open-water flooding equivalent to that of 2018 has likely occurred during 16 years since 1982 (42% of the years). Thus, open-water flooding, a phenomenon rarely reported on in the scientific literature, is a common hydrological process affecting broad areas of the Athabasca sector of the delta. **Chapter 3** synthesizes evidence gathered during a 7-year

research program at the PAD (from which Chapter 2 emerged) to explore the associations between lake ecosystem processes in the PAD and climate indices, assess if upstream industrial activities have enriched concentrations of substances of concern, and provide a foundation for multi-faceted aquatic ecosystem monitoring at the PAD. Integration of evidence from key metrics, including isotope-inferred lake water balance, water chemistry and contaminant enrichment, was used to identify status and trends over space and time and assist with identifying causes of change. The research coincided with a period of marked climatic variation, which provided opportunity to evaluate the effectiveness of the methodologies and responses of lakes to changes in hydrology and climate. Four key recommendations are provided to operationalize the methodologies and approaches presented in Chapter 3, which strive to maximize the information content of cost-effective measurements to sustain a long-term monitoring program. **Chapter 4** extends use of the methodologies beyond the PAD to 63 shallow ponds in the WCSR, to characterize key hydrological processes and their spatial and seasonal patterns in 2022. Peak water levels occurred in spring when consistently low isotope compositions and low concentrations of major nutrients and ions reveal widespread, profound influence of snowmelt runoff. Spatial variability of these measurements increased during summer and fall as connectivity to groundwater waned and influence of evaporation increased at some waterbodies. Water level drawdown was less at waterbodies with strong versus weak groundwater connectivity (median = 23% vs 44% of initial depth, respectively). Geospatial interpolation of a multi-metric 'Vulnerability Index', based on isotope-inferred evaporation-to-inflow ratios, water-level drawdown and water chemistry ordination sample scores, identified that Whooping Crane nest locations in 2022 cluster within areas where waterbodies retained strong connectivity to groundwater.

Collectively, this research aims to inform the design and implementation of sustainable, long-term monitoring programs for shallow aquatic ecosystems in the PAD and WCSR to enable rapid detection of changes and their causes in the face of accelerating industrial development and climate change. The knowledge gained can be used to guide policy decisions under the WBNP Action Plan, and the methodologies may be applied at other

remote, water-rich landscapes to evaluate for changes caused by multiple potential stressors.

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Figure 5.1. Illustration of a proposed ‘Research-to-Inform Monitoring Continuum’ that captures key phases of the research process required to inform and guide implementation of effective and sustainable environmental monitoring programs. Research contained in this dissertation (Chapters 2-4), collectively, spans the Continuum from *Conceptualization* to *Synthesis and Recommendations*.

List of Tables

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Statement of Positionality

My understanding of my role as a scientist has evolved throughout my PhD, reflecting broader shifts in ‘Western Science’ philosophies and approaches. I’m not referring to the scientific method, hypothesis testing or development/refining of major scientific theories. I refer to the increased recognition of Indigenous sovereignty in Canada (e.g., UNDRIP; which was ‘fully’ endorsed in 2016), and more specifically the decolonization of research methodologies, recognition of the value of Indigenous Knowledge systems and co-managed stewardship models built upon respect and reciprocity. My perspectives and opinions have been shaped by these emerging research philosophies and I, a settler on Turtle Island and researcher working in environmental science in northern Canada, will carry these teachings into my own research practices and career endeavors in the future.

“If I have seen further it is by standing on the shoulders of giants”

~Sir Isaac Newton

Chapter 1 | Introduction

Shallow waterbodies (lakes, ponds, wetlands) are abundant and characteristic features of many landscapes in northwestern Canada, where they provide hotspots of productivity and biodiversity, and ecosystem services for wildlife and humans (Dudgeon et al., 2006; Brandt et al., 2013; Reid et al., 2018). Due to their small water volume and high surface area-to-volume ratio, however, shallow waterbodies are vulnerable to effects of climate change and industrial activities (Downing et al., 2006; Downing, 2010), and they have become well recognized as ‘sentinels of change’ (e.g., Schindler and Smol, 2006; Vincent and Pienitz, 2006; Schindler, 2009; Williamson et al., 2009; Bakaic and Medeiros, 2016; Saros et al., 2023). Climate change has already altered hydrological regimes in northwestern Canada via shifts in precipitation patterns, longer ice-free seasons and greater permafrost thaw (Zhang et al., 2000; Prowse et al., 2006, 2009; Vincent et al., 2018; Bonsal et al., 2020; Li et al., 2022; Sharma et al., 2021). Meanwhile, expanding natural resource development, especially in the Boreal ecozone (Kreutzweiser et al., 2013; Webster et al., 2015; Musetta-Lambert et al., 2018), poses additional threats to water quantity and quality (Furgal and Prowse, 2008). Due to their remote location and abundance, our understanding of the individual and cumulative effects of climate change and industrial activities on northern shallow waterbodies remains limited. Consequently, executing an appropriate suite of field-based measurements over sufficient scales of space and time is needed to track responses of shallow waterbodies to multiple potential stressors (Bigg et al., 2017; Heino et al., 2020).

Concern for degradation of shallow waterbodies by effects of climate change and industrial development has garnered intense national and international attention at Wood Buffalo National Park (WBNP) (Schindler and Donahue, 2006; Timoney, 2013; MCFN, 2014; WHC/IUCN, 2017; IEC, 2018; WBNP, 2019). Spanning 45,000 km², across northern Alberta and southern Northwest Territories, WBNP is Canada’s largest National Park (Figure 1.1). The park protects two landscapes with abundant shallow waterbodies, each of which are recognized as a Ramsar Wetland of International Importance and contribute to WBNP’s designation as an UNESCO World Heritage Site (IUCN, 1983). One is the Peace-Athabasca

Delta (PAD), located near the park's southern border. The other, located at the northern end, is the Whooping Crane Summer Range (WCSR).

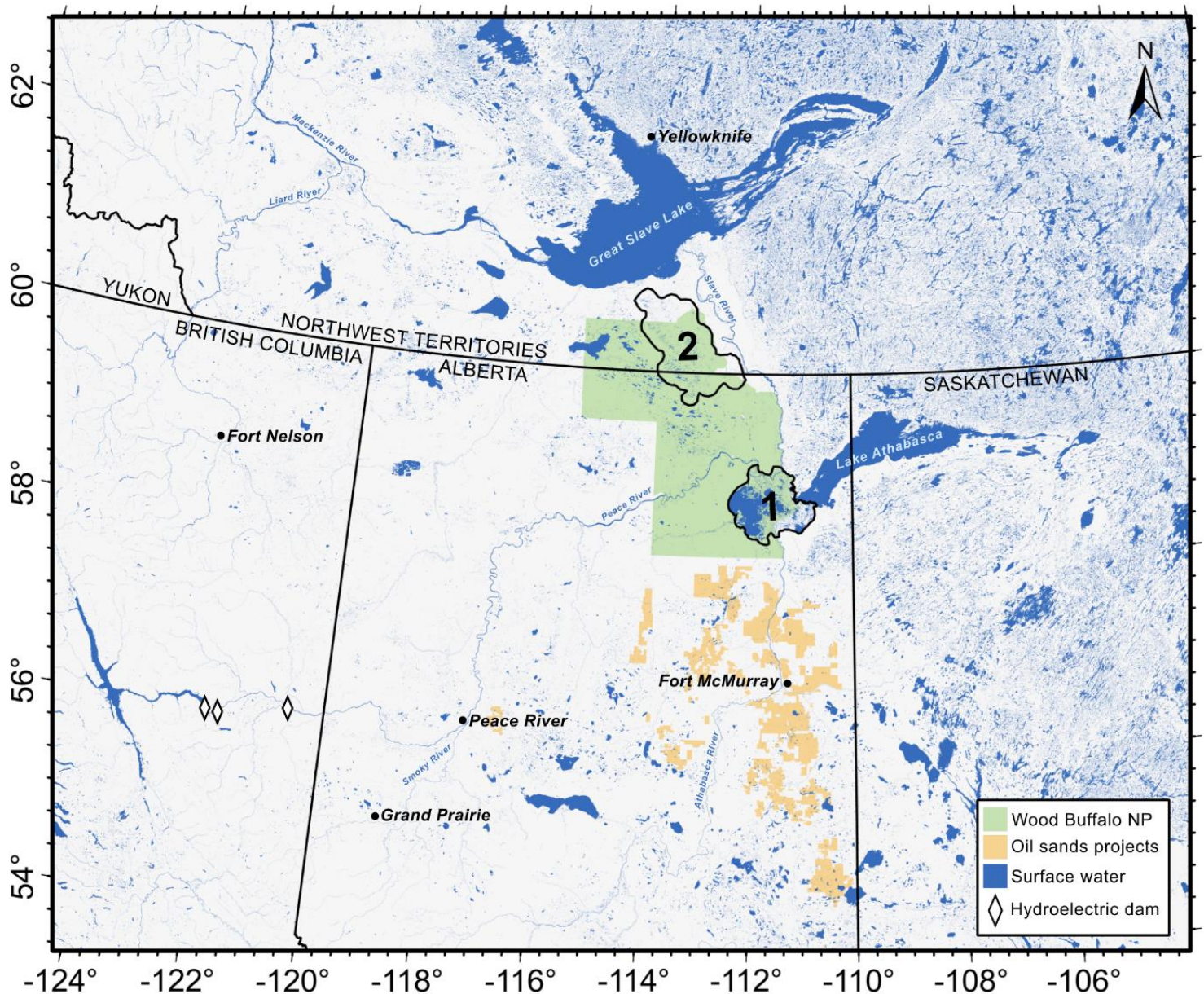


Figure 1.1. Map showing locations of two Ramsar Wetlands in Wood Buffalo National Park (Canada) within northwestern Canada: (1) The Peace-Athabasca Delta, and (2) the Whooping Crane Summer Range.

The PAD is the world's largest freshwater boreal delta and occupies low-relief terrain in northeastern Alberta between the terminus regions of the Peace River and Athabasca River at the western end of Lake Athabasca. Hundreds of shallow lakes and meandering rivers

support productive and diverse habitat for waterfowl and mammals, including the world's largest free-roaming self-regulating wood bison herd (Timoney, 2013). Key hydrological processes that regulate water balance and depth variation of lakes across the PAD include episodic ice-jam and open-water flooding, precipitation (snowmelt and rainfall) and evaporation (Prowse and Conly, 1998, 2000; Beltaos, 2003; Peters et al., 2006a,b, Wolfe et al., 2007; Remmer et al., 2020a,c; Lamontagne et al., 2021). Interaction of groundwater with surface water is not considered to be influential because fine-grained glaciolacustrine sediment from Glacial Lake McConnell and fluvio-deltaic processes has very low conductivity and the delta has low topographic relief (Nielsen, 1972; Prowse and Lalonde, 1996; Peters et al., 2006a). In recent decades, widespread drying of lakes has raised concern over the cumulative effects of multiple potential stressors, and their relative roles remain a focus of attention (e.g., Prowse and Demuth, 1996; Prowse and Conly, 1998, 2000, 2002; Wolfe et al., 2006, 2012, 2020; Beltaos et al., 2014, 2018, 2024; Hall et al., 2019; Lamontagne et al., 2021; Kay et al., 2024a,b; Timoney, 2024). These include hydroelectric regulation of Peace River flow, consumptive water use, climate change, and naturally occurring deltaic processes (PADPG, 1972; Prowse and Conly, 2000; Wolfe et al., 2008, 2012; Kay et al., 2019; Wang et al., 2023; Timoney, 2024). Alberta oil sands mining and processing facilities along the Athabasca River about 200 km upstream of the PAD also raise concern for the risk of accelerated releases of substances of concern which may be delivered to the delta via fluvial and atmospheric pathways (Schindler and Donahue, 2006; Wolfe et al., 2008; Schindler, 2010; Dillon et al., 2011; Timoney, 2013; Lima and Wrona, 2019).

At the WCSR, abundant ponds and wetlands located in karst terrane provide critical breeding habitat for the last self-sustaining migratory population of whooping cranes – the Aransas-Wood Buffalo population (Drake, 1970; Weyer, 1983; McNaughton, 1991; Ford, 1997; Kuyt, 1993; Environment Canada, 2007). This species nearly went extinct in the 1940s due to overhunting and habitat loss (CWS–USFWS, 2007; Austin et al., 2019; ICF, 2019). Intensive conservation efforts have helped increase the population from 16 to over 500 individuals, but the effects of climate warming on water-level variation threatens to undermine this success by degrading critical shoreline breeding habitat (Chavez-Ramirez

and Wehtje, 2012; Butler et al., 2017). For instance, water-level fluctuations during the critical hatching period are believed to negatively affect juvenile recruitment (Kuyt, 1992). Studies by McNaughton (1991) and Déri-Takács et al. (2022) have identified that the ponds are situated within areas where shallow groundwater discharges to the surface, and the water table subsides when precipitation is low. Knowledge of the hydrological processes regulating water balance and depth variation of the ponds and wetlands, however, remains insufficient over space and time to anticipate how the critical shoreline breeding habitat will respond to climate change (Chavez-Ramirez and Wehtje, 2012; WBNP, 2019). Groundwater extraction during the operation of a nearby former zinc and lead mine was once considered a threat to the water table in the WCSR (McNaughton, 1991), and that mine is set to reopen in 2029, pending environmental assessment approval.

Mounting threats to waterbodies in the PAD and WCSR contributed importantly to a petition by the Mikisew Cree First Nation (MCFN) to include Wood Buffalo National Park on the List of World Heritage Sites in Danger (MCFN, 2014). The petition launched a joint Reactive Monitoring Mission by the International Union for Conservation of Nature (IUCN) and UNESCO in 2016 to assess the cumulative environmental threats and impacts on the Outstanding Universal Values of WBNP, which issued several recommendations to the State Party of Canada and Parks Canada Agency for immediate action to prevent a downgrade to the park's World Heritage status (WHC/IUCN, 2017). Of the 17 recommendations, three call for improved monitoring and understanding of the processes that pose threats to aquatic ecosystems in the PAD and WCSR. They include to *“Establish adequate baseline hydrological information of the Peace and Athabasca River Basins to enhance the reference for monitoring and assessing current and future hydrological conditions”* (Recommendation 7), *“ Expand the scope of monitoring and project assessments to encompass possible individual and cumulative impacts on the Outstanding Universal Value of the property and in particular the PAD”* (Recommendation 9), and *“Continue to closely monitor the entire used and potential nesting area of the Whooping Crane within the Greater WBNP Ecosystem so as to be able to respond to possibly changing management requirements”* (Recommendation 16). While these recommendations stress the need for improved monitoring, substantial

primary research is required to effectively guide the design and implementation of long-term aquatic ecosystem monitoring in Wood Buffalo National Park.

The remainder of this thesis is comprised of three primary data chapters, followed by a Synthesis chapter (**Chapter 5**). Two of the data chapters have been published (**Chapter 2** and **Chapter 3**) and one is in review for publication (**Chapter 4**). While structured as stand-alone manuscripts, the research reported in them is based on common sampling methodologies and measurements. Chapters 2 and 3 address the above Recommendations 7 and 9 (WHC/IUCN, 2017), which highlight the longstanding need for an aquatic ecosystem monitoring program in the PAD to track changes by the multiple potential stressors listed above. Hydrological monitoring must serve as the cornerstone of such a program, because drawdown of water levels and lake drying in recent decades by alteration of hydrological processes is a primary concern of stakeholders and rightsholders (MCFN, 2014; Vannini and Vannini, 2019), and hydrological processes exert strong control of lake water chemistry (Wiklund et al., 2012; Remmer et al., 2018) and dispersal of substances of concern (Hall et al., 2012; Wiklund et al., 2014; Kay et al., 2023). The central role of hydrological processes is eloquently articulated by Timoney (2013, p. 5), “In the delta, water is boss, change is the only constant, and creation and destruction exist side by side”. Systematic and repeated seasonal collection of lake water isotope compositions performed by others in our research program at a landscape scale in spring, summer and fall since 2015 has provided exceptional insight into the spatial and temporal variability of hydrological processes across the PAD, including delineation of the extent of ice-jam and open-water flooding, and the identification of ‘hotspots’ in the landscape where evaporation exceeds inflow (Remmer et al., 2018, 2020a,c). Interpolated maps of lake evaporation-to-inflow (E/I) ratios that we refer to as ‘isoscapes’ (Bowen and Revenaugh, 2003; West et al., 2010; Bowen and Good, 2015) were generated from over 1,000 isotope measurements taken at 60 lakes during 2015 to 2019, which provides a foundation for ongoing hydrological research and monitoring in the PAD (Remmer et al., 2020a). While the insights gained from measurement of water isotope compositions have proven to be highly effective for tracking spatiotemporal variability of key hydrological processes and lake water balances, short-lived but influential hydrological

processes, such as intense rainfall and open-water flood events, may be missed (or their effects may be underestimated) if sampling occurs once per season. This limitation was addressed by deploying continuously recording water depth loggers in lakes spanning the hydrological gradients of the delta in 2018 and 2019 (**Chapter 2**). Objectives of this research were to elucidate the timing and magnitude of lake-level responses to key hydrological processes during the open-water season and to refine hydrological categorization of lakes in the PAD. Also, I integrated the lake-level measurements with river hydrometric data and a digital elevation model to define the geographical area of the delta that was inundated by open-water flooding in 2018 and 2019, which improved upon estimates reported in Remmer et al. (2020c) based on seasonal measurements of water isotope composition. Lastly, long-term river hydrometric data were used to assess the frequency of open-water season flooding by the Athabasca River, a hydrological process that has long been under-recognized.

Chapter 3 builds forward from Chapter 2 and was motivated by the value of compiling and synthesizing information generated from systematic, repeated hydrological and contaminants measurements made during 2015-2021 at 60 lakes across the delta published across five journal articles (e.g., Remmer et al., 2020a,c, Owca et al., 2020; Savage et al., 2021; Neary et al., 2021b [**Chapter 2**]), and unpublished lake water chemistry measurements to identify aquatic ecosystem changes and the probable causes, and inform sampling approaches and metrics for a long-term lake monitoring program at the PAD and other hydrologically complex landscapes with abundant shallow waterbodies.

During a field sampling campaign at the PAD, a conversation between my supervisors and the Warden of WBNP sparked an invitation to apply the methodologies developed for lakes in the PAD to ponds and wetlands in the WCSR to improve understanding of the hydrological processes that influence water-level variations and inform how whooping crane conservation strategies may need to respond to ongoing climate change. This opportunity allowed me to contribute to the study design and collaborate with local community members, government agencies, and other stakeholders to gain support for a new 5-year research program launched in 2022 in the WCSR. **Chapter 4** of this thesis presents the first

foundational year of research when I used a suite of methodologies demonstrated to be effective at shallow waterbodies in the PAD and other northern freshwater-rich landscapes (e.g., Gibson and Edwards, 2002; Gibson et al., 2005, 2022; Brock et al., 2007; Turner et al., 2014; Arp et al., 2015; MacDonald et al., 2017, 2021; Stadnyk et al., 2023) at a substantial number of shallow ponds and wetlands in the Sass-Klewi Nesting Area (SKNA) of the WCSR, where whooping crane nest density is highest. In Chapter 4, I integrate measurements of water isotope composition, depth, and chemistry at 63 well-dispersed shallow ponds and wetlands in 2022 to assess the roles of inputs from snowmelt, rainfall and groundwater, and losses via evaporation and water table subsidence on water balances and pond depth during a year with unusually thick snowpack, delayed onset of the freshet and below-average rainfall.

Chapter 5 presents the key findings generated using comparable methodologies at shallow waterbodies in the two hydrologically complex freshwater landscapes where some of the dominant hydrological gradients differ (floodwater connectivity at the PAD, groundwater connectivity at the WCSR) to improve understanding of their responses to multiple stressors. I inform about the strengths and limitations of the methodologies utilized, and explore implications of the research beyond those presented in the individual chapters for the long-term monitoring and management of shallow waterbodies in Wood Buffalo National Park. I also situate my individual and collective works in a Research-to-Inform Monitoring Continuum where key phases of scientific research are illustrated as the basis to inform the design and implementation of long-term aquatic ecosystem monitoring. This thesis contains four appendices that provide supplementary materials for Chapter 3 (**Appendix 1**), Chapter 4 (**Appendix 2**) and Chapter 5 (**Appendix 3.1** and **Appendix 3.2**). These appendices include Technical Reports that document methodologies, key findings and standard operating procedures (SOPs) for Parks Canada Agency, which served as stepping stones for published work presented in this thesis. I hope that many aspects of this research reach stakeholders and rightsholders in WBNP and helps to inform the design and implementation of monitoring program capable of tracking responses of shallow waterbodies to complex interactions

between climate change and other stressors in the Peace-Athabasca Delta, Whooping Crane Summer Range, and elsewhere.

Chapter 2 | A new lake classification scheme for the Peace-Athabasca Delta (Canada) characterizes hydrological processes that cause lake-level variation

Citation: Neary, L.K., Remmer, C.R., Krist, J., Wolfe, B.B. and Hall, R.I. (2021). A new lake classification scheme for the Peace-Athabasca Delta (Canada) characterizes hydrological processes that cause lake-level variation. *Journal of Hydrological Processes: Regional Studies* **38**, 100948. <https://doi.org/10.1016/j.ejrh.2021.100948>.

2.1 Introduction

Water security is a rising concern in western Canada, where climate warming has reduced snowpack and glacier volumes and altered the timing and amount of streamflow to downstream ecosystems (Bonsal et al., 2019; Debeer and Sharp, 2007; Hugonnet et al., 2021; Sauchyn et al., 2015; Tennant and Menounos, 2013; Wolfe et al., 2008). At the Peace-Athabasca Delta (PAD) in northeastern Alberta, major energy projects upstream pose additional potential threats to security of water supply (Lebel et al., 2011; Prowse and Conly, 1996, 2000; Prowse et al., 2006; Schindler and Donahue, 2006). These include regulation of Peace River flow for hydroelectric production and withdrawal of Athabasca River water within the Alberta oil sands mining region. The delta's waterways provide important access to traditional territory of Indigenous people, and the hundreds of small (<1 km²), shallow lakes furnish productive habitat for a diversity of wildlife where lake-level fluctuations influence population density (Hood, 2020; Straka et al., 2018; Vannini and Vannini, 2019; Ward et al., 2018, 2020). Eighty percent of the PAD is protected within Wood Buffalo National Park (WBNP), a UNESCO World Heritage Site, and the delta is a Ramsar Wetland of International Importance. Much of the remaining area is the traditional territory of the Athabasca Chipewyan First Nation (IR 201). Recent observations of lake drying and decline of river levels have led to a petition to include WBNP on UNESCO's List of World Heritage in Danger (MCFN, 2014), recommendations to improve understanding and causes of lake-level variations over space and time (WHC/IUCN, 2017), and an action plan to implement research and monitoring approaches to assess cumulative impacts and inform decision-making by stakeholders (WBNP, 2019).

The abundant shallow lakes of the PAD have been distinguished into three main hydrological categories (open-drainage, restricted-drainage, closed-drainage) based on differences in their connectivity to the river channel network, which exerts strong control on lake water balance in the low-relief terrain (PADPG, 1973; Peters et al., 2006; Pietroniro et al., 1999; Prowse and Demuth, 1996; Timoney, 2013; Wolfe et al., 2007). Open-drainage lakes occur at the lowest elevations and receive continuous to near-continuous river through-flow. Closed-drainage lakes occupy the highest elevations, are isolated from the channel network, and are thought to receive floodwaters only during episodic rise of water to extreme levels caused by ice-jam events on the Peace and Athabasca rivers and their tributaries. During intervals between ice-jam floods, maintenance of their water levels relies on input from snowmelt and rainfall to the lake and catchment to offset losses by evaporation. Restricted-drainage lakes have been characterized as occurring at intermediate elevations and they experience more frequent connectivity to rivers during ice-jam floods as well as periods of open-water elevated river flow that promote intermittent reconnection to distributary channels. The restricted-drainage category, thus, includes a broad lake class between the extremes of open- and closed-drainage lakes, and are presumed to be less prone to desiccation than closed-drainage lakes. ‘Perched basin’ is a frequently used term that includes both restricted- and closed-drainage lakes to distinguish isolated lakes in the landscape from open-drainage lakes connected to the river network, and to acknowledge a hydrological continuum exists. A hydro-limnological survey by Wolfe et al. (2007) further defined a subset of shallow perched basins as rainfall-influenced, based on their shallow depth and small water volume that promotes rapid desiccation by evaporation and refilling by rainfall events.

The three main hydrological categories described above were established decades ago during initial field surveys of the PAD when understanding of the hydrology of the landscape was incomplete. Systematic sampling of water isotope composition during spring, mid-summer and fall of a recent 5-year period (2015-2019) at a network of ~60 lakes and 9 river sites across the PAD has vastly improved our knowledge of hydrological processes regulating lake water balance and provided information on spatio-temporal patterns of variation in the

influence of river floodwaters, precipitation and evaporation on lake water balance (Remmer et al., 2020a). Results demonstrate greater heterogeneity of lake water balance than can be captured by the conventional use of the three hydrological lake categories. For example, water balance of restricted-drainage lakes can vary across the delta from positive to negative at the same time, and this also occurs for closed-drainage lakes. Seasonal and inter-annual variation is sufficiently high that perched basins may exhibit water balance characteristic of the closed-drainage category in some seasons and years and characteristics of the restricted-drainage category in other seasons and years. Notably, the systematic water isotope monitoring during summers of 2017-2019 captured recurring evidence of strongly positive lake water balance across central and southern (Athabasca sector) portions of the PAD due to input of river floodwaters during the open-water season. Among the extensive body of hydrological literature on the PAD, this mechanism of lake replenishment has rarely been reported, except for Peters et al. (2006, 2021) who also identified the effectiveness of high river-level events during the open-water season in recharging slightly elevated lakes in the Athabasca sector. Classification of lakes into open-, restricted- and closed-drainage categories, thus, does not explicitly distinguish lakes that are strongly influenced by open-water flooding from those influenced by ice-jam flooding or lateral connectivity stemming from other hydrological processes associated with the river channel network.

Knowledge of open-water flooding in the Athabasca sector, and influence of other hydrological processes on the delta's lakes, may be advanced by using water-level loggers to obtain continuous, real-time measurements of lake-level fluctuations. Such continuous measurements may be used to pinpoint the timing, magnitude and extent of open-water flooding and its influence on lake levels, metrics that cannot easily be captured using 'snapshot' measurements of water isotope tracers alone when periodic sample collection does not coincide with influential hydrological processes and events. Fergus et al. (2020) demonstrated the value of supplementing water isotope tracer information with singular measurements of lake-level drawdown based on high-water marks to develop a national-scale hydrological baseline for lakes across the conterminous United States. To our knowledge, however, water-level loggers have not been deployed for continuous lake-level

measurements at multiple sites across a floodplain landscape as dynamic as the PAD, yet they present an approach that can broaden and enhance our understanding of influential hydrological processes and contribute useful information for lake monitoring programs.

Here, we utilize continuous hourly lake-level measurements at 48 lakes in 2018 and 53 lakes in 2019 across the PAD to elucidate the timing and magnitude of lake-level responses to key hydrological processes during the open-water season. The knowledge gained is used to refine hydrological categorization of lakes in the PAD. We also integrate the lake-level measurements with river hydrometric data and a digital elevation model to define the geographical area of the PAD inundated by open-water flooding in 2018 and 2019, thus improving upon estimates reported in Remmer et al. (2020a) based on three ‘spot’ measurements of water isotope composition per year. We also use long-term river hydrometric data to speculate on the frequency of open-water season lake flooding by the Athabasca River during the past four decades. Finally, we assess the utility of water-level loggers as a complementary lake monitoring tool to water isotope tracers and other approaches.

2.2 Study Area

2.2.1 Peace-Athabasca Delta

Spanning an area of ~6000 km², the PAD is the world’s largest freshwater boreal delta (Figure 2.1; Peters et al., 2006). Elevational differences between lakes and rivers, and the complex hydrological pathways connecting them, create three distinctive hydrological ‘regions’ across the PAD: the northern relic Peace sector, the southern active Athabasca sector, and the central area occupied by large open-drainage lakes (Lake Claire, Mamawi Lake; PADPG, 1973; Wolfe et al., 2007). Based on the Canadian Digital Elevation Model (CDEM; accuracy = 0-10 m for the region), the light blue region spanning the central area of the delta in Figure 2.1 represents areas between 205-210 masl, and broadly corresponds with the spatial extent of the ‘open-drainage network’. This includes lakes Claire, Mamawi and Richardson (PAD 38), and the channels that connect them with the Athabasca and Peace rivers and the western end of Lake Athabasca (Timoney, 2013). The Peace sector has greater topographic relief than the Athabasca sector and possesses numerous bedrock inliers (>225 masl) that

occur in the catchments of many lakes. Comparatively, only a narrow area of land in the southern margin of the Athabasca sector along the Athabasca River is elevated above 220 masl.

The directional movement of water in the delta is influenced by variation in water levels of the Athabasca and Peace rivers and Lake Athabasca (Figure 2.1). Water enters the southern Athabasca sector via the north-flowing Athabasca River and its distributaries, including the

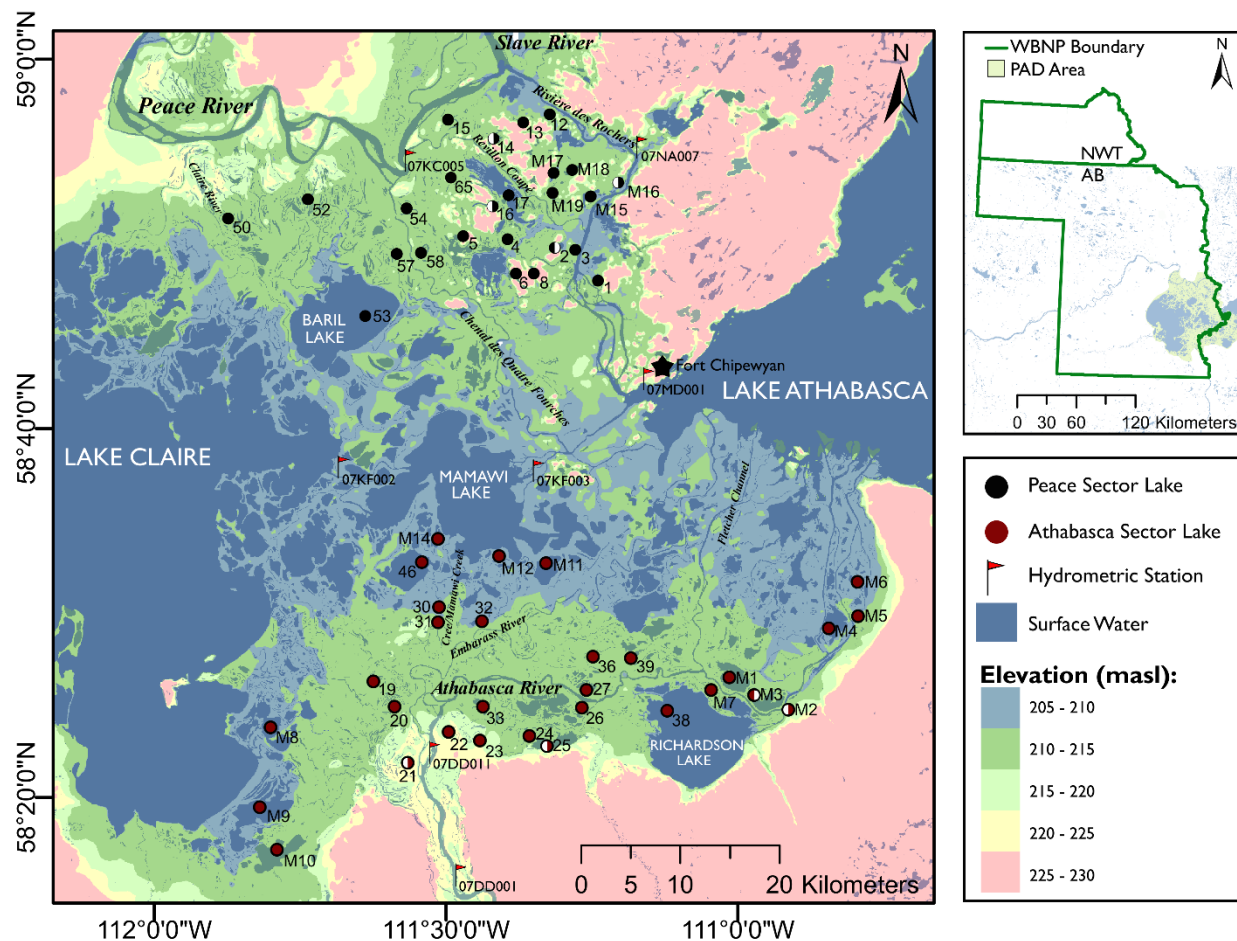


Figure 2.1. Map showing locations of lake sampling sites within the Peace-Athabasca Delta, northern Alberta. Black circles identify lakes located in the Peace sector ($n = 25$) and red circles identify lakes located in the Athabasca sector ($n = 30$). Semi-circles indicate the year in which the lake-level logger was retrieved: left-filled semi-circle identify logger retrieved only in 2018, right-filled semi-circle identify logger retrieved only in 2019 and solid circles identify logger retrieved in both 2018 and 2019. Hydrometric stations operated by Environment and Climate Change Canada are indicated with red flags. Figure was created using ArcGIS Desktop version 10.7.1 and assembled by Laura Neary using the following data sources (shapefiles): Government of Canada (national park boundaries: <https://open.canada.ca/data/en/dataset/9e1507cd-f25c-4c64-995b-6563bf9d65bd>; water bodies: <https://open.canada.ca/data/en/dataset/448ec403-6635-456b-8ced-d3ac24143add>; geospatial CDEM: <https://maps.canada.ca/czs/index-en.html>), PAD area: shapefile generated by Jelle Faber.

Embarras River that carries water northward via Cree/Mamawi Creek into Mamawi Lake or via Fletcher Channel into Lake Athabasca. The Athabasca River and Lake Athabasca are typically at a higher elevation than the Peace River. Thus, for most of the time channels in the Peace sector (Rivière des Rochers, Chenal des Quatre Fourches, Revillon Coupé) carry outflow from Lake Athabasca northward to the Peace River and, ultimately, to the Slave River. In early spring, southward flow reversals can occur in these channels when Peace River water levels rise and, more typically, when ice-jams form.

2.2.2 Study lakes

The lakes from which water-level loggers (i.e., pressure transducers) were retrieved at the end of the open-water seasons ($n_{2018} = 48$, $n_{2019} = 53$) span surface areas from approximately $<1 \text{ km}^2$ to 38 km^2 (Richardson Lake / PAD 38) and are roughly divided between the Peace sector ($n_{2018} = 22$, $n_{2019} = 24$) and the Athabasca sector ($n_{2018} = 26$, $n_{2019} = 29$; Figure 2.1, Table 2.1). Most (36/53) of the lake sites studied were included in a survey conducted in fall of 2000 by Wolfe et al. (2007). The other 17 lakes (lakes M1 to M19) are a subset also sampled by community-based monitoring groups to track variation in muskrat abundance. The entire lake set is largely equivalent to that reported in Remmer et al. (2020a) and spans the range of lake hydrological conditions in the delta.

2.2.3 Meteorological conditions and spring ice-jam flooding in 2018 and 2019

Based on the 1981-2010 climate normal measured at the Fort Chipewyan airport station, mean monthly air temperatures range from -20°C to 15°C and average annual precipitation is $\sim 365 \text{ mm}$ (Figure 2.2). Monthly temperatures during the two study years were similar to the climate normal with the following exceptions. In May-June 2018, average monthly air temperatures were warmer than the climate normal. This was followed by cooler-than-average air temperature in September. In 2019, air temperature was colder in February and warmer in March than the climate normal. Cumulative precipitation in 2018 (345 mm) was 20 mm less than the climate normal. Cumulative precipitation in 2019 (443 mm) exceeded the annual climate normal by late September. During the open-water season, substantial

Table 2.1. Summary of mean, initial, minimum, maximum and maximum change of lake depth (above logger) in the Peace-Athabasca Delta for open-water seasons (May–September) of 2018 and 2019. Lakes that received spring floodwater in 2018 are indicated with an asterisk and in spring 2019 are bolded. Lake site names and their coordinates: <https://doi.org/10.5683/SP2/D5TVYG>.

Site	Mean (m)		Initial (m)		Minimum (m)		Maximum (m)		Max. Change (cm) ^a		Lake-level Pattern ^b		Drainage Category ^c	
	2018	2019	2018	2019	2018	2019	2018	2019	2018	2019	2018	2019		
Peace sector														
PAD 1	1.10	0.80	1.09	0.89	1.00	0.71	1.24	0.91	-9.45	-17.16	S	D	c	
PAD 2	1.10		1.18		0.99		1.19		-19.02		D		c	
PAD 3	0.68	0.38	0.76	0.41	0.54	0.33	0.80	0.47	-22.56	-8.41	D	S	c	
PAD 4	0.64	0.44	0.71	0.54	0.50	0.33	0.78	0.55	-20.24	-21.03	D	D	c	
PAD 5	1.14	0.88	1.19	0.96	1.03	0.81	1.20	1.00	-15.48	-14.81	D	S	c	
PAD 6	1.16	0.86	1.16	0.90	1.04	0.77	1.29	0.95	-11.80	-13.05	S	S	c	
PAD 8*	1.43	1.25	1.40	0.96	1.15	0.88	1.83	1.75	+64.13	+71.11	GR	GR	r	
PAD 12	0.52	0.19	0.60	0.32	0.39	0.07	0.63	0.34	-21.43	-25.60	D	D	c	
PAD 13	1.44	1.22	1.48	1.30	1.36	1.13	1.49	1.31	-11.46	-16.70	S	D	r	
PAD 14*		1.32		1.45		1.24		1.48		-21.09		D		r
PAD 15*	1.38	0.90	1.37	0.99	1.32	0.81	1.44	1.00	-5.48	-17.59	S	D	c	
PAD 16		0.38		0.43		0.28		0.48		-14.36		S		r
PAD 17	1.72	0.39	1.74	0.46	1.67	0.30	1.76	0.47	-7.06	-15.61	S	D	r	
PAD 50*	0.42	0.07	0.46	0.19	0.29	0	0.55	0.19	-16.61	-18.50	D	D	c	
PAD 52	0.89	0.53	0.91	0.59	0.74	0.43	1.00	0.61	-17.04	-15.79	D	D	r	
PAD 53	0.95	0.57	0.93	0.63	0.67	0.46	1.12	0.76	-25.88	-16.80	S	D	p	
PAD 54*	3.64	1.88	3.54	1.85	3.54	1.84	3.71	1.92	-0.58	-0.46	S	S	r	
PAD 57	0.93	0.63	1.02	0.78	0.79	0.50	1.03	0.79	-23.32	-27.55	D	D	c	
PAD 58*	0.67	0.31	0.70	0.41	0.53	0.20	0.75	0.42	-17.04	-21.03	D	D	c	
PAD 65	1.08	0.82	1.09	0.88	1.01	0.74	1.13	0.89	-7.77	-14.08	S	S		
M15	1.12	0.61	1.15	0.67	1.05	0.54	1.17	0.69	-9.75	-13.78	S	S		
M16		0.92		1.00		0.84		1.02		-16.22		D		
M17*	1.04	0.72	1.08	0.79	0.94	0.63	1.11	0.82	-13.72	-15.79	S	D		
M18*	1.37	1.11	1.45	1.19	1.28	1.01	1.45	1.21	-17.10	-18.32	D	D		
M19	1.09	0.71	1.13	0.77	1.00	0.63	1.15	0.80	-12.95	-14.33	S	S		
Athabasca sector														
PAD 19*	1.06	1.00	1.21	1.06	0.91	0.89	1.21	1.09	-29.66	-17.53	D	D	c	
PAD 20*	1.87	0.53	2.11	0.71	1.67	0.41	2.11	0.71	-43.56	-29.87	D	D	r	
PAD 21*		0.61		0.71		0.49		0.72		-22.07		D		r
PAD 22*	1.89	0.88	1.86	0.86	1.78	0.79	2.01	1.00	-7.40	-7.13	S	S	r	
PAD 23*	2.05	1.20	2.18	1.27	1.91	1.09	2.19	1.28	-27.46	-17.74	D	D	r	
PAD 24*	1.43	1.28	1.52	1.12	1.26	1.03	1.52	1.96	-25.66	+89.18	D	SR	r	
PAD 25*		1.23		0.90		0.82		2.24		+138.53		SR		o
PAD 26*	1.06	1.32	1.32	0.94	0.83	0.83	1.78	2.41	+88.42	+154.59	SR	SR	o	
PAD 27*		1.28		1.27		1.21		1.42		-6.07		S		r
PAD 30*	1.03	0.79	1.06	0.69	0.92	0.56	1.44	1.41	+45.48	+79.74	SR	SR	r	
PAD 31*	1.17	1.23	1.22	1.01	1.03	0.90	1.81	2.12	+68.03	+117.10	SR	SR	r	
PAD 32*	1.16	0.89	1.20	0.97	1.03	0.81	1.27	0.98	-17.10	-16.34	D	D	r	
PAD 33*	1.22	1.13	1.62	0.40	0.70	0.33	2.13	2.52	+141.09	+179.83	SR	SR	r	
PAD 36*	1.23	0.98	1.44	0.94	1.11	0.88	1.44	1.11	-33.56	-6.31	D	S	r	
PAD 38*	0.99	1.35	1.26	0.90	0.50	0.64	1.41	2.24	+56.51	+140.79	SR	SR	o	
PAD 39*	1.22	0.90	1.48	0.94	1.05	0.81	1.48	0.94	-43.16	-12.68	D	S	r	
PAD 46*	1.47	1.33	1.79	0.89	1.21	0.81	1.79	2.13	+46.45	+121.71	SR	SR	o	
M1	1.15	1.21	1.24	1.11	1.05	1.03	1.25	1.40	-19.17	+31.85	D	SR		
M2		1.28		1.31		1.20		1.34		-10.49		S		
M3	0.63		0.71		0.54		0.72		-16.58		D			
M4	0.61	0.37	0.69	0.48	0.51	0.29	0.69	0.50	-18.50	-19.23	D	D		
M5	1.21	1.22	1.19	1.20	1.17	1.10	1.24	1.50	-2.62	-9.45	S	S		
M6	0.60	0.41	0.66	0.46	0.50	0.30	0.72	0.53	-16.03	-15.48	D	D		
M7	1.06	1.06	1.26	0.86	0.92	0.72	1.34	1.63	+33.28	+84.52	SR	SR		
M8	1.01	0.76	0.89	0.87	0.88	0.65	1.45	0.94	+45.10	-22.07	GR	D		
M9	0.89	0.65	0.85	0.75	0.80	0.56	1.06	0.79	+19.57	-18.75	GR	D		
M10	1.30	0.63	1.53	0.81	1.03	0.45	1.60	1.05	+51.24	+39.56	GR	GR		
M11	0.77	0.63	0.70	0.71	0.68	0.57	0.84	0.71	-2.10	-14.36	S	S		
M12*	1.23	1.09	1.24	0.98	1.13	0.91	1.35	1.28	+16.49	+37.12	GR	SR		
M14*	1.30	1.30	1.55	1.11	1.14	1.01	1.80	1.75	+41.54	+69.16	GR	SR		

^aMaximum Change (in centimeters) was determined as Minimum - Initial for Drawdown and Stable lakes and as Maximum - June 14 water level for Gradual Rise and Sharp Rise lakes; ^bWater-level pattern defines the main water-level trends observed in Figure 2.3 and 2.4 (D: drawdown; S: stable; GR: gradual rise; SR: sharp rise); ^cHydrological drainage categories based on isotopic and limnological data (c: closed; r: restricted; o: open; p: very shallow (Z_{max} < 50 cm) rainfall-influenced) from Wolfe et al. (2007). Sites without drainage designation were not included in Wolfe et al. (2007).

rainfall occurred in June 2018 (61 mm) and August 2019 (106 mm). Approximately 45 mm of rain was measured on August 14, 2019, which is far more than any other single day and unusual for the region.

Widespread ice-jam flooding in late April – early May 2018 delivered river floodwaters to nearly half of all lake sites (Table 2.1). The extent of river floodwaters was delineated by Remmer et al. (2020b) and includes a large central area within the Athabasca sector and small areas of the northern Peace sector. Of the lakes where we deployed water-level loggers, this event flooded 19 lakes (73%) in the Athabasca sector, and 8 lakes (36%) in the Peace sector. Spring flooding was more limited in 2019 and was detected in 8 (28%) of the Athabasca sector lakes and no lakes in the Peace sector where loggers were deployed (Table 2.1; Remmer et al., 2020a).

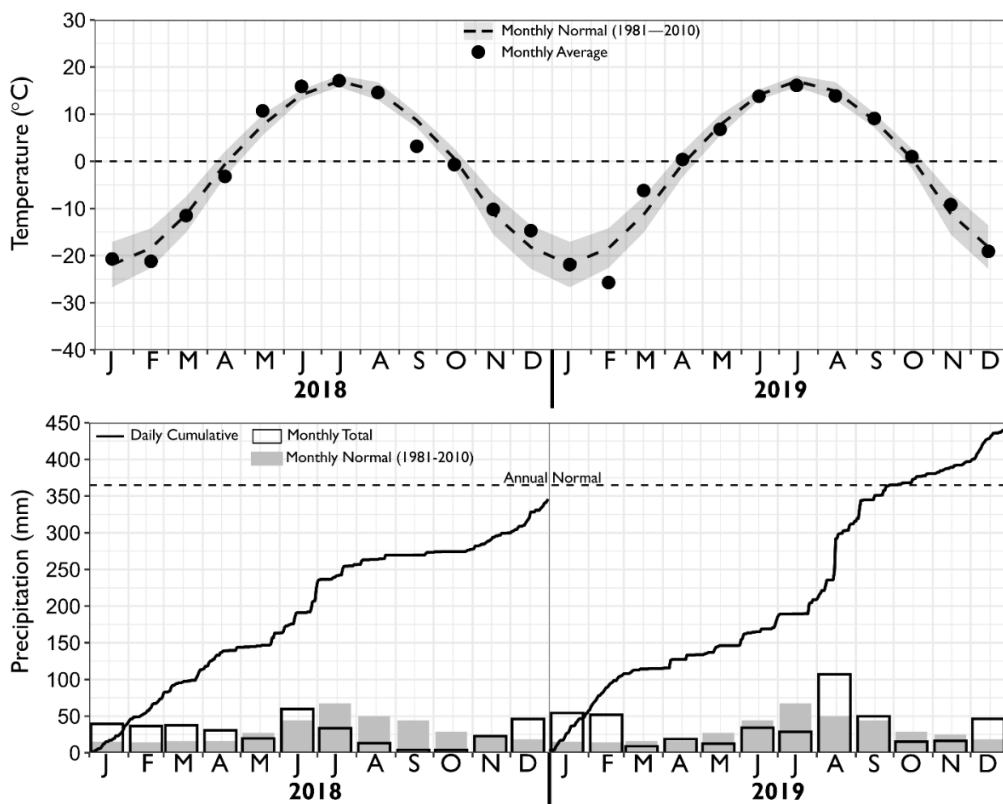


Figure 2.2. Meteorological data for January 1, 2018 to December 31, 2019 recorded at the Fort Chipewyan Airport, Fort Chipewyan, Alberta, Canada (ID: 3072659). The upper panel includes average monthly air temperatures (°C) with respect to the monthly climate normal (1981–2010; ID: 3072658) identified by a black dashed line surrounded by grey area representing standard deviations. The lower panel includes cumulative daily precipitation (mm) for 2018 and 2019 represented by the solid black line and black-outlined bars representing monthly total precipitation, with respect to the climate normal (1981–2010) precipitation (grey bars). The annual climate normal (1981–2010) precipitation is identified by a horizontal dashed line.

2.3 Methods

2.3.1 Field methods

Water-level loggers (Onset HOBO pressure transducers; Model U20-001-01, 4-m range, 0.3-cm accuracy) were installed at a central location of each lake in mid-May 2018 and 2019, shortly after ice-off (Figure 2.1). Most study lakes have a simple bathymetry where maximum depth occurs near the center of the lake. Exceptions include oxbow lakes (PAD 15, 54), which are remnant river channels and where the thalweg is likely closer to the outer bank. For these two lakes, we deployed loggers equidistant from the shorelines on one of the arms, which are not at their deepest location. Each logger was attached to a ‘rock-sock’ anchor by zip-tie and a wooden float by nylon rope. Water pressure (psi) and temperature (°C) were recorded every hour until removed in mid-September of each year. GPS coordinates were logged immediately after each device was deployed to aid their retrieval and guide re-deployment at a consistent location the following year. In both years, a single logger was also deployed outside the field house in Fort Chipewyan to measure variation in local atmospheric pressure and air temperature. These data were used to convert water pressure readings at each lake to water depth (m) using the Barometric Compensation Data Assistant in the HOBOWare software. RStudio v1.3.1073 (R Core Team, 2019) software and external packages (*pracma*, *lubridate*, *ggplot2*) were used to calculate and graph daily (24-hour) running means (Borchers, 2019; Grolemund and Wickham, 2011; Wickham, 2016). From this point forward, we simply refer to the water depth measurements as ‘lake level’ or ‘lake-level patterns or variation’.

2.3.2 River hydrometric data and Canadian digital elevation model

Water-level data were obtained from hydrometric stations maintained by the Water Survey of Canada (<https://wateroffice.ec.gc.ca/>) for river sites (Athabasca River (07D001, 07DD011), Peace River (07K005), Rivière des Rochers (07NA007)), Lake Claire (07KF002), Mamawi Lake (07KF003), and Lake Athabasca (07KF003) to compare to our lake-level data (Figure 2.1). Hydrometric data for the Peace River were unavailable during early spring of 2018 because river ice damaged the hydrometric station.

A Canada-wide digital elevation model (CDEM) was included as a map layer in ArcMap 10.7.1 to explore and visualize relations between elevation and spatial distribution of lake-level patterns. The CDEM was created in 2016 by the Department of Natural Resources Canada and is accessible online at: <https://maps.canada.ca/czs/index-en.html>. The coverage and resolution of the CDEM varies according to latitude and uses the North American Datum 1983 as a reference system for horizontal coordinates. Elevations are orthometric and expressed relative to mean sea level (Canadian Geodetic Vertical Datum of 1928). For the study region, the CDEM offers a spatial base resolution of 0.75" x 0.75" arc seconds and a measured altimetric accuracy of 0-10 meters.

2.3.3 Classification of lake-level patterns

Based on visual observation of individual lake-level patterns, geographical location and simple numerical analyses, study lakes were sorted into the following four categories for 2018 and 2019: 1) 'drawdown' (≥ 15 cm decline), 2) 'stable' (< 15 cm decline), 3) 'gradual rise' and 4) 'sharp rise'. Lake-level decline was calculated as the minimum lake level subtracted by the initial lake level at the time of logger deployment (Table 2.1). Initial lake level was selected as the benchmark, rather than maximum lake level, because the latter is time dependent and varies from lake to lake. The threshold used to separate lakes into the drawdown versus stable categories (15 cm) was based on approximation of the median drawdown for lakes in 2018 and 2019 (17.0 cm and 16.2 cm, respectively; Figure 2.3). Temporal patterns of 'gradual rise' and 'sharp rise' were visually discernable as lake levels that increased slowly or quickly, respectively. For study lakes located adjacent to large open-drainage lakes (i.e., Claire and Mamawi), we also visually assessed for lake-level correlation between the study lake and adjacent open-drainage lake as evidence of gradual rise due to inundation by rising levels in the open-drainage network. Estimates of maximum lake-level change for gradual and sharp rise lakes (Table 2.1) were obtained from the lake-level maximum subtracted by the lake level measured on June 14th for both years. June 14th was selected to minimize the potential effects from spring flooding and is ~24 hours before the earliest lake-level rise associated with open-water flooding.

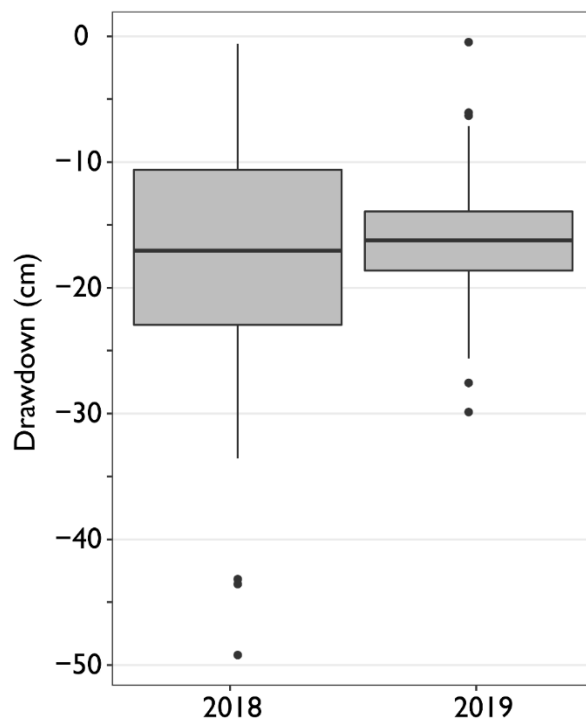


Figure 2.3. Boxplot showing variation in maximum drawdown of lake level in 2018 and 2019 for lakes that did not display a gradual or sharp rise.

2.4 Results

Lakes in both sectors of the PAD displayed lake-level variations of the stable, drawdown and gradual rise categories during 2018 and 2019 (Figures 2.4, 2.5). However, only lakes in the Athabasca sector exhibited the sharp-rise pattern in lake levels during 2018 and 2019 (Figures 2.4, 2.5). Given this difference and previous research demonstrating lake water balances contrast greatly between the Peace and Athabasca sectors (Wolfe et al., 2007; Remmer et al., 2020a), we present and describe the results separately for each sector. We also report spatial distribution of lake-level patterns with respect to land-surface elevations and proximity to active river channels.

2.4.1 Peace sector

Including both study years, 25 lakes in the Peace sector fell into the drawdown category, which is more than any of the other categories (Figure 2.4A, B). Ten lakes (45.5%; PAD 2, 3, 4, 5, 12, 50, 52, 53, 57, 58) displayed drawdown in 2018, and 15 lakes (68.2%; PAD 1, 4, 12, 13, 14, 15, 17, 50, 52, 53, 57, 58, M16, M17, M18) in 2019. Seven of the lakes fell into the drawdown category in both years. All but one of the drawdown lakes is elevated above 210

masl (Figure 2.6). Baril Lake (PAD 53) is the sole exception located within the 205-210 masl range. Baril Lake also shows higher frequency variation compared to other lakes in this category possibly due to its larger fetch that generates greater wind-driven lake-level fluctuation. In 2018, lake-level decline began at lakes in this category in early July and continued until loggers were retrieved in mid-September. In contrast, lake-level decline began earlier in 2019 (mid-May) and ended earlier (mid-August) when substantial rainfall coincided with a sudden small rise in lake levels (4.8 ± 1.2 cm). Among the lakes in this category, PAD 50 drew down sufficiently to expose the logger to the atmosphere after mid-July 2019. Two of the 10 drawdown lakes in 2018 shifted to the stable category in 2019 (PAD 3, 5).

The next most common lake-level category in the Peace sector was stable (Figure 2.4C, D). In 2018, 11 lakes (50.0%; PAD 1, 6, 13, 15, 17, 54, 65, M15, M17, M18, M19) exhibited this behaviour. Of these, four lakes (PAD 15, 54, M17, M18) received ice-jam floodwaters in the spring (Remmer et al., 2020b). Eight lakes (33.3%; PAD 3, 5, 6, 16, 54, 65, M15, M19) fell in this category in 2019 when spring flooding was less extensive. Lakes in the stable category are situated east of the Chenal des Quatre Fourches and span a range of elevations (205-225 masl; Figure 2.6). Lakes in the stable category displayed a sudden small rise (5.0 ± 0.9 cm) coincident with the major rain event in mid-August 2019, as observed for lakes in the drawdown category. Six of the 11 stable lakes in 2018 shifted to the drawdown category in 2019 (PAD 1, 13, 15, 17, M17, M18).

One lake in the Peace sector, PAD 8, exhibited a gradual rise in lake level, which coincided with the rise and fall of Lake Athabasca and its outlet channel, the Rivière des Rochers, but not with variation of Peace River levels (Figure 2.4E-H). This lake is connected to the Rivière des Rochers via a single distributary channel (Chilloney's Creek), which serves as both an inflow or outflow depending on relative lake levels of PAD 8 and the Rivière des Rochers.

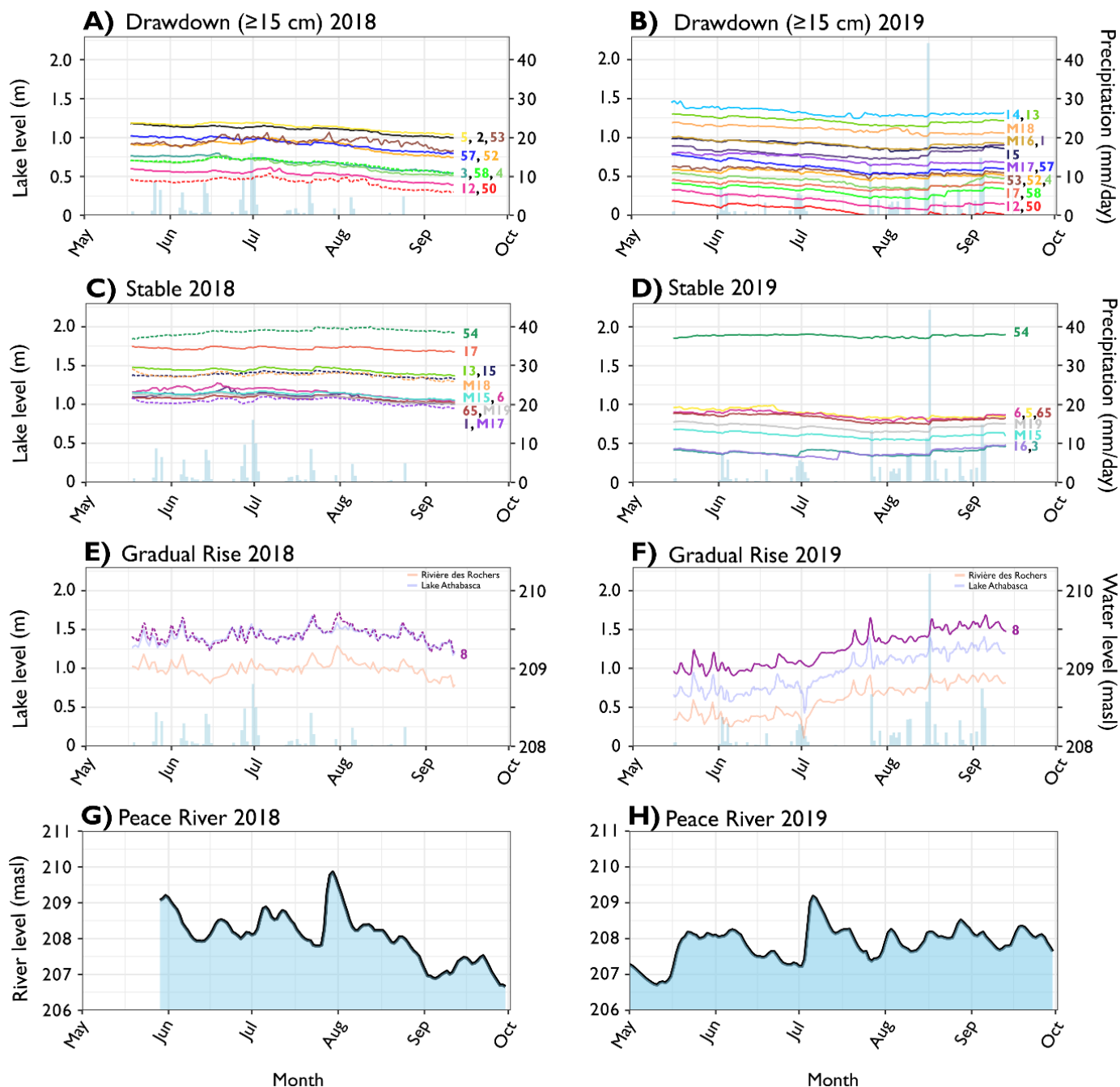


Figure 2.4. Temporal patterns of lake-level variation in the Peace sector (Panels A – F) from May to October during the open-water seasons of 2018 (left column) and 2019 (right column): Drawdown (A, B), Stable (C, D) and Gradual Rise (E, F). Hydrometric data showing the water-level variation of Lake Athabasca (ID: 07MD001) and the Rivière des Rocher (ID: 07NA007) are included in panels E and F. Also shown in A – F (as vertical light-blue bars) is daily precipitation (mm) measured at Fort Chipewyan Airport (ID: 3072659). Panels G and H show the daily water level of the Peace River (masl) in 2018 and 2019, measured below the Chenal des Quatre Fourches (ID: 07KC005). Dotted lines in panels A, C and E are lakes that received floodwater during the spring 2018 ice-jam flood prior to logger installment (Remmer et al., 2020b).

None of the lakes in the Peace sector displayed a sharp rise in lake level in 2018 and 2019, and temporal patterns of lake-level variation in the three other categories show little to no association with variation of Peace River levels during the open-water season (Figure 2.4G, H).

2.4.2 Athabasca sector

Similar to the Peace sector, drawdown was the most common lake-level pattern in the Athabasca sector during the two study years (Figure 2.5A, B). Eleven lakes fell in this category in 2018 (42.3%; PAD 19, 20, 23, 24, 32, 36, 39, M1, M3, M4, M6) and 10 lakes in 2019 (34.5%; PAD 19, 20, 21, 23, 32, M4, M6, M8, M9). Six lakes fell into this category in both study years. The timing and duration of lake-level decline was comparable to that observed in the Peace sector in both study years. Of the 11 drawdown lakes in 2018, 8 received floodwaters during the spring prior to logger deployment. Lakes in the drawdown category in 2018 and 2019 are situated across a wide range of elevations (205- 225 masl; Figure 2.6). Three of the drawdown lakes in 2018 displayed a different lake-level pattern in 2019; one was classified as stable and two as sharp rise.

In contrast to the Peace sector, relatively few lakes fell into the stable category in the Athabasca sector in 2018 and 2019 (Figure 2.5C, D). This includes 3 lakes in 2018 (11.5%; PAD 22, M5, M11) and 7 lakes in 2019 (24.1%; PAD 22, 27, 36, 39, M5, M6, M11). All but one of these lakes are situated above 215 masl (Figure 2.6). The exception, M11, is located southeast of Mamawi Lake at 205-210 masl. All three of the lakes in 2018 remained in the stable category in 2019.

Five lakes (19.2%; M8, M9, M10, M12, M14) displayed gradual rise of lake level in 2018 and one lake (3.4%; M10) fell in this category in 2019 (Figure 2.5E, F). Lake-level variations in these lakes coincided with those of the open-drainage network (lakes Claire, Mamawi and Athabasca; Figure 2.5E, F). Fewer lakes with gradual rise in 2019, compared to 2018, corresponds with 0.5 m lower peak lake levels in the open-drainage network (Figure 2.5E, F). In 2019, lake level at M10 declined gradually between mid-May and early August, similar to lakes in the drawdown category, and then rose gradually after mid-August when adjacent

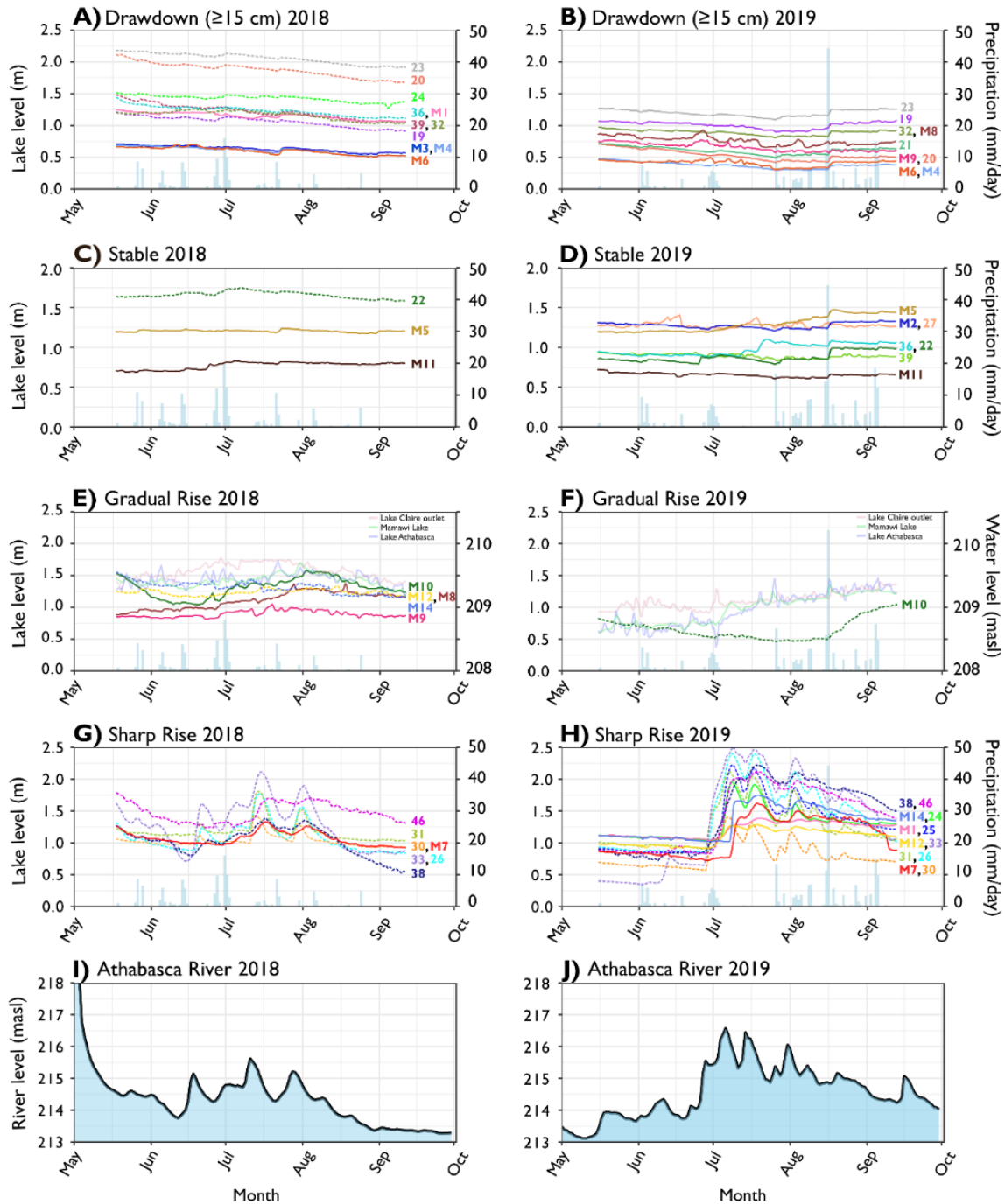


Figure 2.5. Temporal patterns of lake-level variation in the Athabasca sector (Panels A–H) May to October during the open-water seasons of 2018 (left column) and 2019 (right column): Drawdown (A, B), Stable (C, D), Gradual Rise (E, F) and Sharp Rise (G, H). Hydrometric data showing the water-level variation of the three largest lakes that comprise the open-water network including Lake Claire, Mamawi Lake and Lake Athabasca (ID's: 07KF002, 07KF003, 07MD001, respectively), are included in panels E and F. Also shown in A-H (as vertical light-blue bars) is daily precipitation (mm) measured at Fort Chipewyan Airport (ID: 3072659). Panels I and J show the daily water level of the Athabasca River (masl) in 2018 and 2019, measured at Embarras Airport (ID: 07DD001). Dotted lines in panels A, C, E, F, G, and H are lakes that received floodwater during the spring (prior to logger installment; Remmer et al., 2020a,b).

Lake Claire reached maximum lake level. Lakes in the gradual rise category are situated at elevations between 205-210 masl and occur south-east of Lake Claire (M8, M9, M10) and along the southern edge of Mamawi Lake (M12, M14; Figure 2.6). In 2019, M12 and M14 shifted to the sharp rise category, and M8 and M9 exhibited the drawdown pattern.

Several lakes in the Athabasca sector displayed sharp rise(s) of lake level, which was not observed in lakes of the Peace sector (Figure 2.5G, H). In 2018, 7 lakes (26.9%; PAD 26, 30, 31, 33, 38, 46, M7) fell into this category, with two to four sharp rises evident in late June, early July, mid-July and early August. These same 7 lakes, plus 5 others (41.4%; PAD 24, 25, M1, M12, M14), displayed sharp rises in early July, mid-July, late July and early August in 2019. In most instances, lake levels declined markedly after each rise and coincided with sharp rise and fall in Athabasca River levels during both years, suggesting open-water river flooding is the cause of these lake-level variations (Figure 2.5I, J). Exceptions to this include three lakes where lake level did not decline markedly after a sharp rise (PAD 46 in 2018, M1 and M12 in 2019). Prolonged high lake levels in PAD 46 and M12 may reflect inundation by rising levels of adjacent Mamawi Lake after input of river floodwaters in July. M1 lacks an adjacent open-drainage lake, which suggests the peak river levels in early July 2019 only briefly exceeded the sill elevation for this lake on one occasion, and the lake remained bank-full thereafter. Maximum rise of lake level ranged from 31-170 cm, with a mean of 73 cm in 2018 and 104 cm in 2019 (Table 2.1). River floodwaters reached more lakes in 2019 when the Athabasca River (at Embarras Airport) reached a maximum level of ~216.5 masl, 1.1 m higher than the maximum level in 2018 (~215.4 masl) (Figure 2.5). In both years, open-water flooding occurred south of the Athabasca River, and along downstream portions of Cree/Mamawi Creek and adjacent to Mamawi Lake (Figure 2.6).

2.5 Discussion

Continuous hourly lake-level measurements at ~50 lakes across the Peace-Athabasca Delta during two open-water seasons (2018, 2019) revealed four distinctive patterns of lake-level variation: drawdown (≥ 15 cm decline), stable (< 15 cm decline), gradual rise and sharp rise. The number of lakes that displayed each lake-level pattern differed between years and sectors of the PAD. In the Peace sector, drawdown and stable patterns were equally

prevalent in 2018 ($n = 10$ and 11 , respectively), but drawdown was more common than stable in 2019 (15 and 8 , respectively). In the Athabasca sector, drawdown was the most prevalent pattern in 2018 (10), followed by sharp rise (7) and gradual rise (5). In 2019, sharp rise (12) was most common, followed by drawdown and stable (9 and 7 , respectively). Lakes in the stable and drawdown categories consistently occur at elevations greater than 215 masl in the Peace sector, in contrast to the Athabasca sector where lakes with these lake-level patterns were located across a range of elevations (205 - 220 masl). Gradual rise lakes consistently occur at low elevations (205 - 210 masl) and are situated adjacent to Lake Claire and Mamawi Lake in the Athabasca sector, whereas the sole gradual rise lake in the Peace sector possesses a connection to the Rivière des Rochers via Chilloney's Creek. Lakes that had a sharp rise of lake level were only found within the Athabasca sector at low elevations (205 - 215 masl) and adjacent to active river channels (Athabasca and Embarras rivers, Cree/Mamawi Creek).

Lakes in the PAD have long been recognized to span a broad hydrological spectrum and be potentially influenced by several processes including snowmelt, ice-jam flooding, rainfall, open-water flooding, water-level fluctuations in the open-drainage network and evaporation (PADPG, 1973; Peters et al., 2006; Pietroniro et al., 1999; Prowse and Demuth, 1996; Remmer et al., 2018, 2020a,b; Wolfe et al., 2008). Lake-level variations are a sensitive measure and a direct consequence of the relative influence of many of these hydrological processes. Based on systematic lake-level patterns (Figures 2.4, 2.5) and their spatial distribution (Figure 2.6), we propose a new hydrological classification system for lakes in the PAD, defined by key hydrological processes (Table 2.2).

2.5.1 Hydrological processes that define a new lake categorization system for the PAD

During the open-water season, we identify a singular dominant hydrological process, among the several processes operating across the PAD, that defines each of the four new lake hydrological categories (Table 2.2). 1) Drawdown lakes are strongly influenced by evaporation, as reflected by their consistent, gradual lake-level declines throughout the summer that exceed 15 cm. However, because ice-jam floodwaters in 2018 raised lake levels of several drawdown lakes in the Athabasca sector, surface outflow also may have

contributed to their lake-level decline early during the open-water season. 2) Stable lakes did not experience substantial drawdown (< 15 cm) because of summer rainfall that offset evaporation. Lake morphology and catchment characteristics, including watershed-to-lake area and surface area-to-volume ratios, and localized rainfall events, may explain the differing relative influence of rainfall, its associated runoff and evaporation on drawdown and stable lakes. These factors are difficult to quantify because of the low relief terrain and there is only one meteorological station (Fort Chipewyan airport) in this remote region. 3) Gradual rise lakes are characterized by inflow due to hydrological connectivity with adjacent large open-drainage lakes (principally lakes Claire and Mamawi) and distributary channels of the open-drainage network. 4) Sharp rise lakes correspond with rise of the Athabasca River beyond its flood stage, identifying open-water season flooding as the main hydrological process that characterizes these lakes. Ice-jam flooding and snowmelt can influence all lakes across the delta during the early open-water season (PADPG, 1973; Prowse and Conly, 1998; Timoney, 2013). However, these hydrological processes are not readily identified in the lake-level records because loggers cannot be installed until after ice-off and thus after snowmelt and ice-jam flooding have occurred.

The new lake hydrological classification system offers distinct advantages over the conventional closed-, restricted- and open-drainage classification scheme. Table 2.2 lists the number of lakes in our study set that fall into the former drainage categories based on water isotope and water chemistry data from 2001 (Wolfe et al., 2007). Closed-drainage lakes (and their subset of rainfall-influenced lakes) have now been partitioned into drawdown and stable lakes based on the prevailing influence of evaporation and rainfall, respectively. The broad spectrum of restricted-drainage lakes is reflected by distribution among all four of the new categories, further demonstrating that restricted-drainage lakes are influenced by a wide range of hydrological processes. A distinct advantage is that the new categories distinguish formally restricted-drainage lakes that receive open-water flooding via direct inflow of Athabasca River water (i.e., sharp rise lakes) versus inundation or connection with the open-drainage network (i.e., gradual rise lakes). Conventionally, three lakes have been identified as open-drainage (Claire, Mamawi, Richardson; Timoney, 2013),

but they span two of the four new categories proposed here. Lakes Claire and Mamawi displayed gradual rise of lake levels, as would be expected of large, through-flow lakes in flat terrain that receive inflow from rivers with sharply rising lake levels. However, lake levels rose sharply at Richardson Lake (PAD 38) during open-water seasons of 2018 and 2019. Lake levels at PAD 25, 26 and 46, identified as open-drainage by Wolfe et al. (2007), also rose sharply. Thus, the new lake classification system identifies the non-uniform influence of hydrological processes in this previously defined category of open-drainage lakes. It also reveals that a broad suite of hydrological processes influence lake levels in the delta and shifts in several of these processes can lead to lake-level drawdown. These include climate-induced increase in duration of the open-water season and increased evaporation, reduced open-water flooding and decline of precipitation.

Table 2.2. Lake level functions for the four lake-level categories. Bolded terms identify the dominant hydrological process for each category. Former drainage categories for lake sites that overlap with Wolfe et al. (2007) are included for comparative purposes. Lake sites that do not overlap are designated as “uncategorized”.

Category	Lake Level Functions	Former Drainage Categories (#)
Drawdown	$LL(f): [IJF + S_m + R_r - O - E]$	-closed (11) -restricted (11) -rainfall-influenced (1) -uncategorized (9)
Stable	$LL(f): [IJF + S_m + \mathbf{R_r} - O - E]$	-restricted (8) -closed (5) -rainfall-influenced (1) -uncategorized (7)
Gradual Rise	$LL(f): [IJF + S_m + R_r + \mathbf{ODN} - O - E]$	-restricted (1) -uncategorized (5)
Sharp Rise	$LL(f): [IJF + S_m + R_r + \mathbf{OWF} - O - E]$	-open (4) -restricted (4) -uncategorized (4)

LL = lake level, IJF = ice-jam floodwaters, S_m = snowmelt, R_r = rainfall, ODN = open-drainage network, OWF = open-water floodwaters, O = surface outflow, E = evaporation.

2.5.2 Characterization of open-water flooding

Recently, substantial open-water flooding was detected in the Athabasca sector using seasonal measurement of water isotope compositions (Remmer et al., 2020a) and remote sensing (Peters et al., 2021). Hourly lake-level data improves our understanding of the influence of this hydrological process by capturing the precise timing that floodwaters enter a lake, the magnitude of lake-level rise and the spatial extent of flooding. In 2018, open-water flooding of sharp rise lakes coincided with increases in the water level of the Athabasca River during 2-3 intervals in mid-June, mid-July and late July, which raised lake levels by 0.33-1.41

meters (Figure 2.4; Table 2.1). In 2019, open-water flooding occurred during 3-5 intervals, including three in July and two in early August, which raised lake levels of sharp rise lakes by 0.31-1.79 meters (Figure 2.5; Table 2.1). Each one of these flood events occurred during an interval of one to two weeks. Continuous lake-level measurements demonstrate that multiple pulses of river floodwater may enter lakes in the Athabasca sector during the open-water season potentially resulting in lake-level increases exceeding 1.5 m. Given that most lakes are <1 m deep, and some are less than 0.5 m deep, open-water flooding is clearly an important recharge mechanism for lakes in the Athabasca sector. High levels of the Athabasca River in 2018 and 2019 also promoted gradual rise of water levels at Mamawi Lake and Lake Claire, which inundated nearby basins (gradual rise lakes; Figure 2.4, 2.5). Although Athabasca River peaked at ~216.5 masl in 2019, more than 1 m higher than in 2018 (~215.4 masl), spatial extent of floodwaters was similar (Figure 2.6). Flooded regions captured by sharp rise lakes include along the southern margin of the delta and the Cree/Mamawi Creek corridor in the Athabasca sector, and likely define the areas most susceptible to the direct effects of open-water flooding. Our interpretation of the spatial extent of open-water flooding includes gradual rise lakes (Figure 2.6), which were indirectly influenced by the high discharge on the Athabasca River via the open-drainage network. Notably, open-water flooding did not extend to the terminus region of the Athabasca River, which includes the traditional territory of the Athabasca Chipewyan First Nation located adjacent to the WBNP boundary at the southeastern edge of the PAD, likely due to diversion of flow through Cree/Mamawi Creek caused by the Embarras Breakthrough in 1982 (Kay et al., 2019; Wolfe et al., 2008). Indeed, the lake-depth data reveal that drawdown is common among lakes at the Athabasca River terminus in the absence of open-water flooding (e.g., M1, M3, M4, M6; Figure 2.5, 2.6). This is an important finding because it identifies systematic lake-level drawdown which has negative consequences for land users and access to traditional lands. We consider the spatial extent of open-water flooding depicted in Figure 2.6 to be more accurate than that approximated in Figure 4 of Remmer et al. (2020a), in part because the latter is influenced by the timing of water sample collection for isotope analysis.

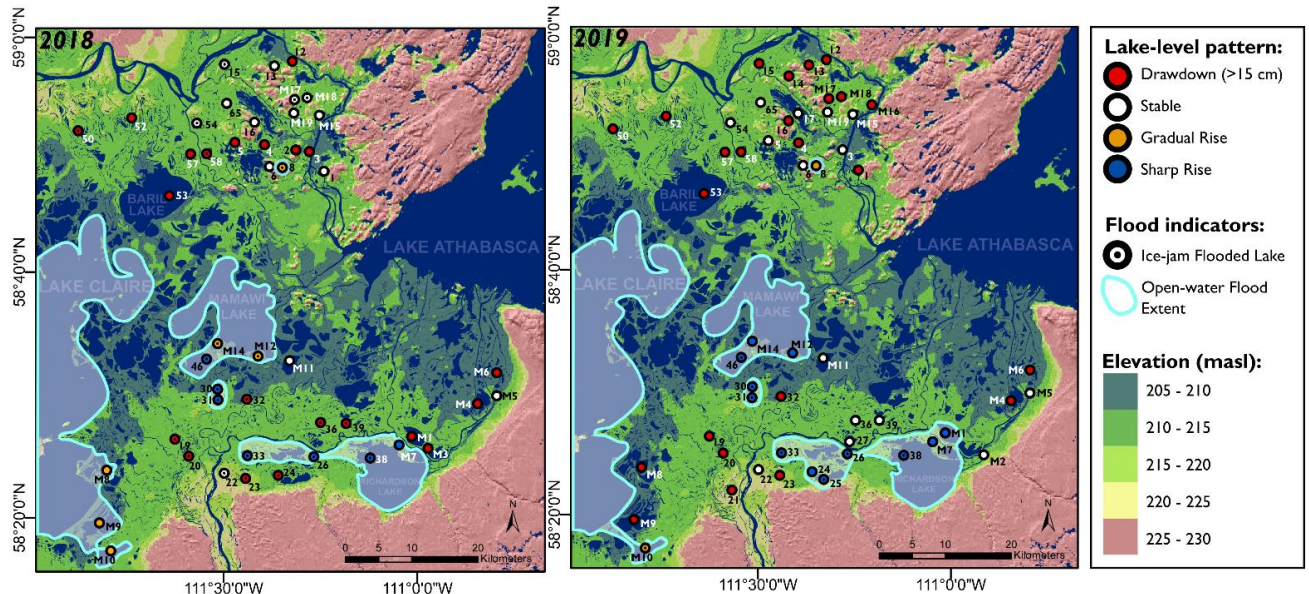


Figure 2.6. Maps showing location of lake-level patterns (left: 2018; right 2019) and estimated open-water flood extent incorporating lakes in both gradual rise and sharp rise categories, superimposed on land surface elevation of the Peace-Athabasca Delta, Alberta. Figure was created using ArcGIS Desktop version 10.7.1 and assembled by Laura Neary using the following data sources (shapefiles): Government of Canada (water bodies: <https://open.canada.ca/data/en/dataset/448ec403-6635-456b-8ced-d3ac24143add>; geospatial CDEM: <https://maps.canada.ca/czs/index-en.html>).

Periodic ice-jam flooding in spring is well known as an important source of replenishment for lakes in the PAD (Prowse and Conly, 2000; Prowse and Lalonde, 1996; Timoney et al., 1997) and causes of reduced frequency of this hydrological process have been subject to much debate given concerns of lake-level drawdown during recent decades (e.g., Beltaos, 2018; Wolfe et al., 2020). Less attention has been placed on the frequency of open-water flood events probably because, unlike ice-jam floods, they are difficult to predict or observe as they typically generate smaller, more localized flooding and can occur at any time during the open-water season. We examined the Athabasca River hydrometric record at the Old Fort gauging station (ID: 07DD011; Figure 2.1) to assess the frequency of potential open-water flood events equivalent to or greater than 2018 and 2019 since the Embarras Breakthrough in 1982 (Table 2.3; Figure 2.7). We utilized data recorded at Old Fort hydrometric station because the station at Embarras Airport only reports water levels since 2014. At Old Fort station, peak water level was ~214.5 masl during 2018 and ~215.4 masl during 2019. The water-level record of the Athabasca River at this station shows that the 2018 threshold has been exceeded during 16 years since 1982 (42% of years), demonstrating

that open-water flooding of similar or greater magnitude as 2018 has been a common occurrence during the past 38 years. Given that peak water levels of the Athabasca River have exceeded the 2019 threshold during only 8 of those years (21%), it is likely that open-water flooding has infrequently extended to the Athabasca River terminus during this time. This interpretation is consistent with paleolimnological records that depict reduced influence of river floodwaters in this region of the Athabasca sector since 1982 (Kay et al., 2019). The record at Old Fort station also shows an extended interval of low peak summer river levels during 1998-2010. Since then, peak river levels have exceeded the 2018 threshold in at least 6 years since 2011 (note missing data for 2014 and 2015). We also note that the timing of peak water level at Old Fort station in any given year may span a ~6-week interval. Such marked seasonal and annual temporal variation in peak river levels of the Athabasca River and associated open-water flooding highlights the importance of monitoring the frequency of these important recharge events using continuous lake-level recorders.

Table 2.3. Annual maximum open-water season level (m) of the Athabasca River between May 30 and August 17 measured at the Old Fort gauging station (ID: 07DD011) from 1982-2019. Note that data are not available (NA) for 2014 and 2015.

Year	Maximum level (m)	Day of the Year	Julian Day Number (JDN)
1982	215.22	12-Jul	193
1983	214.17	29-Jul	210
1984	214.06	14-Jun	166
1985	214.74	05-Jul	186
1986	216.04	25-Jul	206
1987	213.85	09-Aug	221
1988	215.07	13-Jul	195
1989	215.09	10-Aug	222
1990	215.81	20-Jun	171
1991	214.36	17-Jun	168
1992	212.73	22-Jun	173
1993	212.86	31-Jul	212
1994	214.35	12-Jul	193
1995	214.94	17-Aug	229
1996	215.83	07-Jun	159
1997	217.39	01-Jul	182
1998	213.62	07-Jul	188
1999	214.37	15-Jul	196
2000	214.22	30-Jul	212
2001	214.82	01-Aug	213
2002	212.75	06-Jul	187
2003	214.45	29-Jun	180
2004	213.99	18-Jul	200
2005	213.37	01-Jun	152
2006	213.52	01-Jun	152
2007	213.93	13-Jun	164
2008	213.96	19-Jun	171
2009	214.09	16-Jul	197
2010	212.53	22-Jun	173
2011	215.64	16-Jul	197
2012	215.54	01-Aug	214
2013	215.84	17-Jun	168

2014	NA	NA	NA
2015	NA	NA	NA
2016	214.06	16-Jun	168
2017	214.97	17-Jul	168
2018	214.55	11-Jul	192
2019	215.45	05-Jul	186

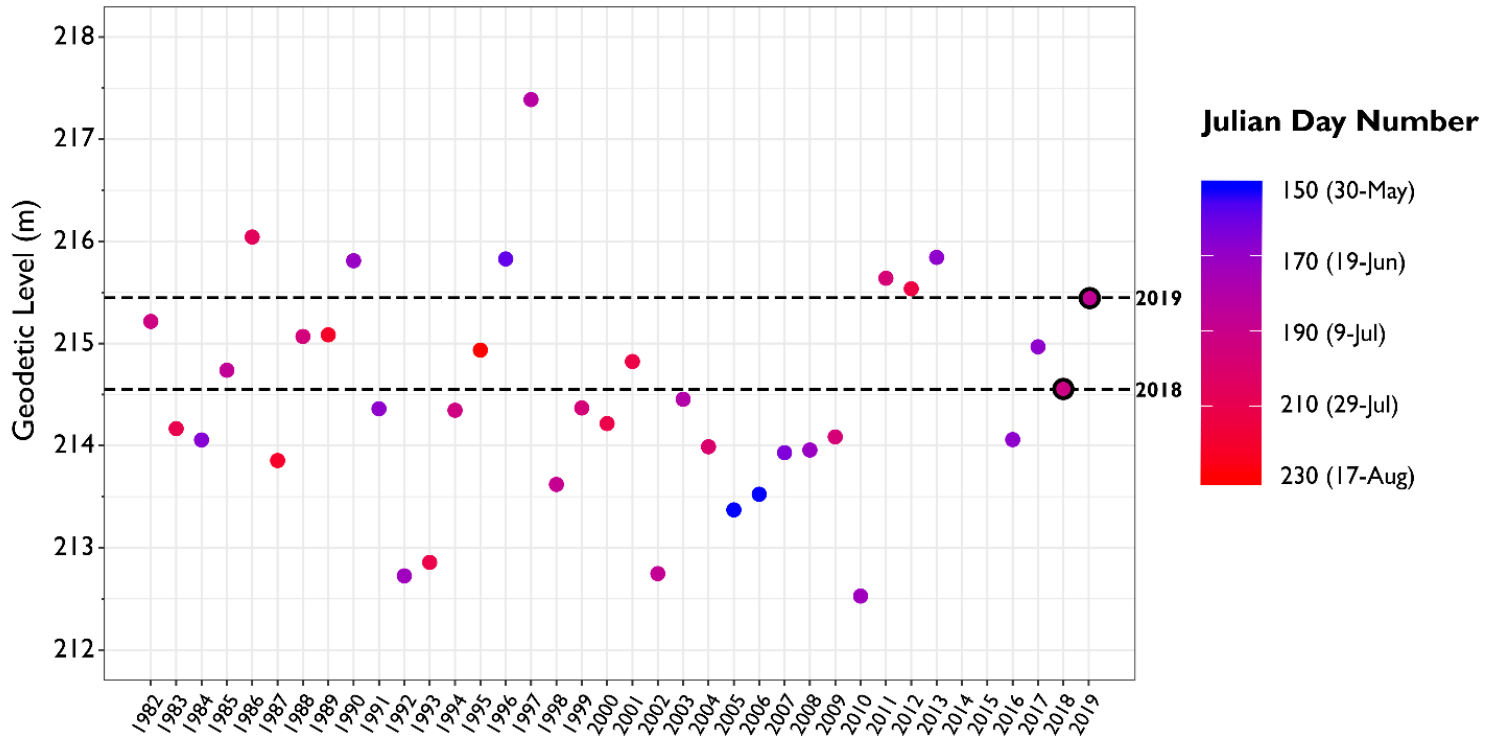


Figure 2.7. Maximum water-level (masl) of the Athabasca River recorded between May 30 and August 17 at the Old Fort gauging station (ID: 07DD011) from 1982 to 2019. The colour of the points indicates the date (Julian Day Number) when each of the annual open-water season peaks occurred. Horizontal dashed lines and black-outlined coloured points indicate the peak water level recorded in the summer of 2018 (214.5 m) and 2019 (215.4 m). Note that data are missing for 2014 and 2015.

2.5.3 Implications for monitoring

Given projected declines in summer discharge of east-flowing rivers fed by high-elevation glaciers and snowpack in the Rocky Mountains (Chernos et al., 2020; Debeer and Sharp, 2007; Hugonnet et al. 2021; Tennant and Menounos, 2013), peak summer levels in the Athabasca River and associated open-water flooding at the PAD are likely to decline in coming decades. If so, frequency of lakes in the ‘sharp rise’ and ‘gradual rise’ categories will decrease and shift to the ‘stable’ and/or ‘drawdown’ categories depending on amount of rainfall runoff and evaporative water loss. Combined with predicted declines in ice-jam flood frequency (Beltaos and Bonsal, 2021; Lamontagne et al., 2021) and longer duration of the

open-water season promoting greater evaporative water loss (Barnett et al., 2005; Schindler and Donahue, 2006), lake levels are likely to decline across broad areas of the PAD.

Ongoing monitoring of lake levels can inform stewardship decisions for Wood Buffalo National Park, a priority of the federal Action Plan (WBNP, 2019), and ensure long-term sustainability of the delta's ecosystems during the ongoing climate crisis. For example, lake-level monitoring has become an effective management tool for the North Saskatchewan Watershed Alliance to track long-term water-level trends at 33 lakes across the province of Alberta (Islam and Seneka, 2015; NSWA, 2017). Analogous to Figure 4.1a in NSWA (2017), we summarize the number of lakes in each hydrological category using pie charts to demonstrate how these data can be used to track the influence of shifting hydrological conditions across sectors and years (Figure 2.8). This end-product highlights the importance of open-water flooding (sharp rise lakes) in the Athabasca sector and evaporation (drawdown lakes) in the Peace sector. The approach captured marked increase of drawdown lakes in the Peace sector and sharp rise lakes in the Athabasca sector between 2018 and 2019, which suggests strong potential for ongoing lake-level monitoring. We recommend lake-level monitoring be integrated with other hydrological monitoring tools, such as water isotope tracers and remote sensing imagery, to quantify relative importance of ice-jam flooding, snowmelt and evaporation on lake water balance (Remmer et al., 2020a,b) and to corroborate the spatial extent of river flooding and lake drying (Pavelsky and Smith, 2008; Peters et al., 2021; Töyrä and Pietroniro, 2005). Combined use of these approaches can strengthen monitoring of dynamic lake-rich landscapes, such as the PAD, and better characterize the varying influence of hydrological processes and the timescales at which they operate. Given sufficient financial resources, transferability of this approach to similar remote freshwater landscapes is possible, such as the nearby Slave River Delta (Brock et al., 2007) and Saskatchewan River Delta (MacKinnon et al., 2015), although there may be site-specific hydrological processes to consider such as the influence of seiche events on Great Slave Lake on the former.

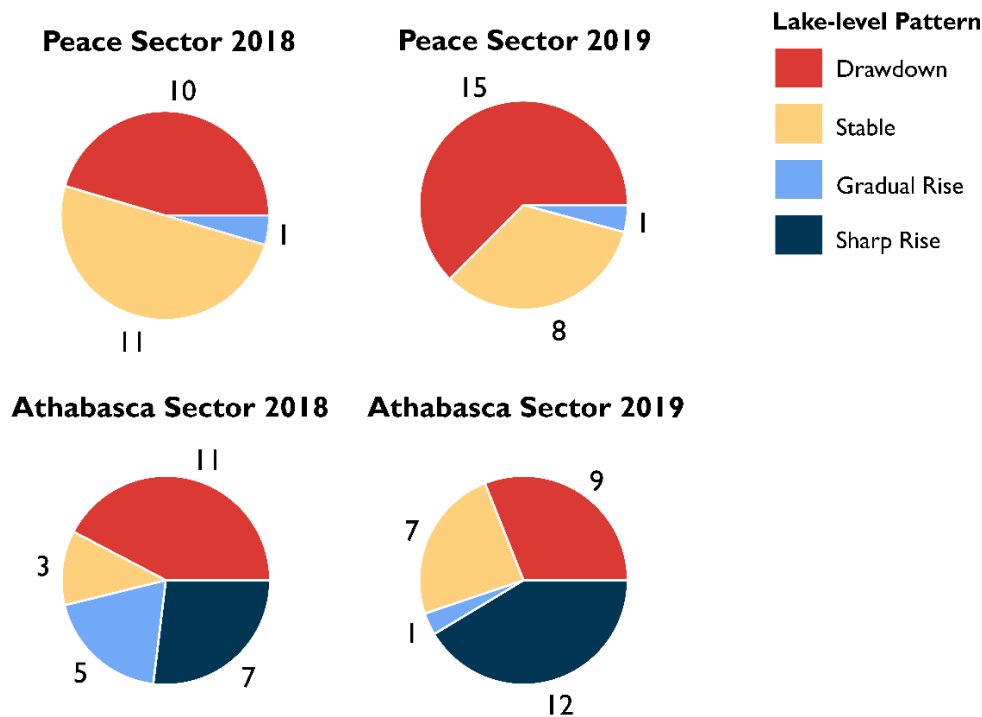


Figure 2.8. Pie charts summarizing the distribution of the four lake-level patterns by sector (Peace vs Athabasca) and year (2018, 2019). Values identify the number of lakes represented by each pie slice.

At the PAD, wildlife abundance, aquatic habitat, and limnological conditions are strongly regulated by hydrological processes that cause changes in lake levels. Muskrat, for example, is a culturally important and ‘keystone’ species used to track aquatic habitat availability and ecological integrity of shallow lakes in the PAD (Straka et al., 2018; Ward et al., 2018, 2020). Muskrat abundance has been shown to respond rapidly to changes in freeze-up water levels as they rely on vegetation under ice to survive throughout winter (Virgl and Messier, 1997). Density of muskrat houses increased by two orders of magnitude after spring flooding in 2014 and slowly declined during subsequent drier years as water depth declined (Straka et al., 2018). Lake level rise caused by input of turbid floodwater dramatically reduces water clarity, concentrations of dissolved nutrients, organic carbon and ions, and biomass of phytoplankton and macrophytes at seasonal timescales or longer (Wiklund et al., 2012). Flooding that causes rapid rise of lake levels is likely to result in greater reductions in these variables than flooding that causes gradual rise. Absence of recent flooding, on the other hand, leads to higher concentration of nutrients and ions, higher pH and greater penetration of solar radiation to the bottom of shallow lakes, leading to higher production of

macrophytes, phytoplankton and their consumers. Such conditions characterize lakes in the stable and drawdown categories (Wiklund et al., 2012). Persistent drawdown of water levels eventually results in loss of littoral habitat and encroachment of terrestrial vegetation such as willow which impedes mobility of large wildlife (e.g., bison) and land users (Timoney, 2013). Spatio-temporal variation in lake types across the PAD supports an abundance and diversity of aquatic and semi-aquatic biota, yet climate change is predicted to lead to increased proportions of lakes in the drawdown and stable categories at the expense of lakes in the gradual rise and rapid rise categories. Clearly, continued monitoring of lake-level variation, coupled with research on links with the ecological structure and function of the lakes, is important to inform about ongoing hydroecological changes in response to climate trends and other stressors.

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Chapter 3 | Synthesis of a hydrological, water chemistry and contaminants research program in the Peace-Athabasca Delta (Canada) to inform long-term monitoring of shallow lakes

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3.1 Introduction

Effective freshwater ecosystem monitoring programs define status, detect trends, and generate data to understand the timing, magnitude of environmental change and the potential causes – information that is imperative for evidence-based decision-making, climate-change adaptation and mitigation planning, and development and assessment of policy and management interventions (Ryder et al., 2010; Lindenmayer et al., 2012; Pearce-Higgins et al., 2022). Key principles include 1) scientifically defined objectives that are aligned with priorities of stakeholders, 2) sound study design, and 3) measurement of responsive, informative variables at appropriate spatial and temporal scales (Field et al., 2007; Lovett et al., 2007; Lindenmayer and Likens, 2010; Lindenmayer et al., 2022). Adherence to these principles requires advanced understanding of the processes influencing ecosystem structure and function (Lindenmayer et al., 2008). Knowledge generated from substantive prior research and long-term primary data acquisition, thus, constitutes an important foundation in support of implementation of successful long-term freshwater ecosystem monitoring programs (Likens and Lambert, 1998; Lindenmayer and Likens, 2010, 2018; Sparrow et al., 2020).

At the Peace-Athabasca Delta (PAD; northeastern Alberta, Canada; Figure 3.1), a Ramsar Wetland of International Importance and traditional territory of three Indigenous nations (Athabasca Chipewyan First Nation (ACFN), Mikisew Cree First Nation (MCFN), and Fort Chipewyan Métis Nation (FCMN)), the need for effective aquatic ecosystem monitoring has long been recognized (PADPG, 1973; PADIC, 1987; PADTS, 1996). More than 50 years ago,

the PAD Project Group (PADPG), a multi-jurisdictional environmental impact assessment of Peace River flow regulation by the W.A.C. Bennett Dam, recommended initiation of a long-term monitoring program to obtain “... understanding of the complex interrelationships of the hydrologic and biological processes and to provide some predictive capability for planning and decision making” (PADPG, 1972; page 142). During the following two decades, periodic environmental reassessments reiterated this need (PADIC, 1987; PADTS, 1996), but a long-term delta-wide lake monitoring program has remained elusive even as concern has grown over aquatic ecosystem degradation (e.g., Prowse and Conly, 1998; Schindler and Donahue, 2006; Schindler and Smol, 2006; Schindler, 2010). Primary concerns include observed and recorded drawdown of lake and river water levels, variably attributed to hydroelectric regulation of Peace River flow and climate (e.g., PADPG, 1973; Prowse and Conly, 1998; Peters et al., 2006b; Wolfe et al., 2012, 2020; Beltaos, 2014, 2018, 2019; Hall et al., 2019; Timoney et al., 2019; Lamontagne et al., 2021), and the potential for pollution of aquatic ecosystems by contaminant emissions from upstream oil sands development (e.g., Timoney and Lee, 2009; Schindler, 2010; Hebert et al., 2013, 2019; Evans et al., 2016). In 2014, MCFN formally petitioned that UNESCO inscribe Wood Buffalo National Park (WBNP) on the List of ‘World Heritage In Danger’, in part because industrial activities pose serious threats to ‘Outstanding Universal Values’ of the PAD’s aquatic ecosystems (note that ~80% of the PAD lies within WBNP), and “provincial and federal monitoring of the effects of upstream activities and climate change on WBNP and the PAD has been wholly inadequate” (MCFN, 2014; page 34). The petition instigated investigations by the World Heritage Convention (WHC) of UNESCO, International Union for Conservation of Nature (IUCN) and Government of Canada (WHC/IUCN, 2017; IEC, 2018; full list of documents at <https://whc.unesco.org/en/list/256/documents/>) which informed priorities for a federal WBNP Action Plan aimed at improving environmental stewardship and increasing inclusion of Indigenous rightsholders (WBNP, 2019). One of the priorities is to implement an “integrated PAD Research and Monitoring program ... to detect cumulative effects on the PAD and generate information that informs land-use management and regulatory decision

making” (WBNP, 2019; page 57), reiterating the unfulfilled priority first identified by the PADPG in the early 1970s.

Locations of monitoring sites

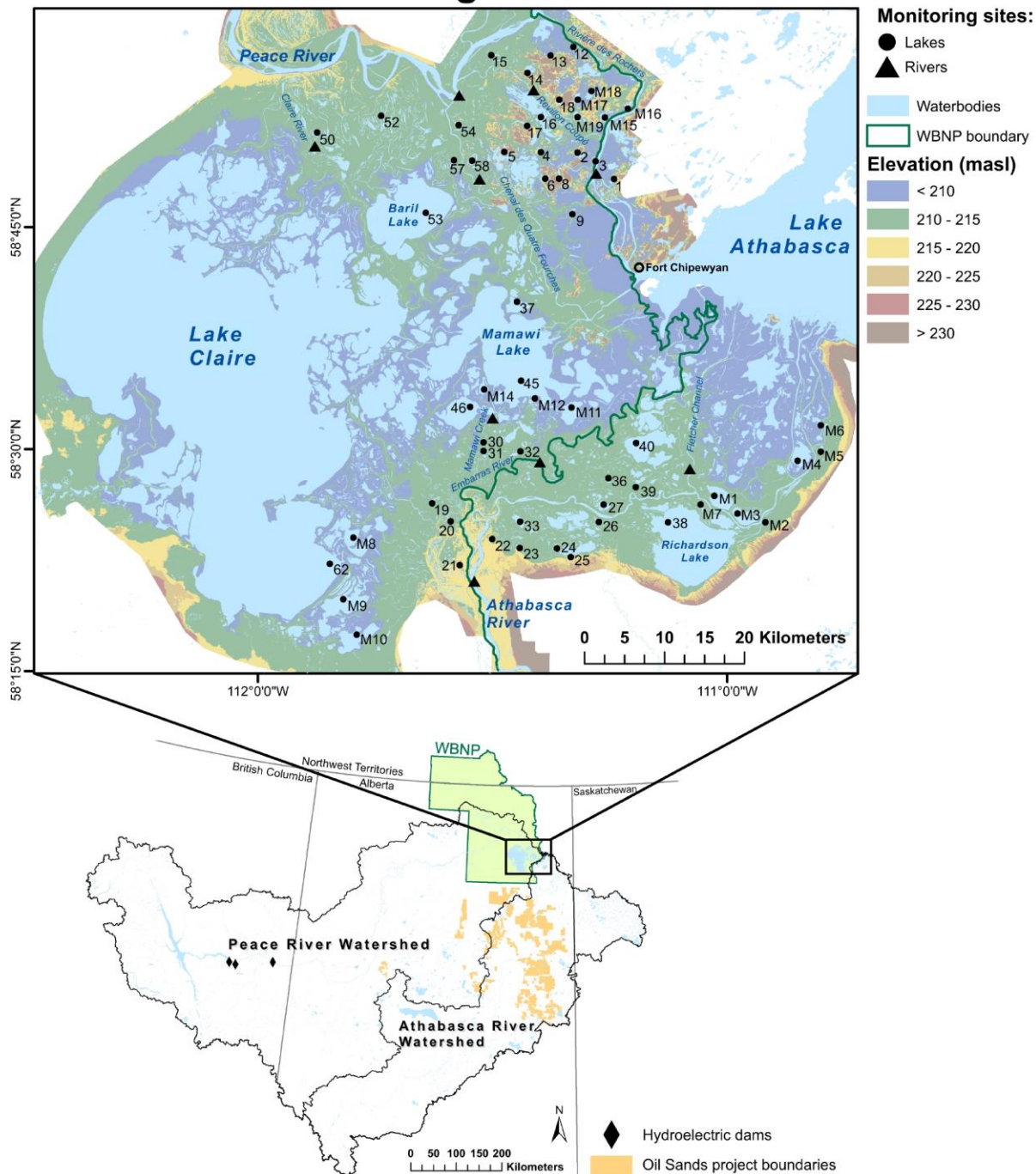


Figure 3.1. Lower map shows locations of the Peace-Athabasca Delta within the Peace River and Athabasca River watersheds and Wood Buffalo National Park (WBNP), the upstream hydroelectric dams on the Peace River (solid black diamonds) and oil sands mining project boundaries (1985-2015) in the

Lower Athabasca River watershed (orange; <http://osip.alberta.ca/library/Dataset/Details/671>). Upper map shows locations of the 60 lakes (solid black circles) and 9 river (solid black triangles) sites sampled during 2015-2021 and the delta's elevation. Map sources include: <https://catalogue.ec.gc.ca/geonetwork/srv/api/records/ca0f663f-acfc-4915-9210-3729a86b2e94> for a high-resolution digital elevation model (2022); <https://natural-resources.canada.ca/science-and-data/science-and-research/earth-sciences/geography/topographic-information/geobase-surface-water-program-geeau/watershed-boundaries/20973> for watershed boundaries; <https://open.canada.ca/data/en/dataset/9e1507cd-f25c-4c64-995b-6563bf9d65bd> for the WBNP boundary (dark green), DMTI Spatial Inc. (2023) for surface water bodies (light blue).

The large size (~6,000 km²), hydrological complexity, and abundance of difficult-to-access lakes (most are inaccessible by boat or road) have presented formidable impediments to establishing a sustained monitoring program that encompasses the broad hydroecological gradients of the PAD (Figure 3.1). The southern Athabasca sector is a low-relief active delta at the Athabasca River's terminus, whereas the northern Peace sector is a relict delta where bedrock inliers provide greater topographic relief (Timoney, 2013). Water flows northward through the delta, from the Athabasca River and its distributaries (including Embarras River, Mamawi Creek, Fletcher Channel) to Mamawi Lake and Lake Athabasca before joining the Peace River via the Chenal des Quatre Fourches, Rivière des Rochers and Revillon Coupé. The Peace River flows eastward along the delta's northern margin and exerts hydrological influence mainly by regulating the rate of outflow (Prowse and Conly, 2000; Timoney, 2013). Direct Peace River inflow occurs episodically, mainly during short-lived ice-jam events when water levels elevate sufficiently to reverse flow direction and inundate broad areas of the delta (Prowse and Conly, 1998, 2000; Timoney, 2013). Both ice-jam and open-water flooding occur in the Athabasca sector (Neary et al., 2021 [**Chapter 2**]). Shallow lakes (typically 0.5 to 3 m deep) are abundant across both sectors and span a broad gradient of hydrological connectivity to the rivers from classically termed open-drainage to restricted- and closed-drainage (PADPG, 1973; Pietroniro et al., 1999), although a new lake classification attributable to influential hydrological processes has been developed using lake-level time series (Neary et al., 2021 [**Chapter 2**]). Perched lakes (i.e., restricted- and closed-drainage lakes; Pietroniro et al., 1999) with weaker connectivity to the river network are most vulnerable to evaporative drawdown and atmospheric deposition of contaminants, whereas open- and restricted-drainage lakes are most vulnerable to fluvial supply of contaminants (Peters et al., 2006a; Kay et al., 2020; Remmer et al., 2020a; Neary et al., 2021 [**Chapter 2**]).

As described by Maclean et al. (2021), Indigenous-led community-based monitoring (CBM) programs have operated in the PAD since 2008 (MCFN) and 2010 (ACFN) at sampling locations selected by Elders and land-users to assess for environmental changes that impair safe access and use of traditional lands and food sources, as per Section 35 (Aboriginal and Treaty Rights) of the Constitution Act 1982. CBM programs have placed emphasis on sampling relatively accessible sites at rivers and open-drainage lakes. Several studies have assessed aspects of lake hydrology and ecology (e.g., Peters et al., 2006a,b; Wolfe et al., 2007b; Gibson et al., 2015; Connor et al., 2018; Straka et al., 2018; Bush et al., 2020; Ward et al., 2020, 2021), but a sustained long-term monitoring program has yet to be established for lakes spanning the full gradient of hydrological connectivity. Systematic measurements at the relatively inaccessible perched lakes are particularly important because they are at the forefront of concern for aquatic ecosystem degradation by alteration of the delta's flood regime, climate change, and atmospheric deposition of contaminants (e.g., PADTS, 1996; Prowse and Conly, 1998; MCFN, 2014; IEC, 2018; WBNP, 2019).

To address this long-held need, we embarked on an extensive 7-year research project during 2015-2021 to provide the foundation for achieving principles of effective lake monitoring. The research involved systematic collection of water-depth measurements and samples of water, sediment and biota at 60 lakes that are spatially dispersed and span the hydrological gradient from open- to closed-drainage. Nineteen culturally important 'Muskrat Survey' lakes, monitored in winter for variation in water depth and muskrat abundance by Indigenous CBMs and WBNP, were included to enhance engagement and relevancy with rightsholders and stakeholders. The methodologies employed in the project evolved from knowledge gained during prior spatial surveys of water for isotope composition and chemistry and surficial sediment for contextualizing paleolimnological research initiated in 2000 to assess the relative roles of Peace River flow regulation and climate on observed drawdown of the delta's perched lakes since the 1980s (reviewed in Wolfe et al., 2012), and approaches applied elsewhere (e.g., Wolfe et al., 2007a; Brock et al., 2009; Turner et al., 2014; for further details on the Genesis of the Study, see Appendix 1). As concern grew for pollution of aquatic ecosystems by rapidly expanding oil sands development in the Lower Athabasca River

watershed (Kelly et al., 2009; Gosselin et al., 2010), our research in the PAD broadened to address critical reviews of monitoring programs in the Alberta Oil Sands Region (AOSR; Gosselin et al., 2010; Dillon et al., 2011). These reviews identified the absence of knowledge of natural concentrations of substances of concern as a key limitation undermining an ability to determine the extent of contamination by oil sands development. We demonstrated that this limitation could be addressed by following a recommendation of the Federal Oil Sands Advisory Panel (Dowdeswell et al., 2010) to employ paleolimnological analyses at floodplain and upland lakes in the PAD to define pre-disturbance baseline concentrations of substances of concern supplied via fluvial and atmospheric pathways, respectively. These baselines were then used to quantify the enrichment of contaminants since onset of oil sands development in stratigraphic intervals and surface sediment (Hall et al., 2012; Wiklund et al., 2012b, 2014).

Here, we synthesize key evidence obtained from informative hydrological measurements (water isotope composition (Remmer et al., 2020a,b; Imran et al., 2024), water-depth variation (Neary et al., 2021 [**Chapter 2**])), water chemistry measurements, and concentrations of oil-sands-indicator metal(loid)s in sediment and periphytic biofilm (Owca et al., 2020; Savage et al., 2021) during the 7-year research project (Figure 3.2). The publications listed in Figure 3.2 have mostly reported results obtained during the first five years (2015-2019) for a single type of measurement. Here, we extend the datasets through 2021 to capture a period of greater hydroclimatic variation. This provides opportunity to further evaluate sensitivity of the methodologies and identify informative metrics that respond to changes in river discharge, open-drainage network water levels, local meteorological conditions (Figure 3.3) and large-scale climate indices during the early ‘drier’ years (2015-2016), middle ‘average’ years (2017-2018), and later ‘wetter’ years (fall 2019 through 2021) when the open-drainage network rose to the highest water level since 1935 (Desrochers et al., 2023). We integrate the knowledge gained to identify aquatic ecosystem changes and probable causes, and provide recommendations for implementing a lake monitoring program at the PAD, which serve as an example of approaches that may be applied at other hydrologically complex landscapes where shallow lakes are exposed to

multiple stressors. Further details on the methodology (site selection, field sampling, lab analyses, numerical analyses) and lake water balance modelling, based on water isotope composition, are presented in Appendix 1. All primary data acquired during this study are provided in Supplementary Material 2 at: <https://cdnsiencepub.com/eprint/VFGEGNYZDAZ2JZWRBTR6/full>.

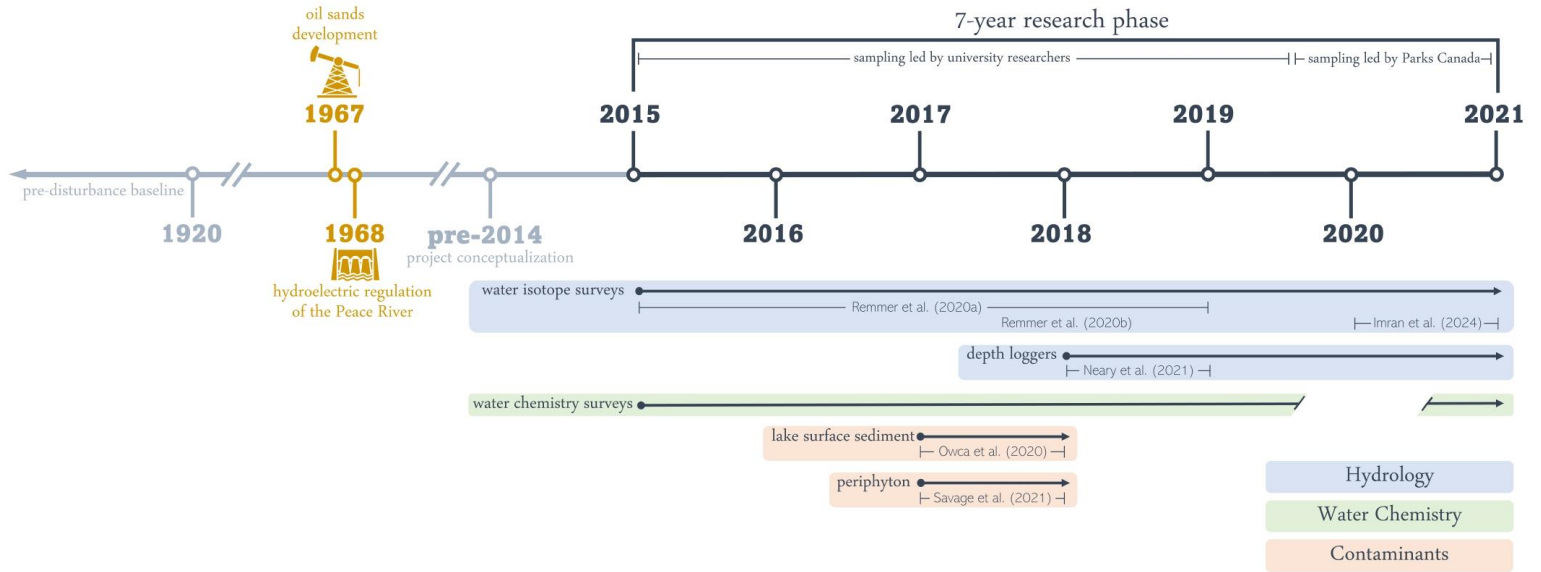


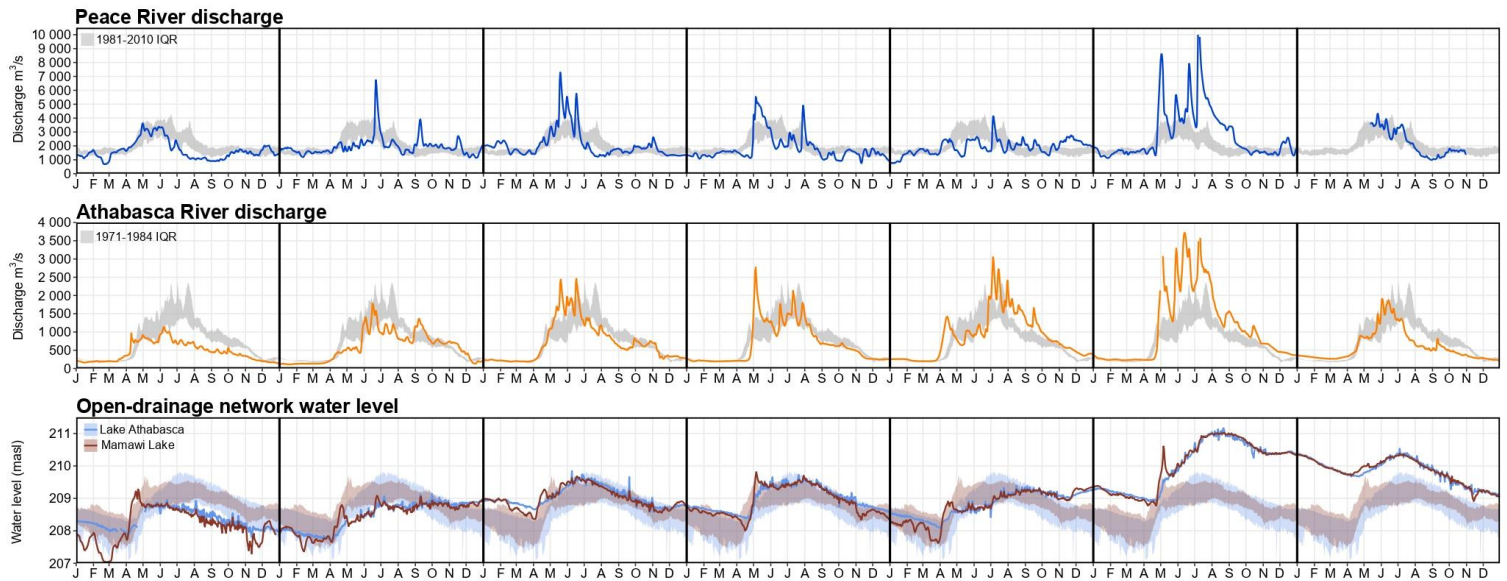
Figure 3.2. Conceptual diagram illustrating when key developments of the research occurred (2014-2021) and the inception of major energy projects. Coloured boxes indicate samples collected for hydrology, water chemistry and contaminants, as identified in the key, and the primary research articles are identified.

3.2 Hydrology

Hydrological processes are well known to exert strong control on water abundance and maintain ecosystem structure and function (Heino et al., 2020; Torgersen et al., 2021), including at the PAD (e.g., PADPG, 1973; Peters et al., 2006a,b; Wolfe et al., 2007b; Remmer et al., 2020a). Thus, we advocate that hydrological monitoring of lake water balance should form the foundation of integrated lake monitoring. As compiled in Table 2.2, several hydrological processes potentially regulate variation of lake water balances in the PAD, as conveyed in Equation (1):

$$\text{lake water balance} = \mathbf{IJF} + \mathbf{OWF} + \mathbf{ODN} + Sm + Rf - E - O \quad \mathbf{EQ (1)}$$

a) Hydrometric data



b) Local meteorological data

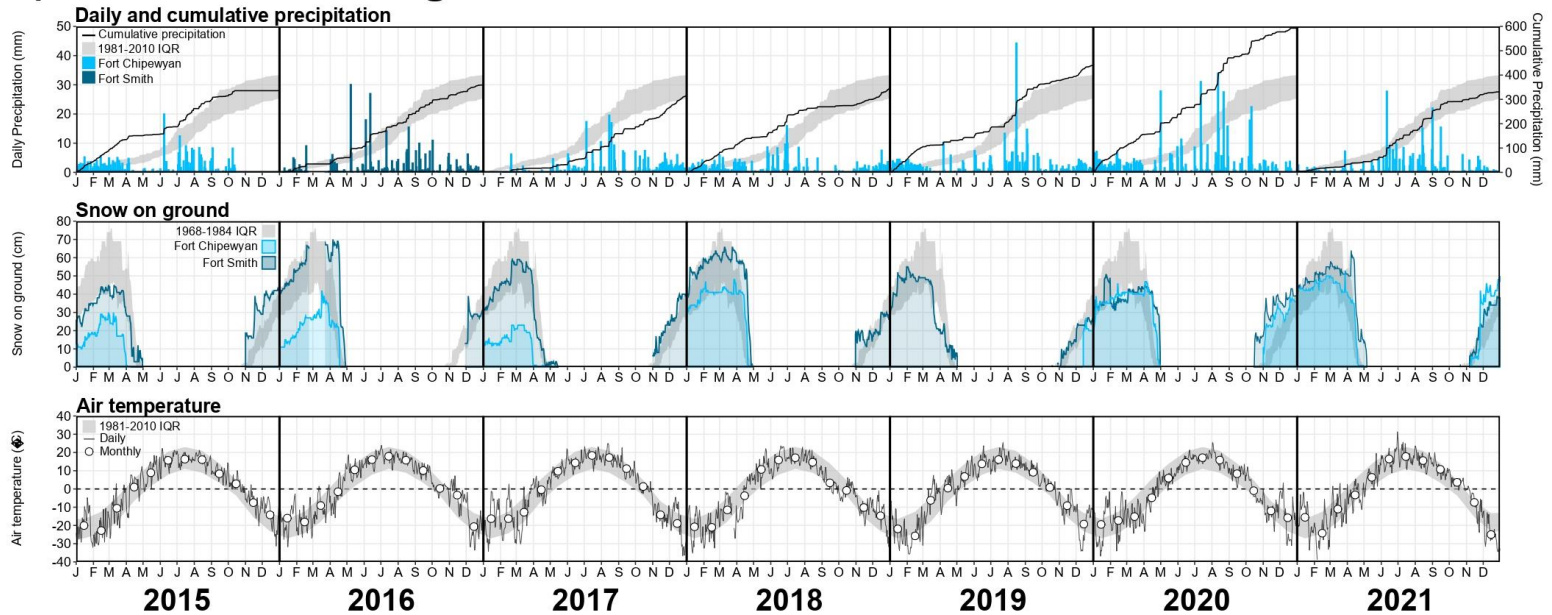


Figure 3.3. a) Hydrometric data (2015-2021): Mean daily discharge (m^3/s) for the Peace River at Peace Point (07KC001) and Athabasca River at Embarras Airport (07DD001), and daily water level (masl) of the open-drainage network (Lake Athabasca (07MD001; blue line) and Mamawi Lake (07KF003; red line)). **b)** Local meteorological data (2015-2021) recorded at Fort Chipewyan, AB (3072659) and Fort Smith, NT (2202202): Daily (vertical bars) and cumulative (black line) precipitation (mm), daily snow on ground (cm) at Fort Chipewyan (dark grey) and Fort Smith (blue), and daily mean air temperature ($^{\circ}\text{C}$). The grey ribbons in all graphs, and the red and blue ribbons on the open-drainage network level panel, present the interquartile range (IQR) of long-term measurements (typically 1981-2010, but with some exceptions as described in Appendix 1).

In EQ (1), input sources derived from river-driven processes are distinguished by bold font, and their variability over space and time strongly influences lake water balances and water levels. Episodic ice-jam floodwaters (**IJF**) from the Athabasca and Peace rivers (and their tributaries) enter lakes in spring when hydroclimatic conditions (e.g., snowmelt and rainfall runoff, air temperature, ice thickness/strength) promote dynamic breakup of river ice (Prowse and Conly, 1998; Beltaos, 2003; Lamontagne et al., 2021). Open-water floods (**OWF**) occur when Athabasca River discharge is sufficiently high to overtop levees and enter lakes during the ice-free season, which occurs more frequently than IJFs in the Athabasca sector (Prowse and Conly, 2000; Peters et al., 2006b, 2021; Neary et al., 2021 [**Chapter 2**]). Flow reversals of the Rivière des Rochers, Revillon Coupé and Chenal des Quatre Fourches occur when the Peace River becomes elevated above Lake Athabasca, and they enhance IJFs or generate localized OWFs in the Peace sector (Timoney, 2013; Timoney and Jasek, 2023). Lateral hydrological connectivity in the centrally located open-drainage network (**ODN**) occurs when river discharge is sufficiently elevated to raise water levels in the open-drainage lakes (Claire, Mamawi, Richardson) above their shorelines and inundate adjacent restricted-drainage lakes (Peters et al., 2006a,b; Neary et al., 2021 [**Chapter 2**]; Timoney and Jasek, 2023). Hydrological processes influenced by meteorological conditions include input from snowmelt (Sm) and rainfall (Rf), which interact with lake catchment characteristics and other potential input sources (i.e., IJF, OWF or ODN). Loss of water potentially occurs via outflow (O) and evaporation (E). Outflow is limited to lakes that possess an outlet channel or during short-lived episodes when floodwaters recede to the sill elevation. In the absence of IJFs and sufficient runoff contributions from the catchment, water levels of perched lakes tend to draw down by evaporation, but at highly variable rates across the landscape (Timoney, 2013; Neary et al., 2021 [**Chapter 2**]). Groundwater is a negligible component of lake water balances due to the low hydraulic conductivity of the delta's glaciolacustrine and deltaic sediments and is not included in EQ (1) (Nielsen, 1972; Prowse and Lalonde, 1996; Peters et al., 2006a). Approaches capable of tracking the occurrence and relative influence of hydrological processes in EQ (1) over space and time are essential for monitoring shifts in lake water balance and water levels and informing water resource stewardship decisions.

Remmer et al. (2020a,b) demonstrated the value and insight provided by systematic, repeated spatial surveys of water isotope composition (2015-2019) to track hydrological processes affecting lakes over space and time at the PAD, including an ability to discern the main input water sources, delineate the extent and magnitude of river flooding, and quantify the influence of evaporation on water balance. A central feature and foundation of using water isotope tracers in this landscape is display of lake evaporation-to-inflow (E/I) ratios and floodwater extent as 'isoscapes' (Figure 3.4a). The isoscapes effectively identify where lake drawdown by evaporation is most intense (orange and red areas in Figure 3.4a where E/I ratios exceed 1.0) and where lakes are most prone to flooding (blue areas where input substantially exceeds evaporation; $E/I < 0.5$). Many of the isoscapes in Figure 3.4a reveal marked spatial variability of lake water balances, where net evaporative water loss occurs in some areas at the same time when positive water balance occurs in other areas due to recent flooding (identified as areas with blue hues bordered by black dashed lines in Figure 3.4a). This occurred in spring of 2016, summer of 2016, 2017 and 2019, and fall of 2015 and 2017-2019 (8 of 21 isoscapes), and thus, is not uncommon in the PAD. In 2016, for example, relatively thin winter snowpack and unusually hot and arid conditions (Figure 3.3) generated strongly negative water balance ($E/I > 1.0$) in perched lakes across the Peace sector and the southwest corner of the Athabasca sector (Figure 3.4a), where lakes have been observed to desiccate (Wolfe et al., 2007b; Remmer et al., 2020a), whereas perched lakes adjacent to distributary channels and open-drainage lakes had strongly positive water balance ($E/I < 0.5$) due to input of river floodwater. Also, high river discharge in summer 2019 raised open-drainage network water levels (Lake Athabasca, Mamawi Lake) sufficiently (Figure 3.3) to promote open-water flooding across much of the Athabasca sector, while many perched lakes in the Peace sector possessed E/I ratios > 1.0 . In contrast, isoscapes for 2020 and 2021 display largely homogeneous low E/I ratios associated with input of river floodwater and snowmelt. As demonstrated by Neary et al. (2021 [**Chapter 2**]), addition of hourly water-depth measurements provides ability to identify the time of flooding and distinguish whether lakes received direct overbank flooding (detected as rapid lake-level rise) or back-flooding via water-level rise in adjacent open-drainage lakes (detected as gradual lake-level rise) (see

also Figure 3.4b). Also, information provided by water-depth loggers helps discern whether lake levels decline or not throughout the open-water season in the absence of flooding and identify influential rainfall events (Neary et al., 2021 [**Chapter 2**]).

The isoscapes (Figure 3.4a) and lake-depth (Figure 3.4b) measurements also reveal profound temporal variation of hydrological processes and lake water balances (Remmer et al., 2020a,b; Neary et al., 2021 [**Chapter 2**]). At interannual timescales, the isoscapes capture substantial variation in the extent of ice-jam induced flooding in spring. Flooding of perched lakes was limited to small areas of the Athabasca sector in spring of 2015-2017 and 2019, whereas larger ice-jam floods in 2018 (Remmer et al., 2020b) and 2020 inundated perched lakes across expansive portions of the Athabasca sector and smaller areas in the Peace sector. Also apparent is a trend towards greater water availability during the 7-year study period, as captured by widespread positive water balances (low E/I ratios) of perched lakes during the last two years and by the substantial proportion of lakes displaying gradual and sharp water-level rise based on hourly depth measurements. These shifts are associated with above average river discharge, open-drainage network water levels and cumulative precipitation (Figure 3.3). Higher relative humidity in 2020 and 2021 also influenced lake water balance by reducing rates of evaporation (Imran et al., 2024). The broad extent of flooding depicted in the isoscapes for 2020 (Figure 3.4a) compares well with evidence from analysis of remotely sensed imagery (Desrochers et al., 2023), which builds confidence in ability of water isotope composition and depth measurements at the network of 60 lakes to track delta-wide hydrological change. Ongoing application of remote sensing could benefit from spatial surveys of water isotope composition and lake-depth variation because they provide information about the hydrological processes responsible for observed changes in water abundance and are unaffected by cloud and vegetation cover. This knowledge is important for evaluating the effectiveness of mitigative actions such as installation of weirs in delta channels and strategic water releases from the W.A.C. Bennett Dam to trigger ice-jam flooding, as contemplated within the WBNP Action Plan (WBNP, 2019; WHC/IUCN, 2023).

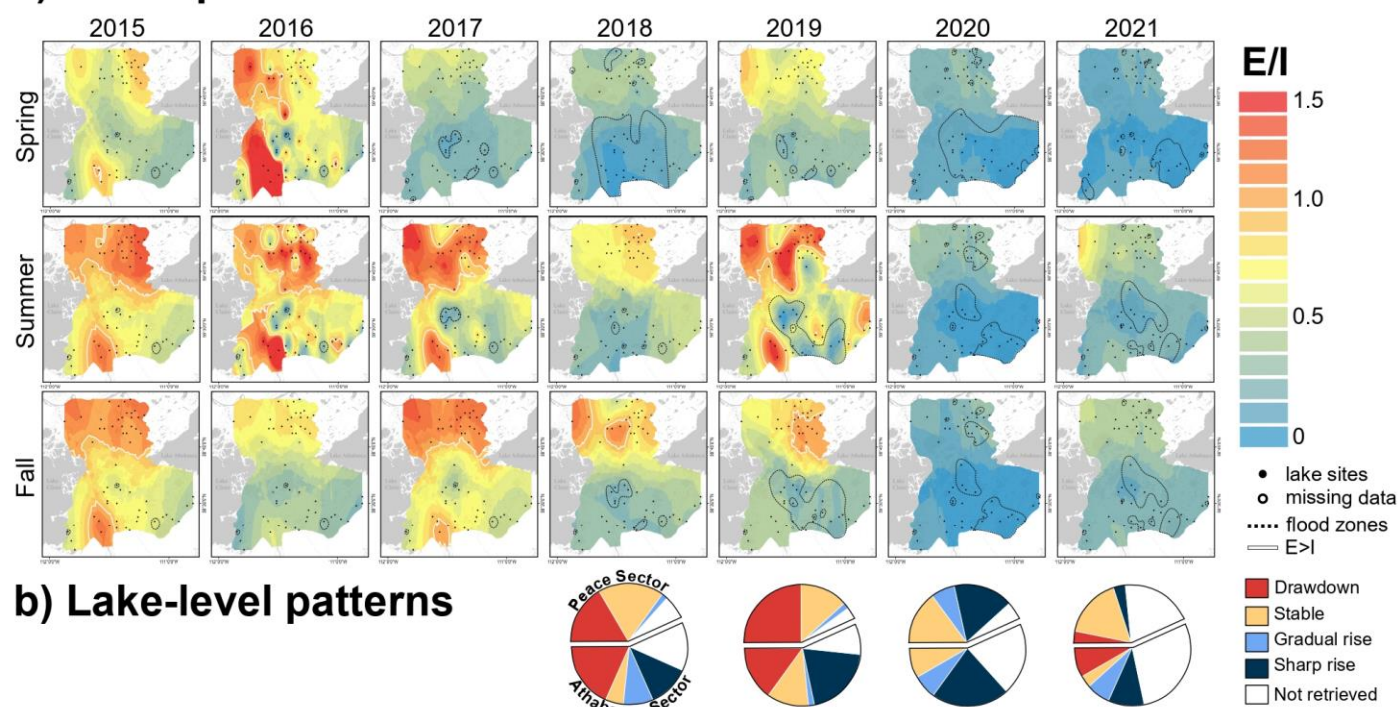
3.3 Water chemistry

Alterations to the flood regime and other hydrological processes in northern floodplains by climate change and human activities are concerning because they exert strong influence on physical and chemical properties of lakes, which ultimately affect ecosystem structure and function (e.g., Marsh and Lesack, 1996; Squires and Lesack, 2002). Monitoring of physical and chemical variables, thus, informs about consequences of hydrological change. At the PAD, water chemistry monitoring may also be used to identify if releases of oil sands process-affected water into the Athabasca River alter concentrations of nutrients and ions (e.g., Hazewinkel and Westcott, 2015).

As demonstrated by Wiklund et al. (2012a), systematic measurement of lake water clarity and chemistry across the PAD and analysis by multivariate ordination provide a means to evaluate ecosystem changes mediated by hydrological processes. Based on sampling during open-water seasons of 2003-2005 at 9 lakes, influx of river floodwater raises concentrations of suspended sediment, total phosphorus (TP), dissolved silica (DSi) and sulfate (SO_4^{2-}) and reduces water clarity, pH and dissolved organic carbon (DOC) concentration. A similar approach was employed to identify patterns of variation in lake water chemistry and turbidity and determine associations with hydrological variables during 2015-2021 (excluding 2020), which includes a larger number of lakes and years than the Wiklund et al. (2012a) study. PCA ordination explains 51.9% of variation along axes 1 (35.9%) and 2 (16.0%) (Figure 3.5). Axis 1 is associated with variation in conductivity and concentrations of dissolved inorganic carbon (DIC) and major ions (Ca^{2+} , Cl^- , Na^+ , HCO_3^- , Mg^{2+} , K^+ ; Figure 3.5a). Sample scores of the non-flooded perched lakes are broadly distributed across axis 1 (Figure 3.5b), which we interpret as reflecting differences in influence of evaporative concentration of ions in the absence of flooding, basin morphometry and catchment characteristics. Most non-flooded perched lakes are situated within largely impermeable fluvial-deltaic sediments and lack groundwater influence (Nielsen, 1972). These lakes span a gradient of solute concentration from near-equivalent to rivers to a few-fold higher (Figure A1.1), which accounts for their broad distribution along axis 1. Sand dunes, which form the southern and southeastern border of the Athabasca sector,

Hydrology

a) Isoscapes



Water Chemistry

c) Limnoscapes

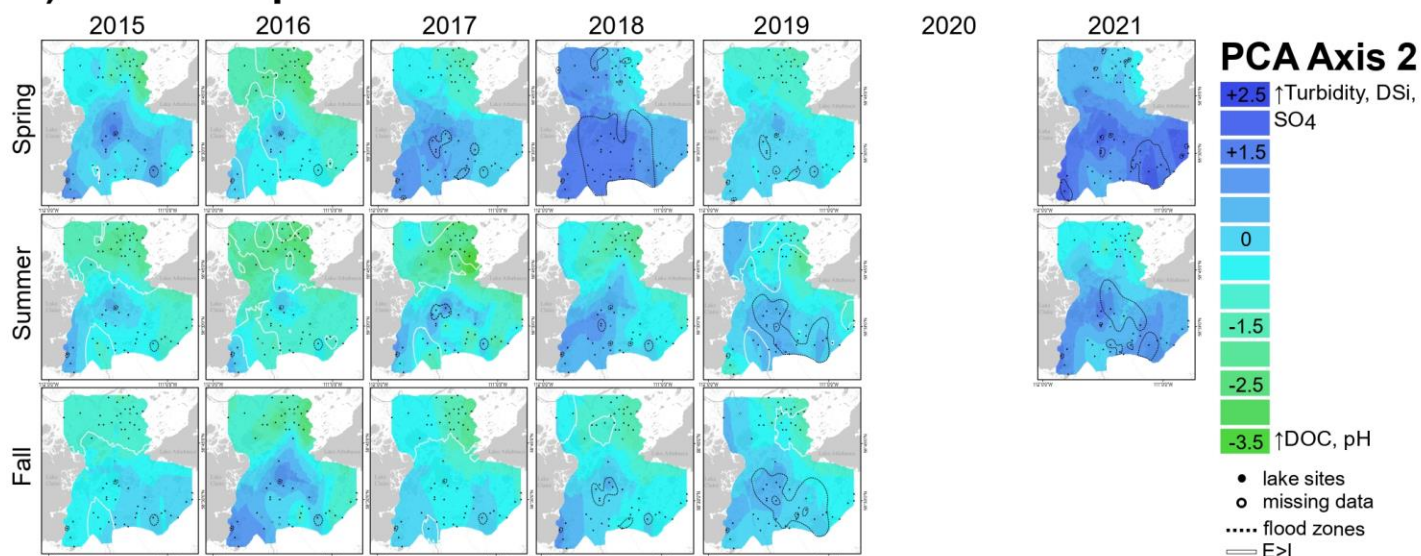


Figure 3.4. a) Isoscapes of lake water balances showing interpolated evaporation-to-inflow (E/I) ratios that are based on water isotope composition measured at 60 lakes in spring, summer and fall of 2015-2021 (2015-2019 isoscapes adapted from Remmer et al. (2020b) using an improved shapefile for the delta). Warm colours indicate high E/I ratios and cool colours indicate low ratios. White contour lines identify regions where E/I exceed 1.0, whereas black lines identify regions where lakes received river

floodwater. **b)** Pie charts showing the proportion of lakes in the Peace ($n = 26$) and Athabasca ($n = 34$) sectors that fell in four lake level categories defined by Neary et al. (2021) based on hourly water depth measurements during open-water seasons of 2018-2021. **c)** ‘Limnoscaples’ showing interpolated Principal Component Analysis (PCA) axis 2 scores of the water chemistry data (from Figure 3.5). Blue hues indicate higher Axis 2 sample scores (associated with higher turbidity and concentrations of dissolved silica and sulphate) which occur in flowing rivers, open-drainage lakes and recently flooded perched lakes. Green hues indicate lower PCA Axis 2 sample scores (associated with higher pH and DOC concentration) typical of non-flooded perched lakes. Locations of the sampled lakes are indicated as black dots. Note: samples for water chemistry analysis were not collected in 2020 due to Covid-19, and samples were missing from too many lakes in fall of 2021 for display as a limnoscape.

likely exist as a large shallow open aquifer. Input of solute-poor water from this aquifer affects a few lakes adjacent to these sand dunes (M2, M5), which are positioned to the far right along axis 1 because ionic content is more dilute than rivers. PCA Axis 2 captures differences in turbidity and chemistry of lake water along the gradient of high to low hydrological connectivity to rivers (Figure 3.5). The variables most strongly associated with this gradient (turbidity (related to suspended sediment content and water clarity), DSi , SO_4^{2-} , pH, DOC) are consistent with those identified by Wiklund et al. (2012a) more than 15 years earlier. Persistence of these relations through the 17 sampling episodes conducted during 2015-2021 identifies use of PCA axis 2 sample scores as a simplified index for monitoring changes in turbidity and chemistry of lake water across a hydrological connectivity gradient in the PAD. Geospatial interpolation of PCA axis 2 scores as ‘limnoscaples’ provides a complementary form of data display to the isoscaples and effectively illustrate substantial spatiotemporal variation in lake water chemistry during the study period (Figure 3.4c). When displayed beside the isoscaples (Figure 3.4a), strong association is evident between variation of lake water balance and chemistry. Areas of the isoscaples with orange and red hues, where lake water balance is negative (i.e., $E/I > 1.0$), typically correspond with green hues in the limnoscaples where low PCA axis 2 scores identify relatively high pH and DOC concentration and relatively low turbidity and concentrations of DSi and SO_4^{2-} . At the other end of the gradients, areas of the isoscaples with blue hues, where lake water balance is strongly positive (i.e., $E/I < 0.5$), correspond well with blue hues in the limnoscaples that identify high PCA axis 2 scores due to high turbidity and high concentrations of DSi and SO_4^{2-} (Figure 3.4a,c). Consistent with this, areas of the limnoscaples with the darkest blue hues (e.g.,

spring 2018, spring and summer 2021) correspond with areas where water isotope composition identified positive lake water balances due to input of river floodwaters.

The size and orientation of the 90% confidence ellipsoids for perched lakes and rivers in the PCA (Figure 3.5b) are broadly consistent with conclusions drawn from Wiklund et al. (2012a) and studies from other river floodplain systems (e.g., Thomaz et al., 2007), where flooding homogenizes limnological conditions of perched lakes and causes them to become more similar to the rivers (see ellipsoids for flooded perched lakes and rivers). In the absence of flooding, limnological conditions diverge and become more heterogeneous among floodplain lakes (see ellipsoid for non-flooded perched lakes) due to differences in local abiotic and biotic factors (i.e., the Flood Pulse Concept; Junk et al., 1989; Thomaz et al., 2007). Predictability of water chemistry variation across the PAD in response to occurrence of flood pulse events enhances the utility of these measurements as a contribution to lake monitoring.

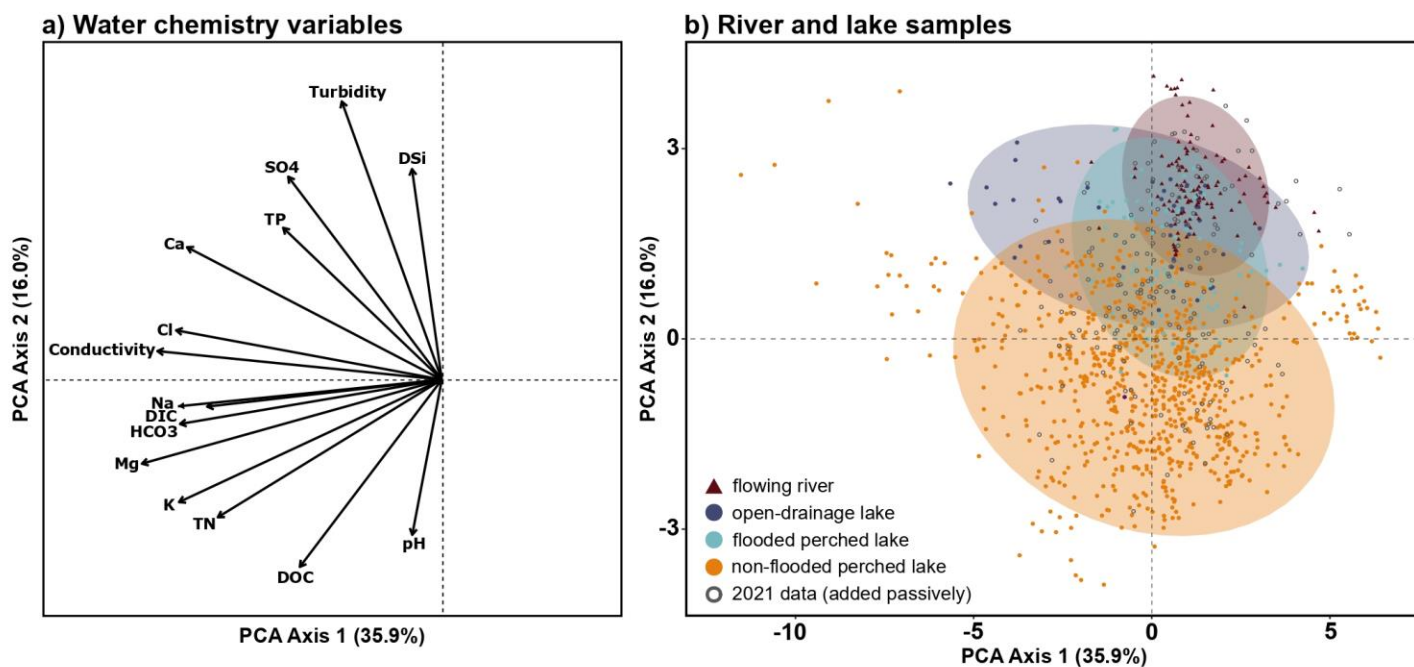


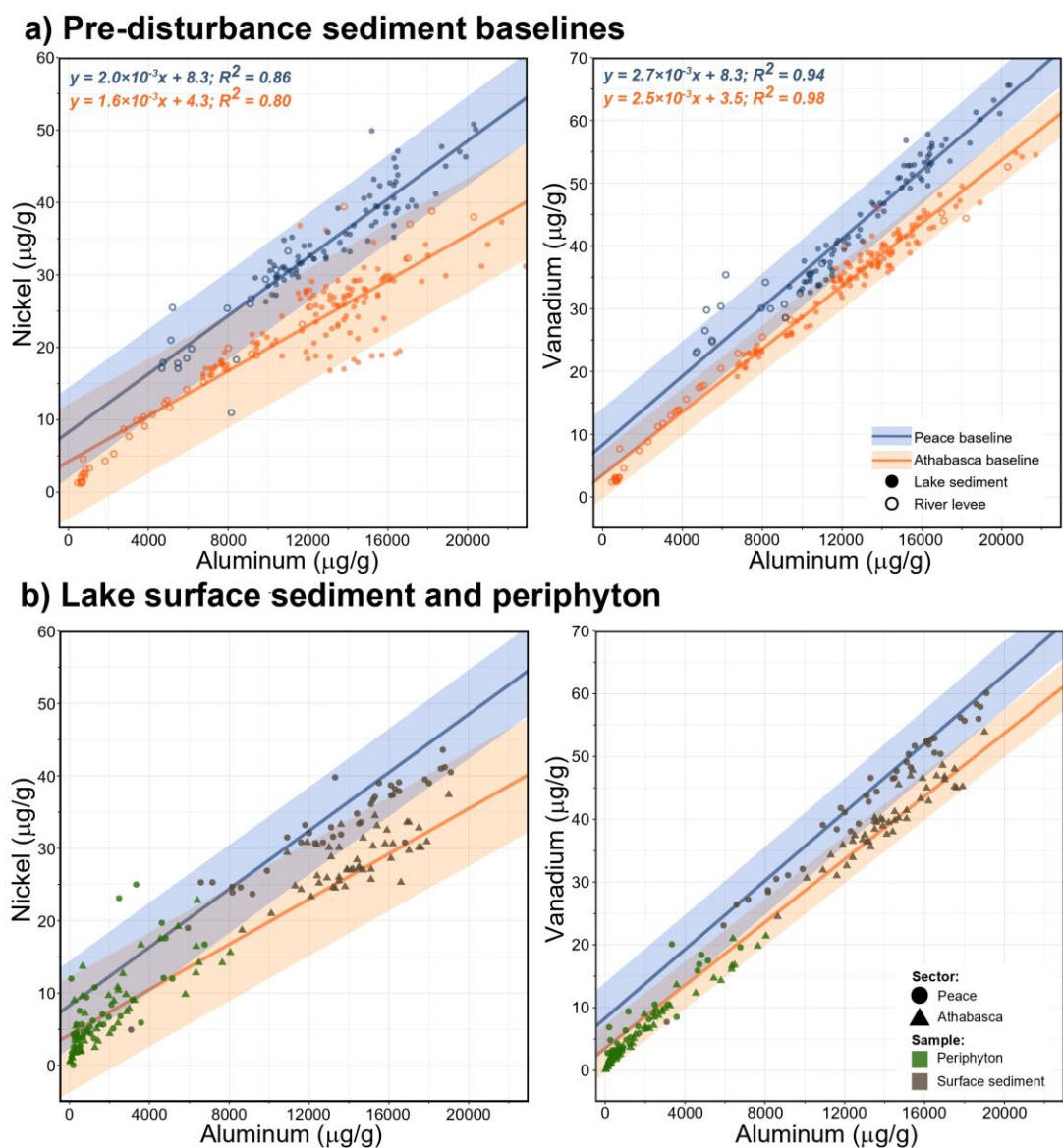
Figure 3.5. Principal components analysis (PCA) ordination plot showing **a)** vectors for 15 water chemistry variables, and **b)** the distribution of sample scores for flowing rivers (purple triangles), open-drainage lakes (dark blue circles), flooded perched lakes (light blue circles) and non-flooded perched lakes (orange circles) sampled in spring, summer and fall of 2015-2019 across the Peace-Athabasca Delta. Confidence ellipsoids (90%) are included for each category. Samples from 2021 were included passively to avoid influence of large-scale flooding during fall 2019 through 2021 (Figure 3.4a; Desrochers et al., 2023) and are identified as open circles with black outline.

3.4 Contaminants in sediment and periphytic biofilm

Monitoring is needed to address concern for aquatic ecosystem degradation in the PAD by contaminant emissions from oil sands development and other anthropogenic activities upstream and upwind (Kelly et al., 2009, 2010; MCFN, 2014, 2016; WHC/IUCN, 2017, 2023; IEC, 2018; Hebert, 2019, WBNP, 2019). Recent leaks from tailings ponds and anticipated governmental legislation allowing release of treated oil sands process-affected water to the Athabasca River are identified as serious threats to the delta's aquatic ecosystems that require ongoing surveillance (Martin, 2015; van den Heuvel, 2015; Hazewinkel and Westcott, 2015; AER, 2023; WHC/IUCN, 2023).

Here we synthesize results from assessment of enrichment of oil-sands-indicator metals nickel (Ni) and vanadium (V) in surface sediment and periphytic biofilm collected from the study lakes in 2017 and a subset of flooded lakes in 2018 relative to pre-disturbance baseline concentrations (Owca et al., 2020; Savage et al., 2021). We focus on Ni and V because they are relatively abundant in the mined bitumen, enriched in mining wastes, potentially toxic to biota, and cost-effective (relative to organic compounds) to screen for evidence of contamination (Gosselin et al., 2010; Shotyk et al., 2021). Enrichment of Ni and V is quantified by comparing geochemically normalized concentrations (using aluminum (Al)) in the surface sediment and biofilm samples to similarly normalized concentrations in lake and river levee sediment deposited before evidence of anthropogenic disturbance (Figure 3.6). Geochemical normalization accounts for variation of elemental concentration caused by differences in flow energy (Kersten and Smedes, 2002), as occurs over space and time in deltas. Pre-disturbance baselines were derived from sediment deposited before ~1920, as determined by ^{210}Pb radiometric dating methodology, because 1920 predates all mining activity in the AOSR and metal(loid) enrichment in the region by long-range atmospheric supply of metal(loid)s (Wiklund et al., 2012b; Kay et al., 2022, 2023). Our program has continuously improved the pre-disturbance baselines by increasing the number of sites and samples beyond those reported in Wiklund et al. (2012b), Owca et al. (2020), Kay et al. (2020, 2023) and Savage et al. (2021). Here we present the most updated pre-disturbance baselines established using radiometrically dated sediment cores from 7 lakes in the Athabasca sector

(PAD 26, 30, 31, 32, 71, M2, M7), 3 lakes in the Peace sector (PAD 52, 65, 67) and riverbank exposures from both rivers (Figure 3.6a). Separate baselines were derived for the Athabasca and Peace sectors because elemental concentrations are naturally higher in sediment transported by the Peace River (Figure 3.6a; Owca et al., 2020; Savage et al., 2021; Kay et al., 2023). High coefficients of determination for the relations between Ni and V with Al ($R^2 = 0.80$ - 0.98) in pre-disturbance sediment demonstrate the effectiveness of Al as a geochemical normalizer (Figure 3.5a).



samples (open circles) deposited before ~1920. **b)** metal concentrations in lake surface sediment (brown) and periphyton (green) from the Peace (circles) and Athabasca (triangles) sectors collected in 2017 and 2018. Baseline data are an amalgamation from Owca et al. (2020), Savage et al. (2021) and Kay et al. (2023), periphyton data are from Savage et al. (2021), and surface sediment data are from Owca et al. (2020).

Al-normalized Ni and V concentrations in surface sediment and periphytic biofilm samples collected in 2017 and 2018 fall close to the pre-disturbance baselines and mostly within the 95% prediction intervals, indicative of little to no enrichment because values remain largely within the range of natural variation (Figure 3.5b), consistent with the findings of Owca et al. (2020) and Savage et al. (2021) that used different iterations of baseline data. This conclusion is also consistent with evidence presented in time series for concentrations of Ni, V, total mercury, and other metals in sediment cores from river-proximal lakes in the Athabasca sector (Kay et al., 2020, 2022, 2023). Periphytic biofilm samples have lower concentrations of Ni, V and Al than sediment, presumably because they are diluted by higher organic matter content in biofilm.

Enrichment factors (EFs) are commonly used to quantify how much the concentration of a substance of concern has increased relative to pre-disturbance baseline (Böes et al., 2011; Birch, 2017). An EF of 1.0 identifies no change relative to baseline, whereas EFs of 2 and 5, for example, represent 2-fold (doubling) and 5-fold increases above baseline, respectively. Thresholds and terminology presented in a widely cited comprehensive review by Birch (2017) provide internationally consistent categorization of the degree of enrichment above pre-disturbance baselines, where EFs ≤ 1.5 represent the range of natural variability or ‘pristine conditions’, EFs >1.5 – 3.0 represent ‘minimal enrichment’, EFs >3.0 – 5.0 represent ‘moderate enrichment’, EFs >5.0 – 10.0 represent ‘considerable enrichment’ and EFs >10.0 represent ‘severe enrichment’. EFs and the Birch (2017) thresholds have been used to evaluate paleolimnological records in the PAD (Kay et al., 2020, 2022, 2023) and Lower Athabasca River (Klemm et al., 2021).

EFs computed for Ni and V in surface sediment samples collected from the lakes in 2017 and 2018 are close to 1.0 and below the threshold for ‘minimal enrichment’, identifying no enrichment above pre-disturbance concentrations (Figure 3.6c). Display of the data as EFs

in Figure 3.7c enhances communication of this finding, beyond presentation of the base data as scatterplots in Figure 3.6b. The methodology can be applied to future samples collected at regular intervals (e.g., every 3 to 5 years), and targeted sampling after widespread lake flooding (as was done in 2018), to assess if EFs rise in response to further expansion of oil sands development, accidental releases of wastewater (as occurred in 2022 and 2023; AER, 2023), and anticipated release of treated wastewater into the Athabasca River should new legislation be enacted (Martin, 2015).

3.5 Integration of hydrology, water chemistry and assessment of contaminant enrichment with hydrometeorological data: Application for ongoing monitoring

Integration of key metrics from the above three sections provides opportunity to assess associations among lake water balance, water chemistry and contaminant enrichment in the PAD and their relations with hydrometeorological conditions. To achieve this, we display the distribution of values as indices during 2015-2019 at each lake for water balance (lake E/I ratios) and water chemistry (PCA axis 2 scores) using boxplots (Figure 3.7a,b). These results capture the time before unusually widespread flooding (Figure 3.4a, 3.5), high precipitation (Figure 3.3b) and high relative humidity (Imran et al., 2024) in 2020-2021, which lowered E/I ratios across the delta (Figure 3.4a). The contaminant enrichment index is based on Ni and V EFs for surface sediment samples obtained in 2017 and 2018 and are displayed as dot plots, because there are 1 or 2 measurements per lake (Figure 3.7c).

Boxplots of the E/I ratios illustrate the broad range of water balances captured by the 60 lakes, from near-zero (where inflow $\gg$ evaporation) to >1.0 (where evaporation exceeds inflow) (Figure 3.7a). Also apparent is that E/I ratios are higher and more variable in the Peace sector, although a few lakes in the Athabasca sector also have high and variable values (e.g., PAD 19, 20, 21, 22, 37 [Jemis Lake]). Mean E/I ratios exceeded 1.0 at 10 lakes in the Peace sector (PAD 3, 5, 12, 17, 50, 52, 57, 58, M15, M19) versus one lake in the Athabasca sector (PAD 19). These lakes may serve as key long-term monitoring sites for early detection of perched lake drawdown. Indeed, we witnessed two lakes with the largest range of E/I ratios progressively desiccate during 2015-2017 (PAD 20 (Athabasca sector), 9 (Peace sector)). At the other end of the gradient, 10 lakes have consistently low E/I ratios. They include the three

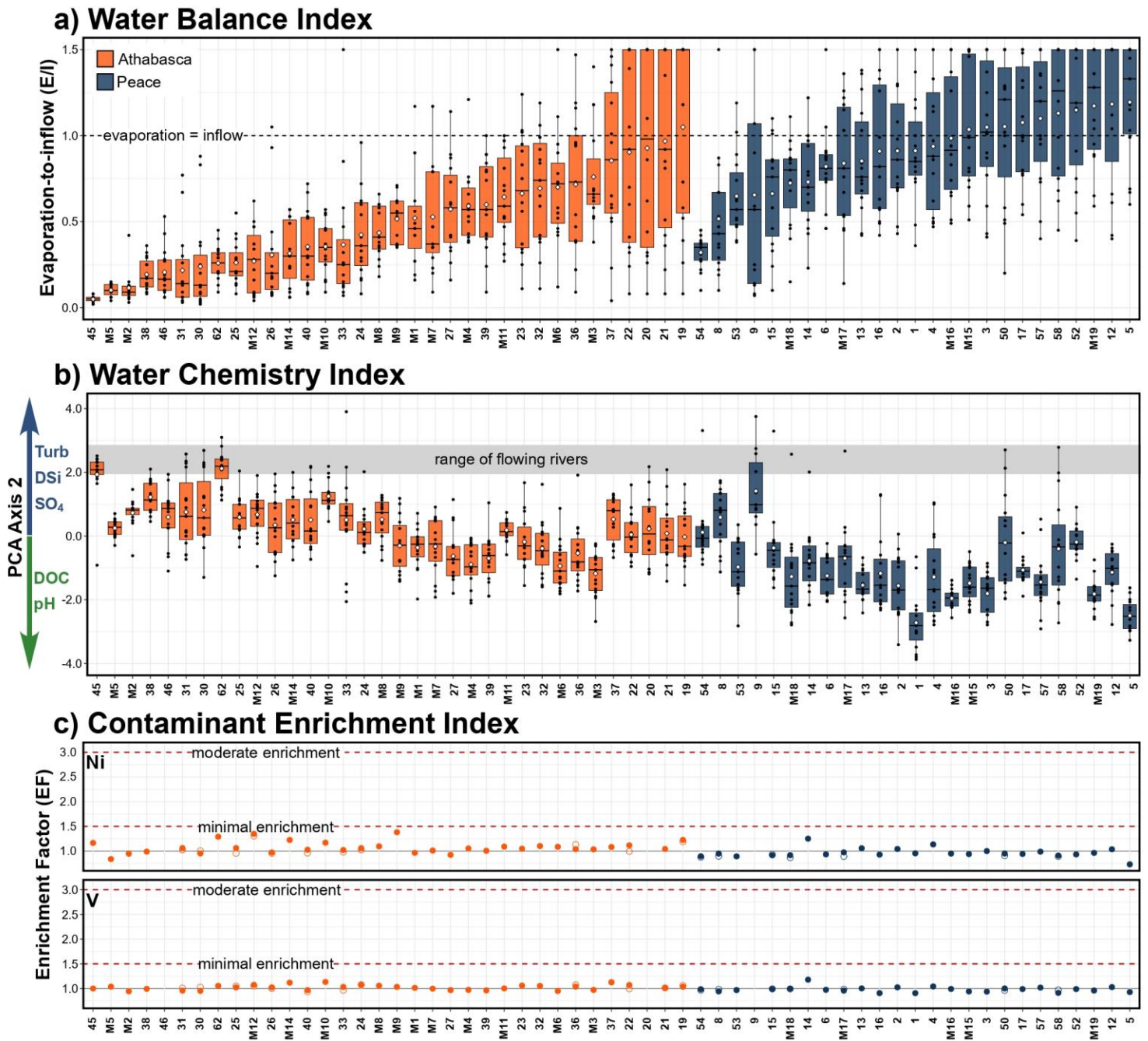


Figure 3.7. Distribution of index scores at each of the study lakes for **a)** Water Balance (based on lake E/I ratios), **b)** Water Chemistry (PCA axis 2 scores) and **c)** Contaminant Enrichment (Ni and V EFs). Boxplots in **a)** and **b)** display values obtained during 15 sampling episodes in spring, summer and fall of 2015-2019 (median as horizontal black lines, mean as open circles). The 'Contaminant Enrichment Index' in **c)** is based on Enrichment Factors (EF) computed for surface sediment collected from all lakes in 2017 (solid circles) and 20 recently flooded lakes in 2018 (open circles) relative to pre-disturbance baseline. Lakes are distinguished according to sector (orange = Athabasca sector, blue = Peace sector), and they are presented in ascending order based on their mean E/I ratio (white circles).

large open-drainage lakes (PAD 38 [Richardson Lake], 45 [Mamawi Lake], 62 [Lake Claire]), several highly flood-prone perched lakes in the Athabasca sector (PAD 25, 30 [Mamawi Pond], 31 [Johnny Cabin Pond], 46 [Otter Lake]), and two lakes uniquely located adjacent to sand dunes at the Athabasca sector's southern margin (M2 [Marta's Lake], M5 [Big Egg Lake]). With the exception of PAD 25, all the highly flood-prone perched lakes are adjacent to distributaries that convey flow augmented since 1982 by the Embarras Breakthrough, a natural avulsion that directs increasing flow from the Athabasca and Embarras rivers via Cree/Mamawi Creek (Figure 3.1; Wolfe et al., 2008; Timoney, 2013; Kay et al., 2019, 2024; Wang et al., 2023). Most of the open-drainage and flood-prone perched lakes possess water chemistry index scores that overlap with or approach the range of values for the flowing rivers (Figure 3.7b). M2 and M5 are exceptions. Their intermediate scores, suggest unique input of water from adjacent sand dunes lowers pH, DOC concentrations and E/I ratios. Despite the marked range of lake water balance and water chemistry captured at the 60 lakes over 15 sampling episodes, EFs for Ni and V, based on surface sediment obtained in 2017 and 2018, span a narrow range (1.0-1.4) below the threshold for 'minimal enrichment' (1.5; Birch, 2017). Mean EFs of 1.0 for Ni and V in lakes of both sectors of the delta (Figure 3.7c) and strong ability (>99% power) to detect minor enrichment (10% enrichment, or a rise in EF by 0.1 units) in lake surface sediment provide evidence for no detectable contamination by upstream industrial activities (Owca et al., 2020). The average and range of environmental conditions, as defined here for each lake during 2015-2019, can serve as reference points for comparison against future monitoring data to quantify the extent of change caused by contaminant emissions and hydrological processes.

Previous studies have suggested associations exist between climatic conditions across the vast Peace and Athabasca river watersheds and hydrological processes operating in the delta (Prowse et al., 2006; Wolfe et al., 2011, 2012; Timoney, 2013; Timoney and Jasek, 2023; Kay et al., 2024). The 7 years of data generated by the research support this contention, based on exploration of the influence of large-scale climate systems (Figure 3.8). Generalized Additive Model (GAM) trendlines of time series of water balance and water chemistry indices for the study lakes display strikingly common patterns with climate

indices for the Pacific Decadal Oscillation (PDO) and El Niño Southern Oscillation (ENSO; Oceanic Niño Index (ONI)). These climate indices characterize modes of low frequency ocean-atmosphere circulation well known to affect variation of hydroclimatic conditions over broad areas of North America, including western Canada (Whitfield et al., 2010; Shabbar et al., 2011). The first two years of the study (2015-2016) include strongly positive values of PDO and ONI, when a strong El Niño phase of ENSO produced warmer winters and greater summer moisture deficits in western Canada (Szeto et al., 2016; Li et al., 2018). Extensive wildfires occurred in the region in 2016 (e.g., Horse River fire at Ft. McMurray), which was then the globally hottest year on record (WMO, 2017; Tan et al., 2018). Peak water balance and water chemistry index values occurred in both sectors of the PAD during 2015-2016, and the GAM-estimated mean E/I ratios exceeded 1.0 for lakes in the Peace sector signifying widespread negative water balance. During 2017-2019, climate indices were near-neutral, and they correspond to average values for the GAM trendlines for the water balance and water chemistry indices. A marked shift to strongly negative values of the climate indices occurred after 2019, which denote onset of a 3-year La Niña phase (2020-2022) that generated cooler winters, thicker snowpacks and greater summer precipitation (Nafziger et al., 2021; Jones, 2022; Jiang et al., 2023). Correspondingly, the GAM trendlines for the water balance and water chemistry indices declined to the lowest values of the 7-year record. Mean E/I ratios fell to below 0.25 for lakes in both sectors of the delta, indicating a marked shift to strongly positive water balances during open water seasons of 2020 and 2021 due to widespread flooding of lakes and wet, humid meteorological conditions.

Throughout the study period, the GAM trendlines for the water balance and water chemistry indices show coincident changes for lakes in the Peace and Athabasca sectors despite that Peace River flow is regulated whereas Athabasca River flow is not (Figure 3.8). Variability of the water balance index trendline is greater for the Peace sector, including greater decline of values in spring of 2018 when ice-jam flooding was extensive in the Athabasca sector but not in the Peace sector (Figure 3.5a), a feature which demonstrates the strong influence of climatic forcing on perched lake water balance in the Peace sector via input of snowmelt and

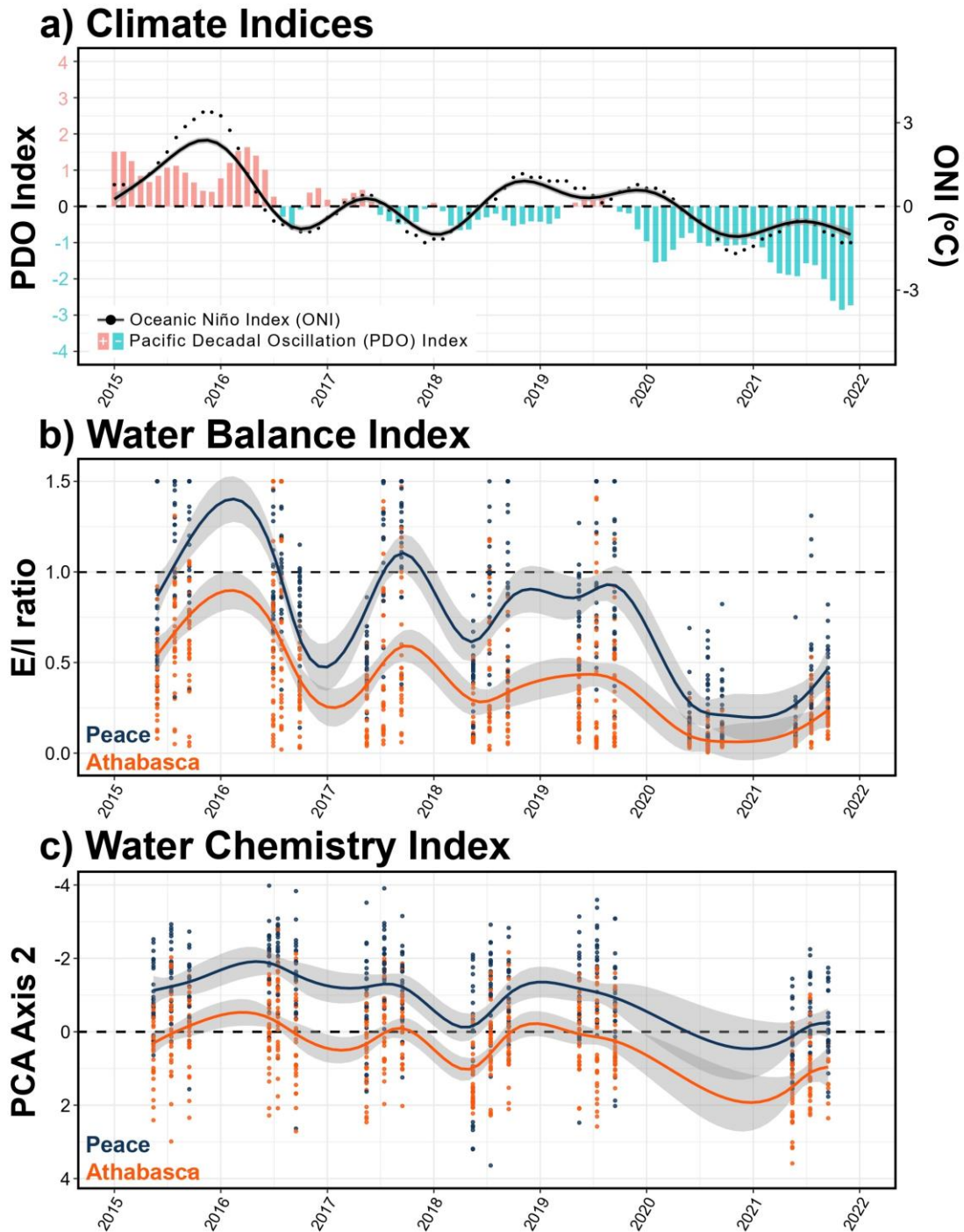


Figure 3.8. Timeseries (2015-2021) for **a)** Climate Indices (3-month running means for the Pacific Decadal Oscillation (PDO) Index (positive values in pink, negative in blue) and the Oceanic Niño Index (ONI) in the Nino 3.4 region (black dots, and GAM trendline as solid black line), and **b)** Water Balance Index (E/I ratios) and **c)** Water Chemistry Index (PCA axis 2 scores) for lakes in the Peace (blue) and Athabasca (orange) sectors of the Peace-Athabasca Delta with GAM trendlines. All GAM trendlines were computed with $k=13$. PDO data were retrieved from: <https://www.ncei.noaa.gov/pub/data/cmb/ersst/v5/index/ersst.v5.pdo.dat>. ONI data were retrieved from: <https://www.ncei.noaa.gov/access/monitoring/enso/sst>.

rainfall runoff. Also, the water balance index trendlines are consistently higher for the lakes in the Peace sector than the Athabasca sector, which captures well the greater influence of floodwaters on lakes in the Athabasca sector due to its lower topographic relief and the greater influence of evaporation on lakes in the Peace sector. Importantly, correspondence between shifts in water balance and water chemistry indices, two independent measures, and climate indices demonstrate the sensitivity and predictability of lake ecosystems to hydroclimatic shifts. These associations may be used to anticipate lake responses in the PAD to future hydroclimatic conditions, as has been done for the Canadian Prairies (Bonsal et al., 2017) and the North Saskatchewan River Basin (Anis et al., 2021).

3.6 Summary and recommendations

Here, we have presented the most informative results and products achieved during a 7-year research phase to guide a long-term aquatic ecosystem monitoring program for lakes of the internationally recognized, hydrologically complex Peace-Athabasca Delta that address pressing concerns for degradation by upstream industrial development. The research phase coincided with a period of marked hydroclimatic variation. This provided opportunity to evaluate the effectiveness of the methodologies and identify informative metrics, and their response to changes in hydrometric, meteorological and large-scale climatic drivers. Serendipitously, hydrological conditions during the 7-year research phase captured a range generally equivalent to the historical (1931-present) range of water level variation at Lake Athabasca (Figure A1.2). Metrics for lake water balance, water chemistry and contaminant enrichment are amenable for display in several formats to inform about status and trends over both space and time, assist with identifying causes of change, and should resonate with decision makers. Geospatial interpolation as isoscapes and limnoscapes captures the considerable spatial and temporal variation in lake water balances and water chemistry across the delta, respectively, which closely track shifts in hydroclimatic conditions. They also inform that concentrations of key indicator metals of oil sands development (Ni, V) remain within the range of natural variation that existed before 1920.

Australia represents a comparable jurisdiction where leading-edge studies have been conducted to guide implementation of effective ecosystem monitoring programs (Eyre et al.,

2011; Anderson et al., 2014; Sparrow et al., 2020; Lindenmayer et al., 2020). In Lindenmayer et al. (2015), a series of recommendations are provided to address questions that may emerge in the coming decades. Their philosophical approach is grounded by the following statements:

“It is critical to start measuring key attributes of ecosystems – and the human and natural process affecting them – now, so that we can track the trajectory of change over time. This will facilitate informed choices about how to manage ecological changes (including interventions where they are required) and promote better understanding by 2050 of how particular ecosystems have been shaped over time. There will be considerable value in building on existing long-term monitoring programmes because this can add significantly to the temporal depth of information (p. 213).”

We also share this point-of-view as it aligns precisely with the motivation for the research we have conducted. The PAD is an internationally renowned landscape long deserving of a lake monitoring program. Below we offer four key recommendations for operationalizing the methodology presented in our synthesis that are capable of evaluating outcomes of mitigative actions and environmental stewardship decisions currently under consideration by the WBNP Action Plan (WBNP, 2019) and tracking changes attributable to multiple potential stressors. The latter include climate change, flow regulation by the Site C dam on the Peace River and possible future releases of treated oil sands process-affected water into the Athabasca River (Hewitt et al., 2020). While the recommendations are specific to the PAD, based on the scientific evidence generated during this study, we suggest the approaches are readily transferable to other hydrologically dynamic and complex landscapes where abundant shallow lakes are threatened by multiple potential stressors (e.g., MacDonald et al., 2021; Gibson et al., 2022).

1. Monitor a sufficient number of well-dispersed lakes to support geospatial interpolation of results and span the delta’s hydrological gradients.

Display of hydrological and water chemistry data as isoscapes and limnoscapes is highly effective because maps communicate large amounts of data with spatial context in a form

readily interpretable by a range of audiences, including stakeholders and decision-makers. Geospatial interpolation of the data may become untenable, however, if the number of sites is too few and inadequately dispersed (Bowen and Good, 2015). For the maps generated in Figure 3.4 from the network of 60 lakes sampled across the PAD, the Moran's I statistic, a metric that quantifies the association of values among adjacent sites, was almost always sufficiently strong to permit use of geostatistical interpolation for the isoscapes and limnoscapes. A reduction in site density would result in lower geospatial autocorrelation and greater uncertainty in the predictive surface, weakening interpretive value. Also, potential stressors exert different effects on lakes depending on their elevation and proximity to rivers and distributary channels, which determine their position along the hydrological connectivity gradient with river water. Consequently, monitoring records will be needed from a substantial number of lakes spanning the hydrological gradients of both sectors of the delta to quantify the effects of strategic water releases from the W.A.C. Bennett Dam on the water balance of perched lakes and evaluate the effectiveness of the planned weirs on water levels of open-drainage lakes, mitigative actions in development by the WBNP Action Plan (WBNP, 2019). This would also be needed to detect hydrologic change caused by pending operation of Site C dam on the Peace River. Although we do not present 'contaminant-scapes' here, spatial display of EFs could be used to further explore the monitoring data in the event they show evidence of enrichment, similar to 'paleoenrichment-scapes' presented in Figure 5 of Kay et al. (2023), and to detect effects of accidental spills and future releases of treated oil sands process-affected water (Martin, 2015; WHC/IUCN, 2023). Incidents of wastewater seepage have already occurred in 2022 and 2023 at Imperial Oil's Kearl Mine (AER, 2023). Success of the mitigative actions and policy changes, and effects of treated oil sands process-affected water releases, cannot be adequately evaluated if only a few lakes are monitored.

As demonstrated above, the 60 lakes sampled during 2015-2021 capture broad gradients of hydrological processes, lake water balance and water chemistry, and the 5-7 years of data provide an extensive and informative foundation for ongoing monitoring. Paleolimnological analyses have been completed at nearly half of these lakes (e.g., Wolfe et al., 2012; Kay et

al., 2024), which provide knowledge on the range of natural variation, and status and trends in hydroecological conditions and concentrations of substances of concern that monitoring records can utilize (e.g., Hall et al., 2012; Wolfe et al., 2012; Kay et al., 2023).

2. Collect samples for water isotope composition at seasonal intervals and deploy water depth data loggers every year.

Information provided by systematic, repeated measurement of water isotope composition and depth variation is needed to identify the hydrological processes responsible for changes in lake water balance and water chemistry, and the main pathways of contaminant deposition (fluvial vs atmospheric). Many of the important hydrological processes play out at seasonal time scales. This includes ice-jam flooding and snowmelt runoff in spring, and open-water flooding, rainfall runoff and evaporation during summer and fall. Seasonal sampling for water isotope composition effectively captures the influence of these processes on lake water balances, and continuous depth measurements also delineate influential hydrological processes and further ensures important hydrological events are not missed when they occur between the seasonal water sampling campaigns (e.g., episodic flooding, storms). Every time a water sample is collected, we recommend that conductivity and turbidity be recorded using a handheld instrument (e.g., YSI multimeter), because this information is easily obtained and can help confirm if input water originated from river flooding or precipitation (e.g., Remmer et al., 2020a,b).

3. Perform systematic water chemistry surveys at 3-year intervals and targeted sampling shortly after widespread flooding of perched lakes.

To maintain sample integrity, water for chemistry analysis must be refrigerated after collection until lab analysis, which presents logistical challenges for remote locations such as the PAD where samples must be expedited to a distant city, and the pool of support staff may be small. Currently, analytical costs are about ten-fold higher for analysis of water chemistry than for water isotope composition. Thus, we recommend that water chemistry surveys be conducted systematically at 3-year intervals in spring, summer and fall at lake and river sites, and targeted sampling be conducted soon after widespread flooding of perched lakes to quantify the influence of floodwaters on water chemistry. This may permit

detection of an increase in salinity and ionic content at flooded perched lakes that could result from release of oil sands wastewater to the Athabasca River (Hazewinkel and Westcott, 2015; Zubot, 2021).

4. Collect surface sediment annually at Mamawi Lake and undertake targeted collection at other lakes after flooding.

The uppermost 1 cm of sediment in lakes of the PAD often contains materials that have deposited over a period of more than one year. Consequently, annual surface sediment collection is unwarranted at all lakes. We advocate for systematic annual sampling at Mamawi Lake (PAD 45), where throughflow of Athabasca and Embarras river water is continuous and sedimentation is rapid, to detect contaminant enrichment at the PAD that could arise from releases of oil sands process-affected water and other anthropogenic activities. This should be augmented by targeted sampling after flooding of perched lakes occurs, as informed by hydrological monitoring, to quantify the influence of floodwaters on concentrations of substances of concern and assess for river pollution (e.g., Owca et al., 2020). Widespread flooding of perched lakes typically occurs in spring, which is when emissions from industrial activities are released as a pulse after months of storage in the winter snowpack (Kelly et al., 2009; Alexander and Chambers, 2016). Surface sediment collection after spring flooding, thus, is an ideal time to maximize detection of contaminant enrichment, should it occur.

We have used Ni and V as cost-effective tracers of potential enrichment from the oil sands industry. However, we further recommend that roughly 50-100 grams of wet sediment should be collected from each lake to provide sufficient material to measure concentrations of metal(loid)s as well as mercury and Polycyclic Aromatic Compounds (PACs), which pose more serious risk to aquatic biota and humans (Irwin et al., 1997; Colavecchia et al., 2006; Driscoll et al., 2013), in the event that enrichment is identified and their analysis becomes warranted.

Knowledge of metal(loid) concentrations in surface sediment samples obtained after widespread lake flooding can also be used to ‘fingerprint’ which river supplied the

floodwaters (i.e., Peace vs. Athabasca) via use of a Bayesian mixing model (Kay et al., 2021, 2024). Distinguishing the source of floodwater and sediment is important in the PAD because concentrations of several substances of concern are naturally higher in Peace River sediment (e.g., Owca et al., 2020). Moreover, assessment of enrichment relative to ‘pre-release’ baseline concentrations in sediment supplied by the Peace River can serve as an ‘experimental control’ to disentangle future changes attributable to regional processes affecting both rivers (e.g., riverbank and watershed erosion by climate-induced change to river flow regimes, regional land-use changes) versus releases of oil sands process-affected water into the Athabasca River only.

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Chapter 4 | Integrating measurements of water isotope composition, depth and chemistry to assess hydrological processes affecting wetland habitat in the Whooping Crane Summer Range (Ramsar Site 240)

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4.1 Introduction

Whooping crane (*Grus americana*) breed in a remote water-rich landscape in northern Alberta and southern Northwest Territories (NWT, Canada; Figure 4.1a), recognized as a Ramsar Wetland of International Importance (Site 240: Whooping Crane Summer Range, hereafter WCSR), where abundant shallow waterbodies provide critical nesting habitat for the only wild and self-sustaining population (the Aransas-Wood Buffalo Population). Whooping crane is a Species at Risk (Endangered) in Canada and the United States (CWS–USFWS, 2007) that nearly became extinct in the 1940s when they were reduced to ~16 individuals (Johns, 2005; Gil-Weir et al., 2012; Pearse et al., 2019). The population has since increased to about 500 individuals (USFWS, 2023; Vartanian and Conkin, 2023) but remains vulnerable to effects of climate change due to its small population size and niche wetland habitat requirement (Chavez-Ramirez and Wehtje, 2012). Marked rise of water levels by runoff from thick snowpack and intense rainfall may reduce reproductive success by flooding nests, whereas drawdown of water levels increases encounters with predators (Kuyt et al., 1992). Water-level fluctuations also indirectly affect whooping crane by altering abundance of aquatic invertebrates and vertebrates that form their diet (Kuyt and Barrie, 1992). In recognition of the importance of wetland habitat for conservation of whooping crane, a long-term priority of the International Recovery Plan is to “expand [the] surface-water monitoring network to measure water level fluctuations and their effect on nesting success” (Recovery Action 1.5.2.1 in CWS–USFWS, 2007; pg. 47). More than 80% of the WCSR is protected within Wood Buffalo National Park (WBNP), where concern for degradation of whooping crane habitat has increased since Mikisew Cree First Nation (MCFN) petitioned UNESCO to downgrade the park's status to ‘World Heritage in Danger’, in

large part over cumulative threats to the ecosystems from industrial development and climate change (MCFN, 2014). A Reactive Monitoring Mission commissioned by UNESCO issued 17 recommendations to avoid the downgrade, including one to “closely monitor the entire used and potential nesting area of the Whooping Crane... so as to be able to respond to possibly changing management requirements” (WHC/IUCN, 2017; pg. 5). The WBNP Action Plan and federal response to a subsequent review by WHC/IUCN indicates commitment to “continue to monitor the nesting area of the whooping crane within Wood Buffalo NP and its wider ecosystem.” (Action WH8 in WBNP, 2019 (pg. 80), 2022 (pg. 64)).

Although international conservation agencies have identified an urgent need for long-term aquatic ecosystem monitoring in the WCSR, the knowledge base remains insufficient to design and implement a monitoring program. The few limnological and hydrological studies conducted in this remote landscape have been informative, yet mainly limited to a few sites, and do not directly serve long-term monitoring. Timoney et al., (1997) identified the ponds are typically shallow (often <50 cm) with circumneutral to alkaline pH and high sulfate concentration. Also, the gypsum-, carbonate- and organic-rich bottom sediments often support dense beds of *Chara*, a calcite-encrusted macrophytic green alga, and benthic diatom communities. In a subsequent habitat assessment, Timoney (1999) identified the ponds and shorelines provide whooping crane with long sight lines for detection of predators, abundant plant material for nests (e.g., bulrush, sedge, cattail) and access to food (aquatic invertebrates and vertebrates). During 1985-1990, McNaughton (1991) investigated the hydrological vulnerability of wetlands to dewatering operations at Pine Point Mine (Figure 4.1a), located 10 km from the northern extent of the estimated breeding range, and climatic variation. Results obtained at nine ponds in the Sass and Klewi areas of the WCSR revealed the waterbodies occupy areas of groundwater discharge in the karst terrane, but groundwater discharge was insufficient to maintain pond water levels during years with low rainfall. Consistent with this, Timoney (1999) identified that water levels fluctuate annually, with maxima after the freshet and minima in late summer. Déri-Takács et al. (2022) analyzed water isotope composition and chemistry measurements in archival records spanning 1974–1978 (Drake, 1970; Tsui, 1982; Tsui and Cruden, 1984; Weyer et al., 1978;

Weyer, 1983; Nobert and Barrie, 1986), and from samples they collected at 17 sites in 2013, including six open-water wetlands (ponds) within the Nyarling area of the WCSR, to advance understanding of groundwater flow across WBNP. The six ponds were characterized as possessing 'Ca-SO₄ type' water, suggesting influence of the gypsum-rich substrate in the Muskeg Formation.

Extensive and focused primary research is typically required to inform the design of sustainable, effective aquatic ecosystem monitoring programs. This includes determination of the sample size and variables capable of detecting important changes, identifying influential processes that cause observed changes, and informing management and mitigative actions (Lovett et al., 2007; Lindenmayer et al., 2008; 2012; 2015; Lindenmayer and Likens, 2018; Sparrow et al., 2020). For example, substantial research has recently been completed on shallow lakes in the Peace-Athabasca Delta (PAD), in southern WBNP (Figure 4.1a), to inform long-term aquatic ecosystem monitoring (Neary et al., 2024 [**Chapter 3**]). Systematic sampling and integrated analysis of water isotope composition, depth variation and chemistry at 60 shallow waterbodies (ponds and lakes) in the PAD during 2015-2021 effectively captured marked variation over space and time in water balance, water levels and water quality. Water isotope composition and chemistry was measured at seasonal timesteps (spring, summer, fall) to track variation in the influence of the major input sources (river floodwater, snowmelt, rainfall) and loss by evaporation (Remmer et al., 2020a,b), and depth was measured hourly to identify the influence of hydrological processes on water levels (Neary et al., 2021 [**Chapter 2**]; Imran, 2024). The methodology developed and applied at the PAD is likely transferrable to wetlands in the WCSR because waterbodies in both areas typically lack outflow channels and, due to their shallow depth and remote location, their water balance is strongly regulated by meteorological conditions (McNaughton, 1991; Kuyt and Barrie, 1992; Timoney et al., 1997; Remmer et al., 2020a; Neary et al., 2024 [**Chapter 3**]). A key difference is that river floodwaters provide important input to lakes in the PAD whereas groundwater exchange occurs in the WCSR, but differences in water chemistry of precipitation and groundwater are likely sufficient to discern the relative influence of these contributing waters to water balances and depth variation in the WCSR (McNaughton, 1991;

Déri-Takács et al., 2022). Thus, we anticipate integration of these measurements will generate important insights into the hydrological processes that affect the vulnerability of waterbodies in WCSR to climate change.

The 2007 Species at Risk Act Recovery Strategy for whooping crane identified six geographically distinct, contiguous ‘nesting areas’ within the WCSR, which span from northern Alberta to Great Slave Lake, as Lobstick, Alberta, Sass, Klewi, Nyarling, and North Nyarling (Environment Canada, 2007). Since this report, the WCSR has expanded, with new nesting areas documented beyond these original six (Wilson and Bidwell, 2019). Here we focus on ponds (open-water wetlands without nests) and nesting wetlands with active nests (emergent marsh with open water and adjacent shrubby fen) in the Sass and Klewi nesting areas (hereafter, SKNA) (Figure 4.1a), constituting 7% of the estimated breeding range, where known breeding has occurred for the longest duration and nest density is greatest (Allen, 1956; Kuyt, 1981, 1993; Timoney, 1999). The objective is to integrate measurements of water isotope composition, depth and chemistry at a substantial number of well-dispersed waterbodies across SKNA during the open-water season of 2022 to determine the influence of hydrological processes including groundwater connectivity, snowmelt, rainfall and evaporation. The year 2022 was characterized by above-average snowpack, a late freshet and little rainfall during summer, which generated a useful suite of conditions to capture influence of key hydrological processes on waterbodies across the landscape and assess their vulnerability to drawdown via evaporation and vertical seepage due to lowering of the water table. These measurements provide a foundation to further investigate the vulnerability of whooping crane breeding habitat to climate change.

4.2 Study Area

SKNA occupies low relief terrain downslope from the regionally elevated Caribou Mountains and within the Slave Upland Mid Boreal Ecoregion of the Taiga Plains (Ecosystem Classification Group, 2007; Figure 4.1a). The landscape is a mosaic of abundant shallow ponds and other classes of wetlands, a few rivers and forested upland underlain by Devonian-aged bedrock of the Muskeg Formation, a semi-permeable aquitard comprised

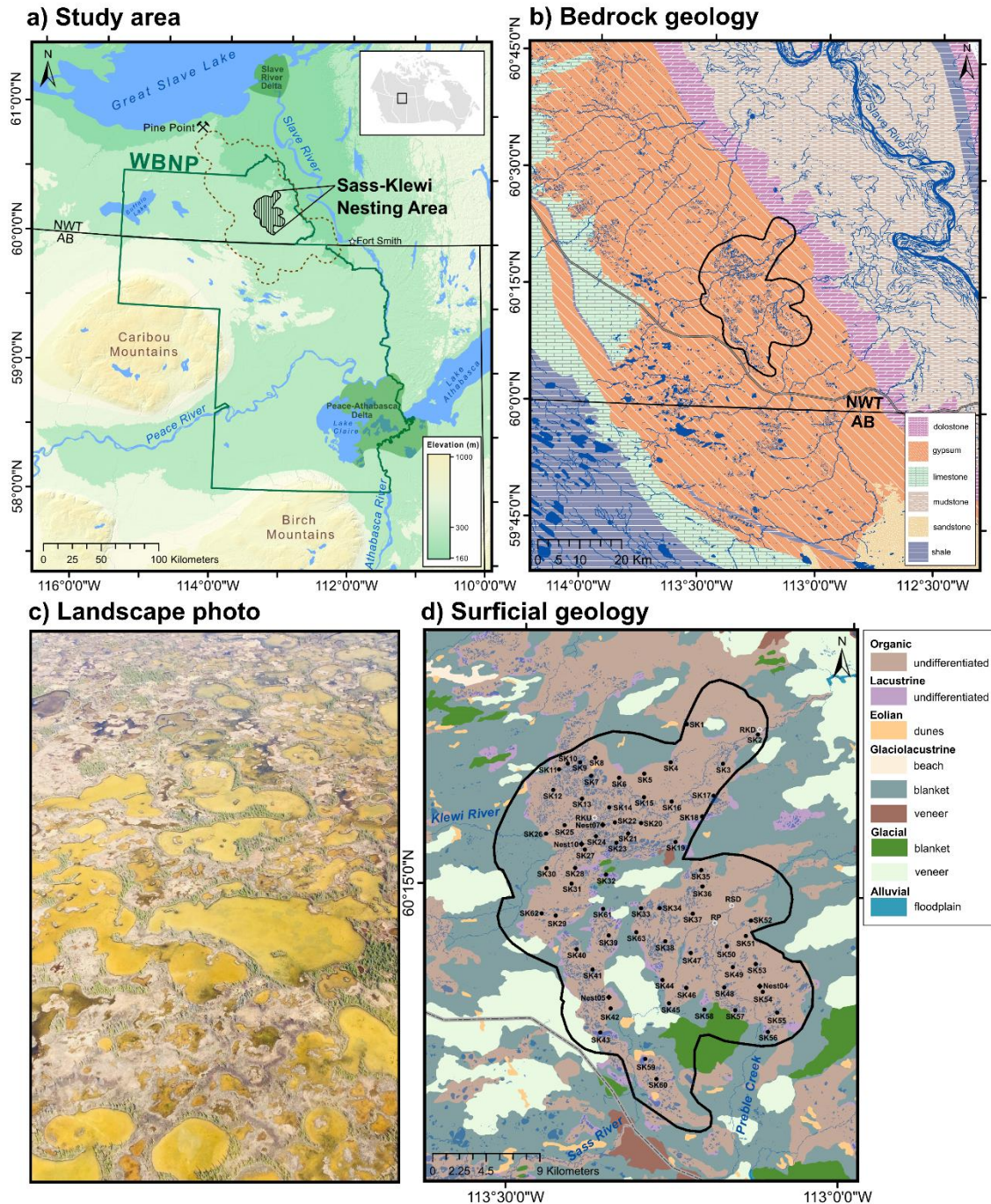


Figure 4.1. Maps showing **a)** the location of Sass-Klewi Nesting Area (black outline) within Wood Buffalo National Park (WBNP; green outline), **b)** bedrock geology and **d)** surficial geology with locations of the study sites. Panel **c)** is a photograph of the landscape taken by John Conkin in May 2019. Study sites within the Sass-Klewi Nesting Area include 63 ponds (black-filled circles), four rivers (open white circles) and four wetlands with active nests (black diamonds). Site selection criteria are provided in Table A2.1. Panel **a)** also shows the whooping crane's estimated breeding range (updated in 2022) as a brown-dotted line and the location of Pine Point Mine. The elevation basemap included in panel **a)** was retrieved from an ArcGIS® online library called 'World Elevation Terrain', the bedrock geology dataset in panel **b)** was modified from Okulitch (2006), Okulitch and Fallas (2007) and Prior et al. (2013) and the surficial geology basemap in panel **d)** was modified from NRCan (2016).

mainly of gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$) (Figure 4.1b,c; Déri-Takács et al., 2022). Immediately to the east is a zone of dolostone ($\text{CaMg}(\text{CO}_3)_2$) bedrock of the Keg River Formation, which has greater permeability than gypsum and, therefore, is considered an aquifer (Tsui and Cruden, 1984). Unconsolidated surficial sediments overlie the bedrock and are primarily organic and of glacial and glaciolacustrine origin (Figure 4.1d). Three rivers (Sass River, Klewi River, Preble Creek) flow northeastward through SKNA (Figure 4.1b,d). Ponds and wetlands (typically <1 m), described by McNaughton (1991; p. 6) as “surface expressions of conditions in the shallow groundwater zone”, occupy the surficial organic deposits, and they range in colour from cream (gypsum- and carbonate-rich) to dark brown (organic-rich) due to variation in composition of bottom sediment (Timoney, 1999). Spruce, tamarack, willows and dwarf shrubs (e.g., birch, labrador tea and bearberry) occupy the often-narrow areas of land between waterbodies, and aquatic plants (bulrush, cattail, graminoids, *Chara*) are abundant adjacent to and within the waterbodies (Allen, 1956; Novakowski, 1966; Timoney, 1999).

The study was conducted in May through September 2022, when monthly mean air temperature at the nearest meteorological station in Fort Smith, NWT (Station ID: 2202202; WMO ID: 71362; ECCC, 2023), located 70-95 km southeast of the sample sites, was colder in February and April than the 1991-2020 climate normal, and warmer than normal in June, August, September and October (Figure 4.2a). Cumulative precipitation (as snow) was above average during February through May, which resulted in almost twice the normal depth of snow (40-60 cm) in mid-April, just prior to the freshet (Figure 4.2b). Substantial snowpack (~30 cm) was sustained until the end of April, which is nearly two weeks longer than normal. Rainfall was well below average between early May and late September, which resulted in markedly lower-than-normal annual precipitation (240 mm vs 355 mm; Figure 4.2b).

4.3 Methods

4.3.1 Site selection and fieldwork

To capture the spatial diversity of hydrological conditions across SKNA and ensure study sites were well-dispersed for geospatial interpolation of the data, 63 ponds were selected

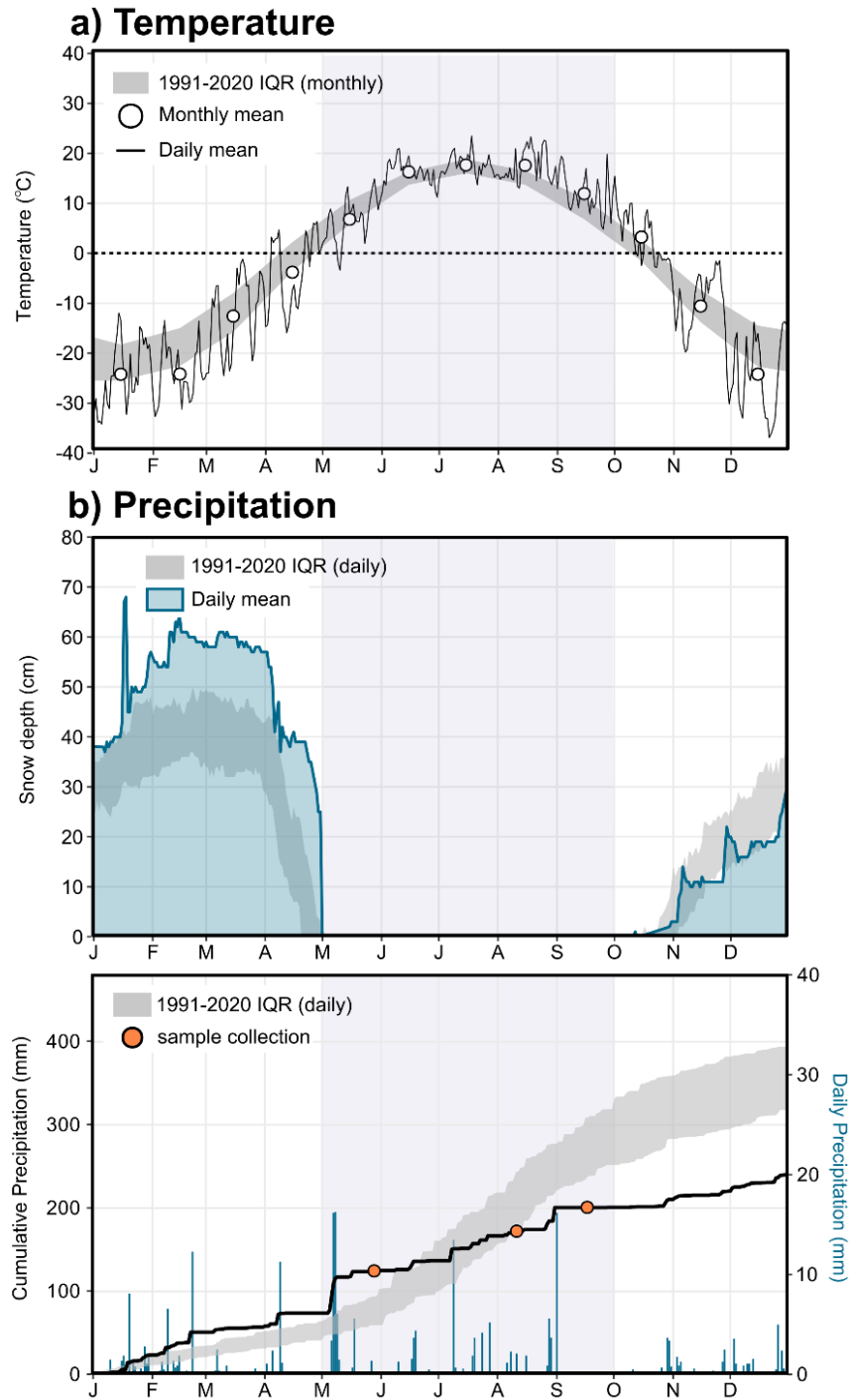


Figure 4.2. Meteorological measurements recorded at Fort Smith, NT (Climate ID: 2202202; WMO ID: 71362; ECCC, 2023) during 2022. Variables include **a)** daily and monthly mean air temperature (°C) and **b)** precipitation as daily snow depth (cm) and cumulative (black line bottom row) and daily (vertical blue bars in bottom row) precipitation (mm). Water sample collection dates are indicated by orange-filled points positioned along the cumulative precipitation line (in lower panel). Climate normal (1991-2020) data are included for comparison and the interquartile ranges (IQR) are identified as grey-shaded ribbons for all panels. The light-grey shaded rectangle included in all panels identifies the ice-free season (May 1–October 1).

from 3x3-km grid cells demarcated across SKNA using high-resolution satellite imagery acquired in 2021 (Figure 4.1d). The selected ponds spanned a range of surface area and colour (Table A2.1). The Sass and Klewi rivers and Preble Creek were also sampled to characterize the water isotope composition and chemistry of the main rivers. Due to the remote location, sampling was performed using a helicopter on fixed floats. To avoid disturbing nesting whooping cranes, ponds were not sampled if active nests were located within 500 m. Four wetlands with active nests were sampled by landing a few hundred meters away and accessing the site by foot. Sampling of the ponds and river sites occurred on May 28-29, August 9-10 and September 15-17, 2022 and sampling of the nesting wetlands occurred on May 17-19 and August 7-8, 2022. The sampling campaigns are referred to, hereafter, as spring (May), summer (August) and fall (September). They are intended to capture the effects of the main hydrological processes (e.g., input of snowmelt and rainfall, exchange with groundwater, loss via evaporation), which vary during the open-water season (following Turner et al., 2014; Remmer et al., 2020a; Neary et al., 2024 [**Chapter 3**]).

During each sampling campaign, *in situ* measurements of temperature, pH, specific conductivity and dissolved oxygen concentration were recorded at each site using a YSI ProDSS multi-parameter water quality meter, water depth was recorded using a tape measure, and water samples were collected for analysis of water isotope composition (30 mL) and chemistry (1 L). Field observations of vegetation, surface water connections to adjacent waterbodies and water levels relative to shorelines were noted at each site and photos were taken. Data loggers (Onset HOBO pressure transducers; Model U20–001–01, 4-m range, 0.3-cm accuracy) were set to record temperature and pressure at hourly intervals starting on June 1 and deployed in each waterbody during the first field campaign in mid-late May and retrieved during the last campaign (August for the nesting wetlands, September for the ponds). Data loggers were attached to the bottom of a rope connecting an anchor to a wooden float and deployed at a central deep-water area of each pond. At the nesting wetlands, the data loggers were deployed near the shore (≤ 25 cm water depth) at the base of a metal T-bar driven into the sediment. Loggers were successfully retrieved from 60 of the

63 ponds and all four nesting wetlands. They were not retrieved from ponds SK34, SK35 and SK38.

4.3.2 Laboratory analyses

Water samples were analyzed for isotope composition at the University of Waterloo – Environmental Isotope Laboratory using a Los Gatos Research Liquid Water Isotope Analyser model T-LWIA-45-EP instrument with precisions (2σ) of ± 0.8 ‰ and ± 0.2 ‰ for hydrogen and oxygen isotope composition, respectively. A suite of standards (Vienna Standard Mean Ocean Water (VSMOW), Standard Light Antarctic Precipitation (SLAP)) from the International Atomic Energy Agency were used as reference materials and duplicate samples were run at a minimum of every tenth sample. Isotope compositions are expressed as δ -values, representing deviations in per mil (‰) from VSMOW such that $\delta_{\text{sample}} = [(R_{\text{sample}}/R_{\text{VSMOW}}) - 1]$, where R is the $^{18}\text{O}/^{16}\text{O}$ or $^2\text{H}/^1\text{H}$ ratio in the sample and VSMOW. Results of $\delta^{18}\text{O}$ and $\delta^2\text{H}$ analyses are normalized to -55.5‰ and -428‰, respectively, for SLAP (Coplen, 1996).

Samples for water chemistry analysis were refrigerated prior to shipment to University of Alberta's Biogeochemical Analytical Services Laboratory for determination of concentrations of total nitrogen (TN), total phosphorus (TP), dissolved organic carbon (DOC), dissolved silica (DSi), dissolved inorganic carbon (DIC), major anions (SO_4^{2-} , HCO_3^- , Cl^-) and major cations (Ca^{2+} , K^+ , Mg^{2+} , Na^{2+}). Concentrations of TN, TP and DSi were determined by Flow Injection Analysis (TN Method: 353.2; TP Method: 4500-P-G; Dissolved Silica Method: 370.1) using a Lachat QuickChem QC8500 FIA Automated Ion Analyzer. Concentrations of DIC and DOC were determined using a Shimadzu TOC-5000A Total Organic Carbon Analyzer (Method 415.1). Ion concentrations were determined using a Thermo ICAP-6300 Inductively Coupled Argon Plasma – Optical Emission Spectrometer (Method: 200.7).

4.3.3 Data analyses

4.3.3.1 Isotope mass-balance model

We developed and applied an isotope-mass balance model to estimate the isotope composition of input water (δ_i) and evaporation-to-inflow (E/I) ratio at each waterbody for each sampling date, following approaches used in other water-rich cold regions (e.g., Yi et al., 2008; MacDonald et al., 2017, 2021; Remmer et al., 2018, 2020a; Neary et al., 2024 [Chapter 3]). Underlying the approach is development of an isotope ‘framework’, which comprises two linear relations between the isotope composition of oxygen and hydrogen. (1) The Local Meteoric Water Line (LMWL) represents the range of measured isotope compositions in precipitation collected at Fort Smith (NWT) during 1960-1969 ($\delta^2\text{H} = 6.7 \delta^{18}\text{O} - 19.2$) by the Global Network for Isotopes in Precipitation (IAEA/WMO, n.d.). (2) The predicted Local Evaporation Line (LEL) represents the expected trajectory of isotope composition in a surface waterbody fed by local amount-weighted mean annual isotope composition of precipitation (δ_P) as it undergoes evaporation, which causes progressive isotopic enrichment (Gibson and Edwards, 2002). The LEL is anchored at δ_P and extends to δ^* , the isotope composition of a terminal waterbody as it approaches complete desiccation. A key threshold on the LEL is the isotope composition of a terminal basin at steady state, where evaporation equals inflow (δ_{SSL}). Here, we used a semi-empirical approach to model and constrain δ_{AS} , the isotope composition of ambient atmospheric moisture, to obtain an LEL slope (4.3) that is near equivalent to a regression through the measured isotope compositions of the waterbodies in 2022. Further details on this approach and the equations used to determine δ_P , δ_{SSL} and δ^* are presented in Appendix 2 and the isotope mass-balance model parameters are reported in Table A2.2.

E/I ratios provide an informative water balance metric, and they were estimated using a series of isotope mass-balance equations detailed in Appendix 2. Briefly, a coupled isotope tracer approach (Yi et al., 2008) was used to estimate the isotope composition of input water (δ_i) as the intersection of the waterbody-specific evaporation line and the LMWL, where the waterbody-specific evaporation line extends from the isotope composition of evaporated water (δ_E) to δ^* and passes through the measured water isotope composition at each

waterbody ($\delta_{\text{waterbody}}$). E/I ratios were then determined using the following equation (Gibson and Edwards, 2002):

$$E/I = \frac{(\delta_I - \delta_{\text{waterbody}})}{(\delta_E - \delta_{\text{waterbody}})}$$

This approach is advantageous because it fundamentally preserves mass and isotopes during evaporation such that calculated E/I ratios using both tracers agree (Yi et al. 2008). Consistent with other studies, we assigned a maximum E/I ratio of 1.5 for waterbodies experiencing strong, non-steady-state evaporation because of increasing uncertainty of model estimates beyond this value (MacDonald et al., 2017; Remmer et al., 2020a; Neary et al., 2024 [**Chapter 3**]). Thus, reported E/I ratios potentially range from 0.0 to 1.5. Positive water balances (i.e., $E/I < 1.0$), are indicative of a waterbody whose water balance is dominated by input. In contrast, negative water balances ($E/I > 1.0$) occur when evaporation exceeds input.

4.3.3.2 Analysis of water depth measurements

Hourly measurements of absolute pressure (psi) obtained from data loggers deployed in the waterbodies were converted to water depth (m) using hourly air pressure and temperature measurements recorded by a logger placed in the air in Fort Smith (NWT) and the Barometric Compensation Assistant in HOBOWare Pro software. Using RStudio and external package *pracma* (Borchers, 2022), a moving average (24-hour) was applied to the hourly water depth measurements to reduce minor high-frequency variability. After compilation, water depths at two ponds (SK7, SK42) displayed abrupt shifts within a 1-hr interval that were likely too rapid to represent a true change in water depth. We suspect these loggers drifted to a new location, where they remained thereafter. To avoid this potentially confounding influence, the change in water depth during the 1-hr interval was subtracted from or added to the remainder of the record. The logger collected in September at SK46 was found along the shoreline, so we omit this entire record. The magnitude of water-level drawdown at each site is expressed as a percentage of the first water depth measurement on June 1st to standardize comparisons among sites with varying initial depths.

4.3.3.3 Analysis of water chemistry measurements

Principal Component Analysis (PCA) was used to explore patterns of variation in the water chemistry variables among the sites. Prior to computing the PCA, all variables except pH and DSi concentration were log-transformed to improve normality and equalize variances, and all variables were centered and standardized. The water chemistry data obtained in May were omitted from the PCA because of very low variability among sites due, presumably, to strong influence from recent input of snowmelt and possibly also groundwater discharge. One site, SK11, was a severe outlier in August and could not be sampled in September due to desiccation, thus it was also removed from the ordination. The ordination plot was created using RStudio using base R (*stats*; R Core Team, 2023) and external packages including *tidyverse* (Wickham et al., 2019) and *factoextra* (Kassambara and Mundt, 2020).

4.3.3.4 Geospatial representation of water Balance, water-level drawdown and water chemistry

Spatial variation of isotope-inferred E/I ratios, water-level drawdown, and water chemistry PCA axis 1 sample scores was explored using geospatially interpolated maps we refer to as ‘isoscapes’, ‘drawdown-scapes’, and ‘limnoscapes’, respectively. Interpolation was performed in ArcMap V10.8.2 using inverse-distance weighting.

4.4 Results and Interpretation

4.4.1 Water isotope composition and isotope-inferred E/I ratios

Isotope compositions of the waterbodies plot closely along the LEL and progress from the low end (near δ_p) in May ($\delta^{18}\text{O}$: -19.8 to -14.3 ‰; $\delta^2\text{H}$: -158.0 to -128.8 ‰) to increasingly higher values in August ($\delta^{18}\text{O}$: -19.8 to -6.2 ‰; $\delta^2\text{H}$: -158.5 to -99.8 ‰) and September ($\delta^{18}\text{O}$: -20.1 to -5.9 ‰; $\delta^2\text{H}$: -157.1 to -96.0 ‰) (Figure 4.3a; Table A2.3). The marginal silhouette plot in Figure 4.3a reveals that (hydrogen) isotope compositions in May span a relatively narrow range, which we interpret is an outcome of substantial, widespread inflow of isotopically depleted snowmelt runoff early during the open-water season. Snowmelt also may have entered shallow aquifers and raised the water table, which could have increased discharge of groundwater to waterbodies in spring. Water isotope compositions spanned

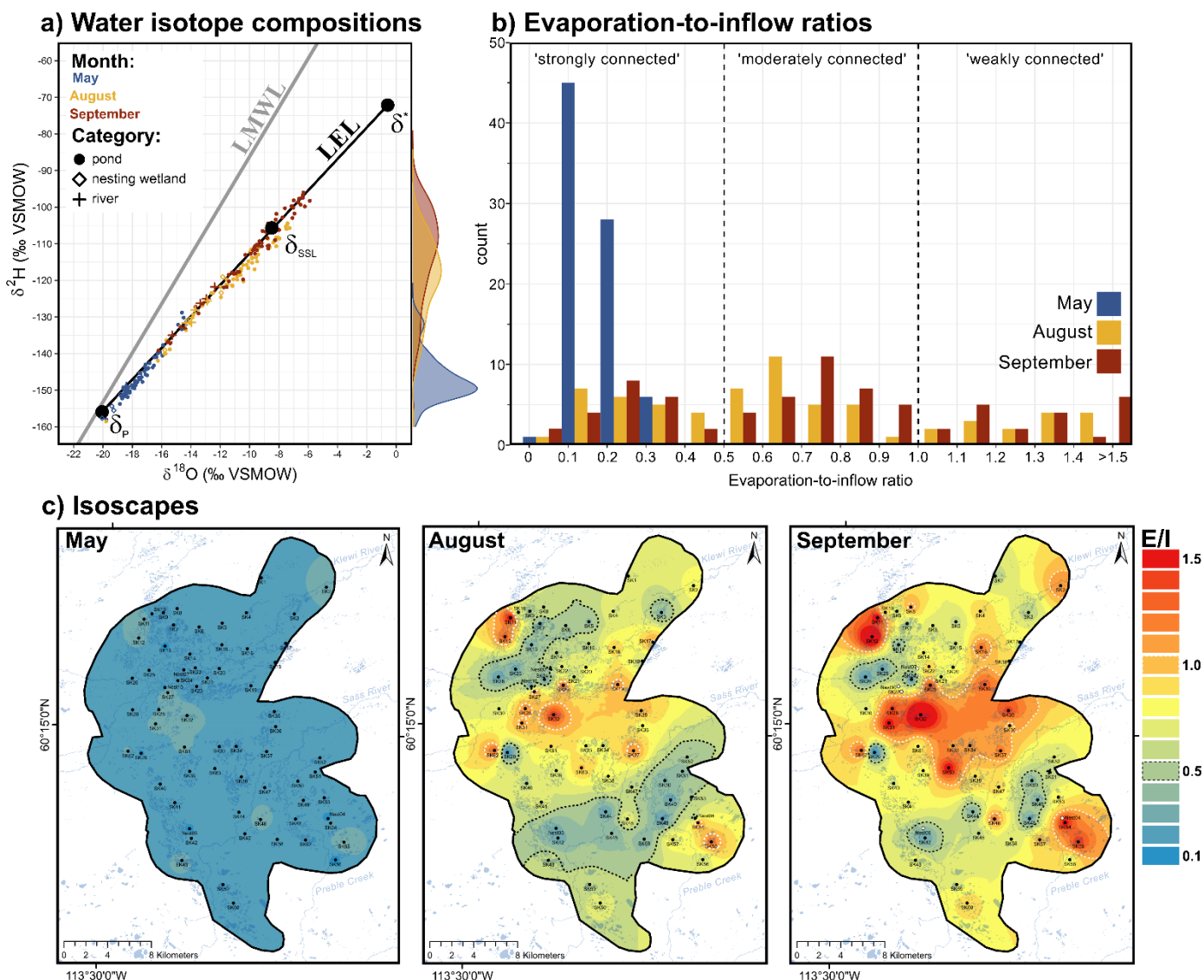


Figure 4.3. a) Isotope framework for 2022, including the Local Meteoric Water Line (LMWL; $\delta^2\text{H} = 6.7 \times \delta^{18}\text{O} - 19.2$) and Local Evaporation Line (LEL; $\delta^2\text{H} = 4.3 \times \delta^{18}\text{O} - 69.6$). The LEL is anchored at δ_P (the isotope composition of mean annual precipitation) and extends to δ^* (the isotope composition of a terminal water body as it approaches complete desiccation). Also included along the LEL is the isotope composition of a terminal lake at steady state (δ_{SSL}). Isotope-mass balance equations used to establish the framework are reported in Appendix 2, and the parameters are contained in Table A2.2. Water isotope compositions of 63 ponds and four nesting wetlands in the Sass-Klewi Nesting Area in May, August and September 2022 are superimposed on the isotope framework, and marginal histograms display the distribution of $\delta^2\text{H}$ (Table A2.3). **b)** Frequency histogram showing the distribution of isotope-inferred evaporation-to-inflow (E/I) ratios in May, August and September 2022 (Table A2.4). **c)** Isoscapes depicting spatial visualization of isotope-inferred E/I ratios in May, August and September 2022. The vertical dashed black lines in panel **b)** and the contours in panel **c)** identify threshold E/I ratios where water balance is strongly positive ($E/I < 0.5$; areas within the black-dashed lines in the isoscapes) and negative ($E/I > 1.0$; areas within the white-dashed lines in the isoscapes), and which were used to define three groundwater connectivity categories: strongly connected, moderately connected and weakly connected. Sites that did not contribute to the interpolated layer in September include SK27 and the four nesting wetlands and are identified by white circles.

broader ranges in August and September, and values exceeded δ_{SSL} at a substantial number of the waterbodies (7 and 18, respectively) due to strong influence of evaporation and scant rainfall. At the same time, isotope compositions remained low at many other waterbodies because of input of an isotopically depleted water source, which is presumed to be mainly groundwater because of the low amount of rainfall between sampling campaigns. Isotope composition of the rivers ($\delta^{18}\text{O}$: -18.5 to -12.4 ‰; $\delta^2\text{H}$: -150.2 to -121.8 ‰) varied less than the ponds and nesting wetlands and did not exceed δ_{SSL} , presumably because they receive near-constant and substantial influx of isotopically depleted groundwater and precipitation.

Isotope-inferred E/I ratios of all waterbodies are below 0.3 in May 2022, but they span the entire range of possible modelled values (0.0 to 1.5) in August and September (Figure 4.3b; Table A2.4). The waterbodies typically lack outflow channels; thus, the main hydrological processes that influence their water balance are input from precipitation and groundwater and loss by evaporation and vertical seepage (the latter via lowering of the water table). Given that rainfall and evaporation are expected to be relatively consistent across SKNA, and vertical seepage does not alter isotope composition of the water, the marked variability of E/I ratios among waterbodies in August and September must be an outcome of spatial variation in groundwater discharge.

Geospatial interpolation of the E/I ratios as ‘isoscapes’ reveals marked variation of water balances across both space and time (Figure 4.3c). Consistent blue hues across the entire landscape in May identify that E/I ratios were uniformly low (mean = 0.15) due to substantial input of snowmelt runoff, and possibly groundwater, early during the open-water season. Spatial variation of E/I ratios increased in August and September, when low E/I ratios persisted in some areas where the hues remain blue, and values rose in other areas where the hues are yellow to red. We added contours to the August and September isoscapes to identify where water balances are negative ($E/I > 1.0$; areas within the white dashed lines) and where they are strongly positive ($E/I < 0.5$; areas within the black dashed lines). The former reveals ‘hotspots’ of net evaporation across a broad, slightly more elevated central region between the Sass and Klewi rivers, and a few smaller areas north of the Klewi River and southeast of Sass River and Preble Creek. In contrast, strongly positive water balances

persist in lower lying areas adjacent to the rivers. These spatial patterns suggest that gradients exist across the landscape in the degree of connectivity between waterbodies and the shallow groundwater, which is consistent with McNaughton's (1991) conclusion that the waterbodies occupy groundwater discharge zones. Shortly after the freshet, the water table is expected to be relatively high due to infiltration of snowmelt, which was substantial in 2022 (Figure 4.2b). However, the water table likely declined throughout the summer in response to the hot, arid weather that generated substantial net evaporation (Figure 4.2a). If so, isotope composition in waterbodies located at slightly higher elevations, away from the low-lying areas and rivers, would have become enriched in response to increasing influence of evaporation as connectivity to groundwater waned, whereas continued input of isotopically depleted groundwater may have offset enrichment by evaporation in waterbodies at slightly lower elevations proximal to the rivers. Declining numbers and aerial extent of waterbodies with strongly positive water balance between August and September provides evidence for gradual decline of the water table (Figure 4.3b, c).

To further investigate the influence of groundwater connectivity over space and time, we established three categories based on the E/I ratios in August and September 2022: 1) waterbodies with $E/I < 0.5$ have strongly positive water balance and are interpreted as 'strongly connected' to groundwater; 2) waterbodies with E/I ratios between 0.5 and 1.0 have moderately positive water balance and are interpreted as 'moderately connected' to groundwater, and 3) waterbodies with $E/I > 1.0$ have negative water balance and are interpreted as 'weakly connected' to groundwater, and some of them may be disconnected from groundwater (Table A2.4). The threshold values that distinguish these categories are identified in the E/I ratio frequency distribution histogram (Figure 4.3b) and isoscapes (Figure 4.3c). We use the assignment of waterbodies to these categories in the analysis of water-level drawdown and water chemistry below to further characterize the influence of groundwater connectivity. For these analyses, we assigned waterbodies in the strongly connected category as those with $E/I < 0.5$ *in August*, when the isoscapes best capture substantial areas of strong groundwater influence before the water table declined further. Conversely, waterbodies in the weakly connected category were defined as those with E/I

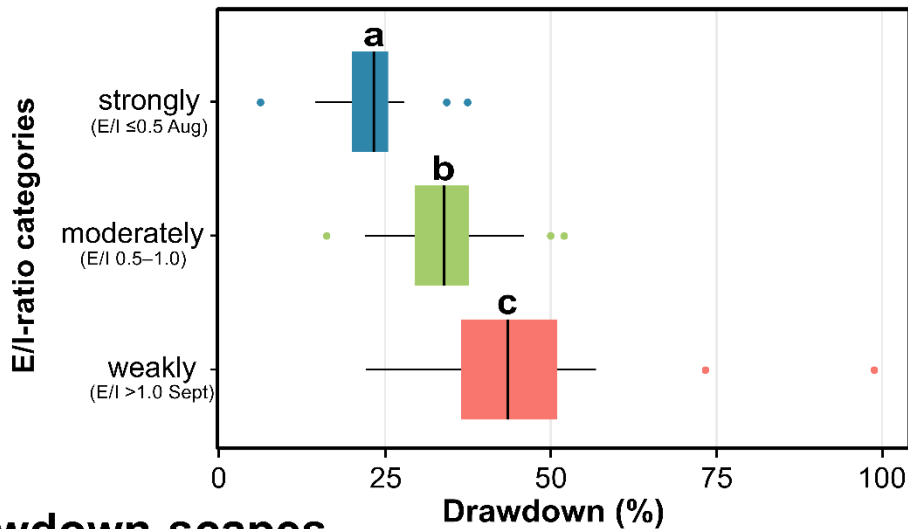
>1.0 in September, when net evaporative water loss is greatest. The remaining waterbodies were placed into the moderately connected category.

4.4.2 Water-level variation

During the arid summer of 2022, water levels declined in all waterbodies sampled across SKNA (Figure A2.1, Table A2.5), and the magnitude of drawdown differs among the three E/I ratio-defined groundwater connectivity categories (Kruskal Wallis test, $\chi^2 = 29.3$, d.f. = 2, $p = 4.3 \times 10^{-7}$; Figure 4.4a). This test is based on measurements recorded in the ponds, because depth loggers were removed from the nesting wetlands before September. The interquartile ranges for the drawdown of ponds in each category possess little to no overlap (Figure 4.4a), and a Dunn's post-hoc test confirms that weakly connected waterbodies drew down significantly more than strongly ($z = 5.4$, $p_{\text{adj}} = 2.1 \times 10^{-7}$) and moderately ($z = -2.5$, $p_{\text{adj}} = 1.3 \times 10^{-2}$) connected waterbodies, and moderately connected waterbodies drew down significantly more than strongly connected waterbodies ($z = 3.4$, $p_{\text{adj}} = 1.4 \times 10^{-3}$). Notably, the drawdown is substantial (medians are 43.8% for weakly connected waterbodies, 33.9% for moderately connected, 23.3% for strongly connected).

Similar to the spatial distribution of E/I ratios, water-level drawdown is not uniform across the landscape (i.e., 'drawdown-scapes' of Figure 4.4b). Patterns of drawdown across SKNA are similar for June-August and June-September, with larger magnitudes observed over the longer period. The greatest drawdown occurs in the central region of SKNA, which includes areas south of Klewi River and north of Sass River, as well as more localized regions in the northwest and southeast of SKNA. Areas where drawdown exceed 40% in September align with areas where E/I ratios exceed 1.0. Waterbodies with the smallest drawdown (e.g., <20% in August, <35% in September) occur along the Klewi River and adjacent areas to the north, and an expansive region along the Sass River, where E/I ratios are <0.5 in August. Waterbodies in the northern Klewi area typically drew down more than those in the Sass area (SK63 is an exception). These findings are consistent with observations by McNaughton (1991) that water-level fluctuations are greater in Klewi than Sass areas, which was attributed to differences in water storage capacity of the overburden on variation in elevation

a) Drawdown magnitude



b) Drawdown-scapes

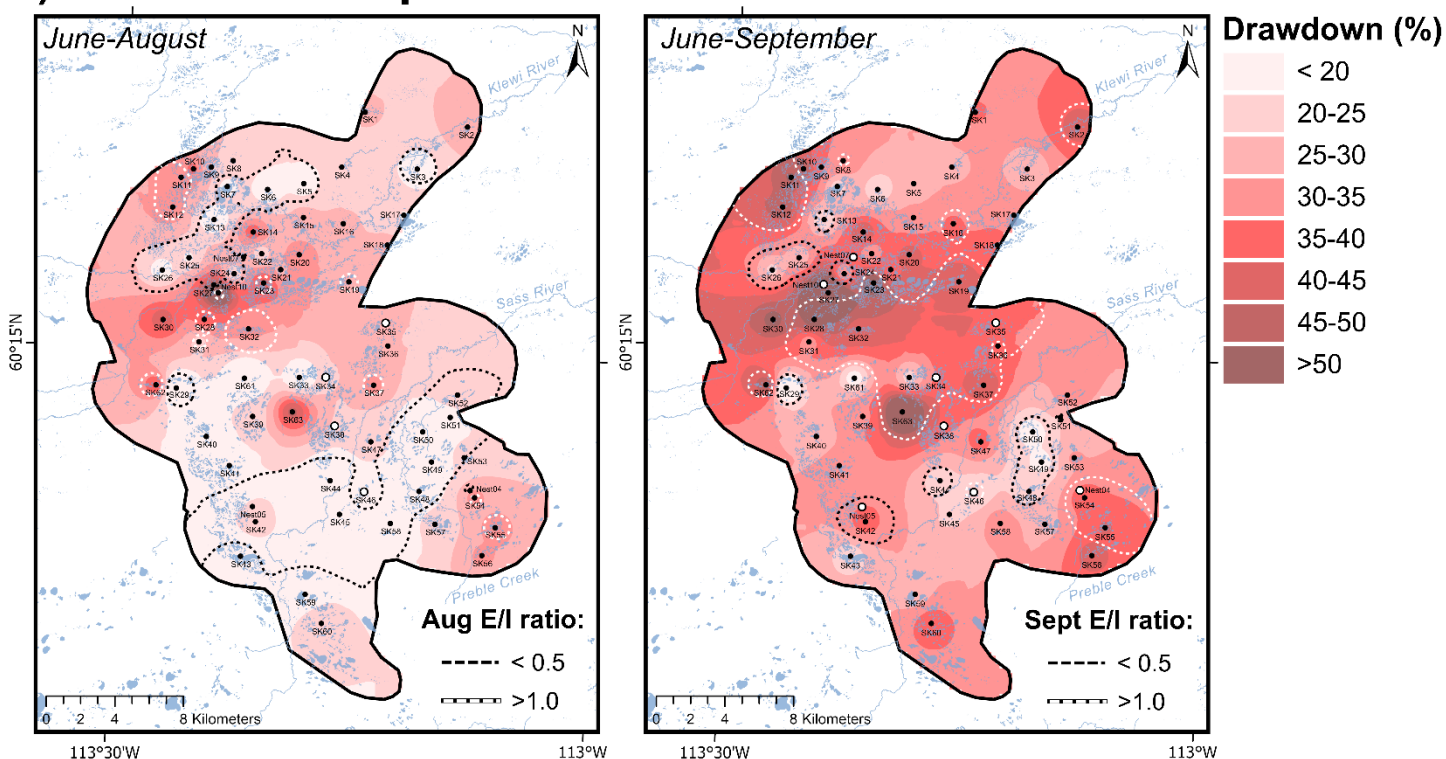


Figure 4.4. **a)** Boxplots showing the distribution of water-level drawdown between June 1 and the sampling campaign in September 2022 (as a percent of the initial depth) for ponds in the three E/I-ratio defined groundwater connectivity categories. Letters placed above each boxplot identify significant differences in median values based on a Dunns post-hoc test. **b)** Drawdown-scapes showing spatial distribution of pond drawdown (%) during intervals: June–August (ponds and nesting wetlands) and June–September (ponds only) across the Sass-Klewi Nesting Area. Black contours represent E/I ratios < 0.5 and white contours represent E/I ratios > 1.0 in respective months (August, September). Sites that did not contribute to the interpolated layer are identified by white circles. Drawdown (%) values are reported in Table A2.5.

of the shallow water table. If so, the groundwater has short residence time, and the water table declines during the open-water season when there is insufficient local recharge by rainfall. Analysis of water chemistry was pursued to further determine if spatial differences in E/I ratios and drawdown among waterbodies are related to the degree of connectivity to groundwater because the chemistry of groundwater typically differs from that of precipitation (Déri-Takács et al., 2022; Welham et al., 2024).

4.4.3 Water chemistry

Based on water chemistry measurements in August and September 2022, the ponds and nesting wetlands in SKNA are alkaline (average pH = 8.1), moderately brackish (average conductivity = 2331 $\mu\text{S}/\text{cm}$) and span from oligotrophic to eutrophic based on TP concentration (5-70 $\mu\text{g}/\text{L}$) (Figure 4.5; Table A2.6). Concentrations of TN range from 452-7500 $\mu\text{g}/\text{L}$ and concentrations of DOC and DIC range from 9-85 mg/L and 10-50 mg/L, respectively. SO_4^{2-} (average = 1229 mg/L) and Ca^{2+} (402 mg/L) are the ions with the highest average concentration, followed by HCO_3^- (155 mg/L), Cl^- (119 mg/L), Mg^{2+} (95 mg/L), Na^+ (94 mg/L) and K^+ (5 mg/L). DSi concentrations range from 1-19 mg/L.

In May, when E/I ratios are low in all waterbodies (≤ 0.3), conductivity and concentrations of most water chemistry variables are relatively low and the is small (illustrated as horizontal grey bars in Figure 4.5). Conductivity and concentrations of major nutrients (TP, TN, DOC) and conservative ions (Cl^- , Mg^{2+} , Na^+ , K^+) are markedly higher in August and September than in May in the moderately and weakly connected waterbodies, but they remain low (i.e., within or near the range of values in May) in the strongly connected waterbodies. The distribution of values for these variables does not change markedly between August and September in any of the three categories. Waterbodies with moderate groundwater connectivity have values in August and September that are intermediate between those in the strongly and weakly connected categories. For example, strongly connected waterbodies are classified as oligotrophic based on TP concentration, whereas moderately and weakly connected waterbodies are mesotrophic and eutrophic, respectively. pH and concentrations of DIC, HCO_3^- , SO_4^{2-} , Ca^{2+} and DSi, however, do not follow this pattern. Between May and August, pH declines in the strongly connected waterbodies but not in the other categories.

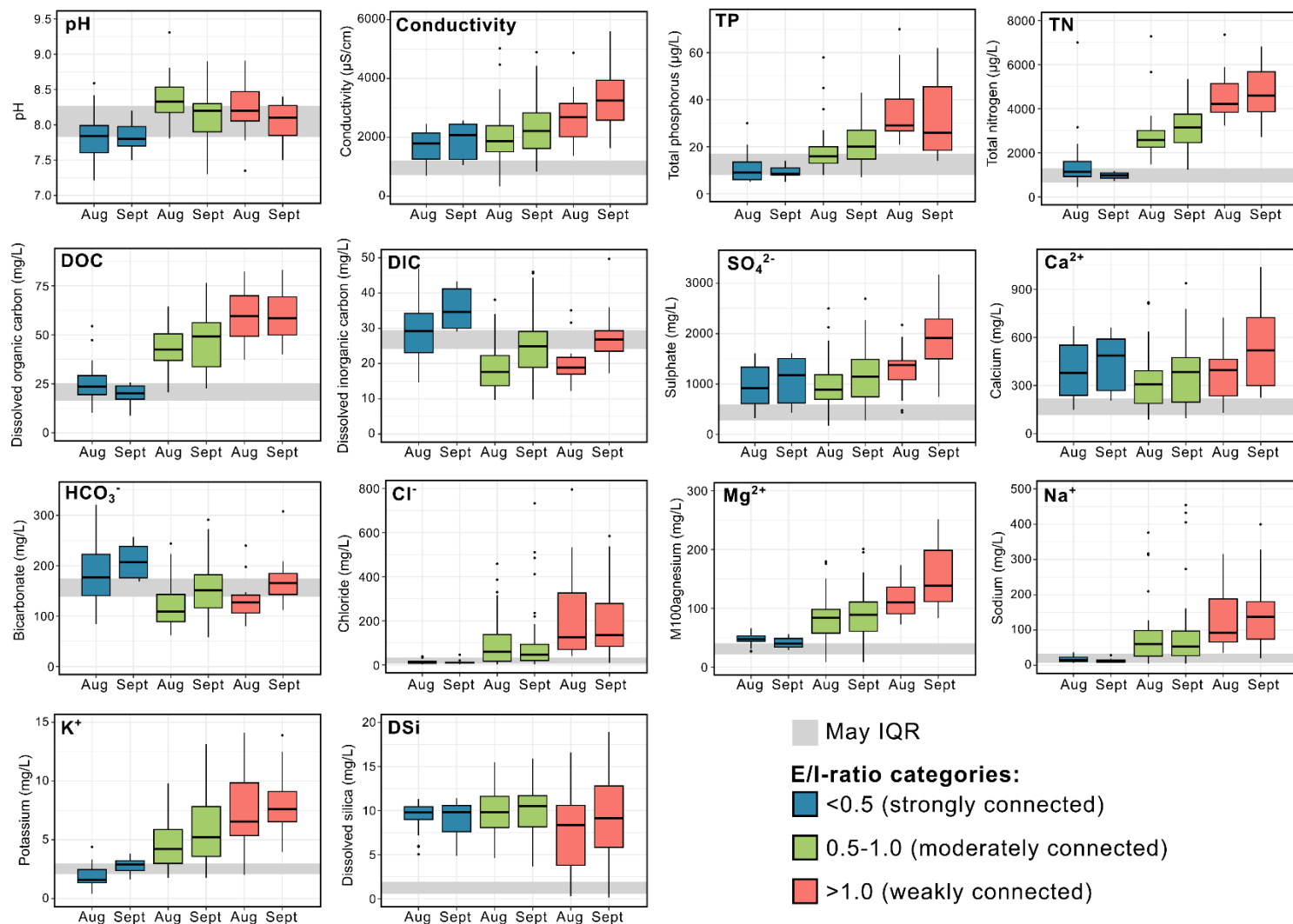


Figure 4.5. Boxplots showing the distributions of values for 14 water chemistry variables in the waterbodies sampled in the Sass-Klewi Nesting Area in 2022. Boxes are coloured according to E/I-ratio defined groundwater connectivity categories. Water chemistry measurements are reported in Table A2.6.

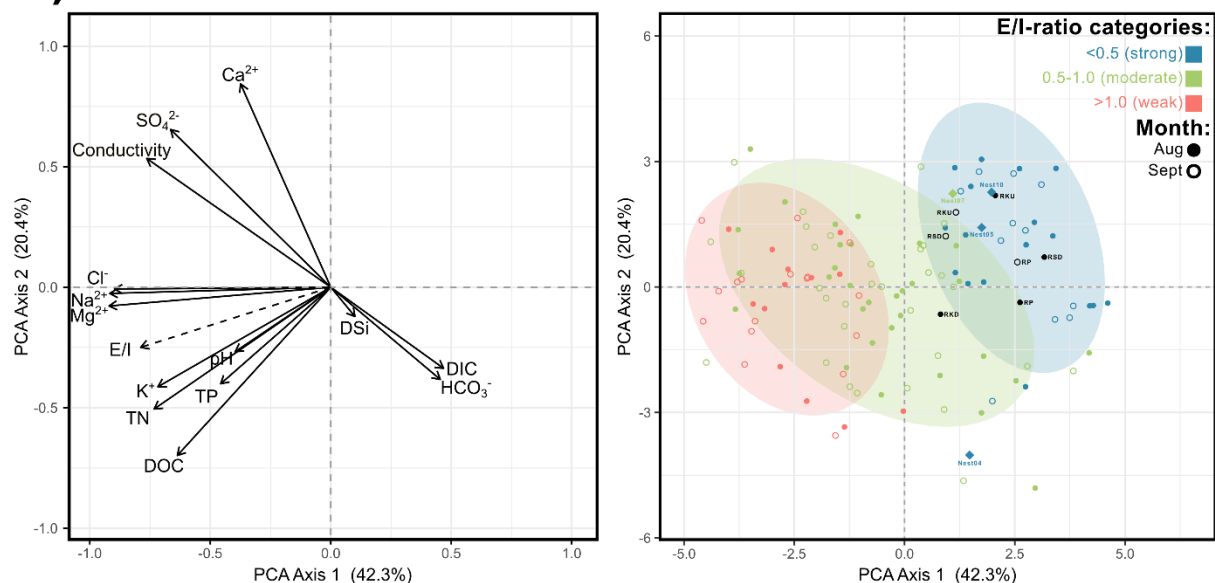
Concentrations of DIC and HCO_3^- decline between May and August in the moderately and weakly connected waterbodies, but not in the strongly connected waterbodies, and they increase between August and September in all three categories. Concentrations of SO_4^{2-} , Ca^{2+} and DSi increase between May and August in all three categories, and the distributions overlap substantially among all three categories.

A PCA of the water chemistry variables, with E/I ratio included passively, reveals how water chemistry in August and September 2022 varies along the gradient of groundwater connectivity. Axes 1 and 2 capture 42.3% and 20.4% of the variation in the water chemistry

variables, respectively (Figure 4.6a; Table A2.8). Axis 1 is most strongly associated with variation in conductivity and concentrations of conservative ions (Na^+ , Mg^{2+} , Cl^- , K^+) and total nitrogen (TN), whose vectors point leftward. Axis 2 is most strongly associated with concentration of Ca^{2+} , whose vector points upward. Several variables are correlated with both axes 1 and 2, including pH and concentrations of TP, DOC, HCO_3^- , DIC, SO_4^{2-} and DSi. Strong positive correlation of the E/I ratio vector with axis 1 reveals the main direction of variation in water chemistry is associated with the gradient of groundwater connectivity. For example, waterbodies with relatively high E/I ratios, which are weakly connected to groundwater, are positioned to the left along axis 1 due to relatively high pH and conductivity, and high concentrations of conservative ions, major nutrients and DOC. In contrast, waterbodies with relatively low E/I ratios, which are strongly connected to groundwater, are positioned to the right along axis 1 due to lower values of these variables. The PCA also reveals that waterbodies with relatively low E/I ratios have relatively high concentrations of DIC and HCO_3^- , chemistry species which are known to be in high concentration in the groundwater (Deri-Tackas et al., 2022). The 90% confidence ellipsoids for the sample scores of the strongly and weakly connected waterbodies do not overlap, suggesting waterbodies in these two categories possess markedly different water chemistry. Sample scores and the 90% confidence ellipsoid for the moderately connected waterbodies are positioned between ellipsoids of the other two categories, and they overlap most extensively with the weakly connected waterbodies.

The separation of sample scores along PCA axis 1 for waterbodies in the three E/I-ratio defined groundwater connectivity categories enables use of the axis 1 sample scores as a synthetic univariate metric to explore spatial variation of multiple water chemistry variables across SKNA associated with differences in groundwater connectivity (Table A2.9). Geospatial interpolation of these sample scores as ‘limnoscapes’ reveals that waterbodies with relatively high axis 1 sample scores, depicted with blue hues, are located mainly in lower-lying areas of the landscape adjacent to the rivers (Figure 4.6b). These areas coincide with the contours in the isoscapes (Figure 4.3c) where E/I ratios are below 0.5 in August (black-dashed lines in Figure 4.6b) because of strong connectivity to isotopically depleted

a) PCA



b) Limnoscapes

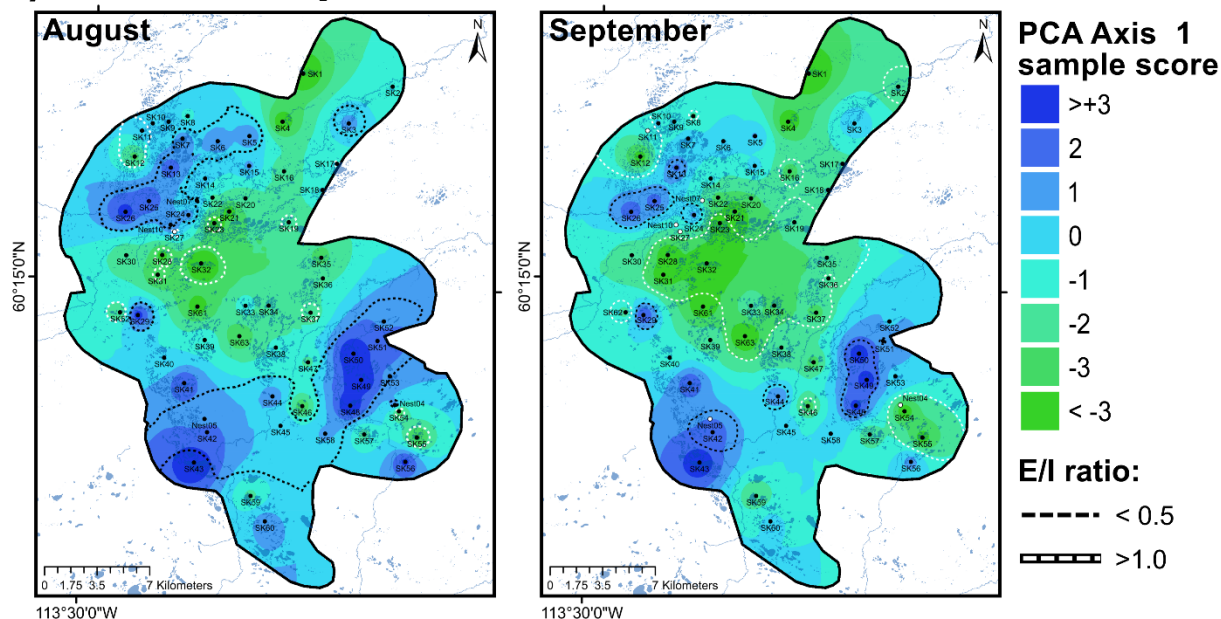


Figure 4.6. a) Principal components analysis (PCA) ordination plots illustrating associations among 14 water chemistry variables (with E/I ratio included passively) (left panel) and the distribution of sample scores for the 63 ponds (circles) and four nesting wetlands (diamonds) and river sites (light blue) sampled in August (closed symbols) and September (open symbols) 2022 in the Sass-Klewi Nesting Area along axes 1 and 2 (62.7% total variation) (right panel) (see also Table A2.8). Sample scores are colour-coded according to E/I-ratio defined groundwater connectivity categories. Confidence ellipsoids (90%) are shown for each category. Correlation coefficients for each variable are reported in Table A2.7. **b)** Limnoscapes depicting spatial visualization of the PCA axis 1 scores. Contour lines representing E/I ratios <0.5 (black dashed line) and >1.0 (white dashed line) in August and September 2022 are superimposed.

groundwater. Relatively low concentrations of major nutrients, DOC and conservative ions, and relatively high concentrations of DIC and HCO_3^- in these waterbodies is consistent with strong influence of groundwater discharge (McNaughton 1991; Déri-Takács et al. 2022). In contrast, waterbodies with relatively low axis 1 sample scores, depicted as green hues in the ‘limnoscapes’, are in slightly more elevated portions of the landscape where E/I ratios typically exceed 1.0 in September (areas within the white dashed lines in Figure 4.6b) due to strong influence of evaporation on waterbodies that are weakly connected to (or disconnected from) groundwater. These waterbodies have relatively high concentrations of major nutrients, DOC and conservative ions, and relatively low concentrations of DIC and HCO_3^- , which is consistent with strong influence of evaporation in the absence of substantial input of groundwater and precipitation. Between August and September, the areas with blue hues decrease and the areas with green hues increase, revealing that concentrations of major nutrients, DOC and most ions increased as inputs from groundwater (via decline of the water table) and precipitation waned, and evaporation increased.

4.5 Discussion

4.5.1 Hydrological processes influencing waterbodies in Sass Klewi Nesting Area

Above-average snow accumulation before a late freshet in 2022, followed by below-average rainfall during the summer months, provided serendipitous opportunity to evaluate the influence of inflow from snowmelt and groundwater discharge, and water loss via evaporation and vertical seepage based on variation of isotope-inferred water balance, depth and water chemistry of shallow waterbodies across a core central area of the WCSR. Understanding the hydrological processes regulating waterbodies across SKNA required integrating all three of these measurements. For instance, hourly water-depth measurements revealed substantial drawdown during the open-water season in nearly all waterbodies, ranging from 6% to 99% of the initial depth on June 1st. However, these measurements alone do not reveal the hydrological processes causing the drawdown. If water depth had been the only measurement, one might conclude that drawdown was primarily driven by net evaporation, given the minimal rainfall during the warm, arid summer of 2022, and that the variability among waterbodies is attributed to basin- and catchment-

specific factors (e.g., Coops et al. 2003; Carroll et al. 2011). Integration of spatiotemporal variation in water isotope composition, however, provided crucial insights, revealing substantial drawdown even in waterbodies with consistently low water isotope composition in August and September, which generated E/I ratios indicative of strongly positive water balances ($E/I < 0.3$). If only water isotope composition had been measured, the strongly positive water balances would have logically suggested that water levels either rose or remained relatively stable in these waterbodies during the open-water season when, instead, they declined substantially. Recognition that drawdown at these waterbodies must involve a process that does not alter the isotope composition led us to identify vertical seepage as the primary mechanism of water loss, because they typically lack outflow channels. The arid summer weather likely created substantial moisture deficit, causing the water table to decline sufficiently to promote vertical seepage and subsequent drawdown of water levels – a phenomenon previously observed in the region during a dry interval (McNaughton, 1991).

A key discovery is that many waterbodies underwent relatively less drawdown because they remained connected to groundwater, as reflected in the sustained low water isotope compositions throughout the open-water period. Water chemistry measurements provide valuable supporting evidence for sustained groundwater connectivity to these waterbodies. This includes maintenance of relatively high concentrations of DIC and HCO_3^- , and low pH and low concentrations of major nutrients (TP, TN, DOC) and conservative ions (Cl^- , Mg^{2+} , Na^+ , K^+) during the summer and fall, similar to groundwater chemistry in this area (Weyer et al., 1978; Weyer, 1983; McNaughton, 1991; Déri-Takács et al., 2022). In contrast, at waterbodies where isotope-inferred E/I ratios exceed 1.0 (i.e., water balance was negative), lower concentrations of DIC and HCO_3^- , along with higher pH and increasing concentrations of major nutrients and conservative ions, provide evidence of weaker groundwater connectivity and stronger influence of evaporation. Geospatial visualization of the three types of measurements across SKNA reveals that waterbodies with sustained groundwater discharge, and where drawdown was caused primarily by vertical seepage, are located in lower elevation portions of the landscape proximal to the rivers. In comparison, waterbodies

with weak (or absent) groundwater discharge, where drawdown is more substantial and driven mainly by evaporation, are located in slightly more elevated areas farther from the rivers.

Given the complex interplay of evaporation and vertical seepage driving widespread, yet variable drawdown of waterbodies across SKNA between spring and fall 2022, we use photographic evidence and measurements obtained at a representative pond from each of the three groundwater-connectivity categories and a schematic diagram to further illustrate and contrast their roles (Figure 4.7). At SK27, a pond in the weakly connected category, water isotope composition increased substantially between spring and summer and the isotope-inferred E/I ratio rose from 0.32 (indicative of strongly positive water balance) to 1.40 (strongly negative) (Figure 4.7a). The water level declined 69% between spring and summer, and the basin desiccated between summer and fall. Unfortunately, the pond was too shallow in summer and fall to obtain sufficient volume of sediment-free water for water chemistry analysis. Nonetheless, the measurements of water isotope composition clearly capture a dominant influence of evaporative water loss. This can only occur if groundwater discharge and water loss via vertical seepage are minimal for all or most of the open-water season. The evidence, thus, points to evaporation as the main mechanism accounting for desiccation of SK27. In the schematic (Figure 4.7b), the relative size of the arrows is used to illustrate that evaporation is the dominant hydrological process affecting drawdown of SK27 and other waterbodies with weak connectivity to groundwater, and the influence of vertical seepage is relatively minor. An outcome is that drawdown of weakly connected waterbodies (median = 43.8%) is greater than drawdown of moderately (median = 33.9%) and strongly (median = 23.3%) connected waterbodies.

At the other end of the hydrological gradient is SK44, a pond in the strongly connected category (Figure 4.7a). Consistently low water isotope composition and strongly positive isotope-inferred water balance during the open-water season ($E/I = 0.09-0.30$), along with consistently high concentrations of DIC and HCO_3^- and low pH and low concentrations of major nutrients and conservative ions (as captured by PCA axis 1 sample scores of +1.7 in summer and fall) indicate sustained, strong influence of groundwater discharge and minimal

influence of evaporation. This resulted in less drawdown at SK44 (22%). In the schematic (Figure 4.7b), the relative size of the arrows indicates that vertical seepage is the dominant hydrological process affecting drawdown at strongly connected waterbodies, and evaporation exerts minimal influence because of sustained groundwater discharge. Consequently, water depth is controlled mainly by the magnitude of decline in the water table, which results in substantially less drawdown via vertical seepage than the drawdown that occurs in weakly connected waterbodies via evaporation.

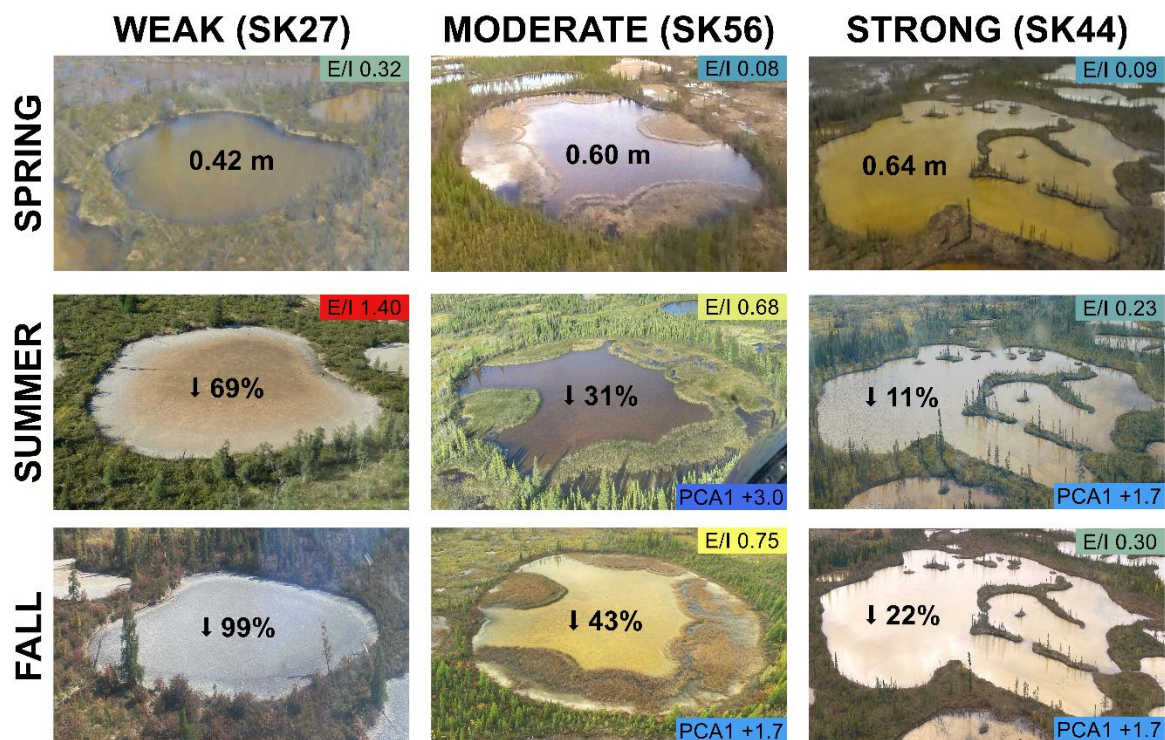
At SK56, a pond in the moderately connected category, results reveal that both evaporation and vertical seepage contributed to drawdown of 43% between spring and fall (Figure 4.7a), which is an intermediate value compared to drawdown at SK27 (99%) and SK44 (22%). This conclusion is based on the marked rise of isotope-inferred E/I ratios from 0.09 in spring to 0.68 in summer and 0.75 in fall, which identifies substantial influence of evaporation on pond water, and the decrease in water chemistry PCA axis 1 sample scores from 3.0 in summer to 1.7 in fall, which reveals that influence of groundwater discharge waned during the late summer. In the schematic (Figure 4.7b), the relative size of the arrows illustrates that evaporation and vertical seepage have similar influence on drawdown of moderately connected waterbodies.

4.5.2 A ‘Vulnerability Index’ for tracking hydrological responses of waterbodies to climate variation

Following an approach presented by Lintern et al. (2016), we developed a multi-metric index that incorporates variability across all three metrics to assess the hydrological vulnerability of the remote, shallow waterbodies in SKNA to climate variation. Here, we consider hydrological vulnerability to reflect the likelihood of desiccation of waterbodies by evaporation and/or vertical seepage, which is a key stressor for whooping crane (CWS–USFWS 2007). To generate the Vulnerability Index, isotope-inferred E/I ratios in August, magnitude of water-level change between June 1st and the August sampling episode, and water chemistry PCA axis 1 scores in August were each scaled from 0-1 using the formula:

$$x_{scaled} = \frac{x - x_{min}}{x_{max} - x_{min}}$$

a) Representative ponds



b) Schematic

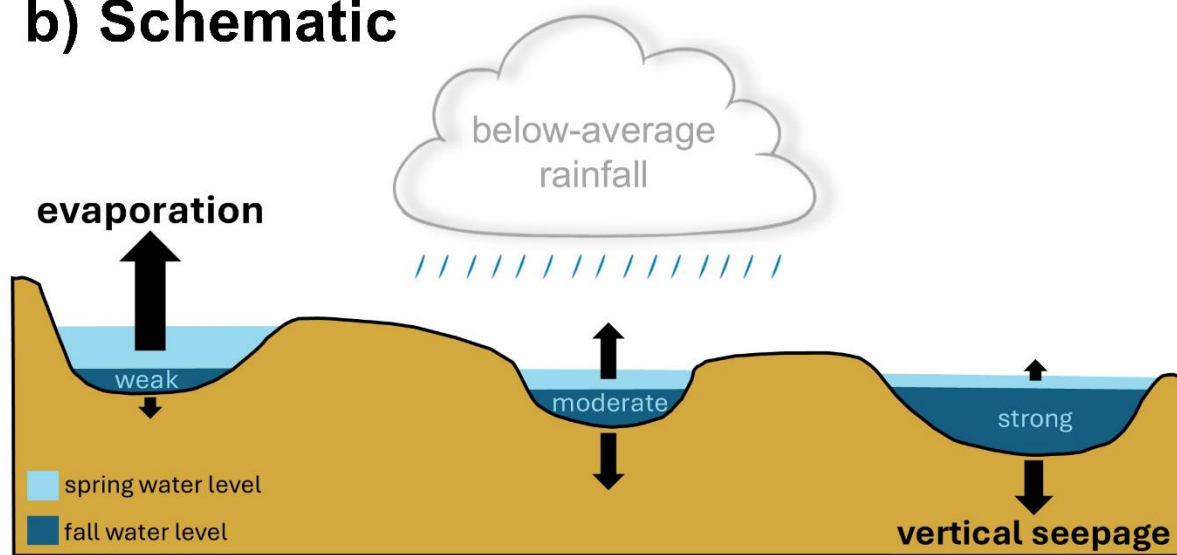


Figure 4.7. a) Images of three ponds in Sass-Klewi Nesting Area in spring (May), summer (August) and fall (September) 2022 that are representative of the three E/I-ratio defined groundwater connectivity categories: weak (SK27), moderate (SK56) and strong (SK44). Initial pond depths are labelled on the ponds in the spring, whereas the magnitude of drawdown (%), relative to the initial depth, is labelled on the ponds in the summer and fall. Isotope-inferred E/I ratios and water chemistry PCA axis 1 sample scores are annotated in the corners of each image, and they are colored according to the palettes presented in Figures 4.3 and 4.6. **b)** Schematic diagram illustrating pathways of water loss that influence drawdown in response to below-average rainfall. Black arrows illustrate the relative magnitude of evaporation and vertical seepage on ponds, which led to varying degrees of pond drawdown in 2022.

where x is the value of the metric in a waterbody and x_{max} and x_{min} represent the maximum and minimum values for the dataset, respectively. The Vulnerability Index was then determined as the average value of the three scaled metrics (Table A2.9). Thus, the higher the index value, the closer the waterbody is to desiccation in August. One of the three metrics was not available at six ponds (SK11, SK27, SK34, SK35, SK38, SK46) and the drawdown magnitude at the four nesting wetlands was not representative due to deployment of the data loggers in shallow water (<25 cm) along their margin, so their Vulnerability Index scores were determined as the average value of the two available metrics. The three key metrics selected for the ‘Vulnerability Index’ effectively summarize each measurement type and exhibit strong linear relations with the Vulnerability Index (Figure A2.2). Thus, the index provides a more robust assessment of the hydrological vulnerability of waterbodies to desiccation than could be achieved by any one variable alone. We developed the Vulnerability Index using measurements in June to August because this interval is more ecologically relevant for whooping crane, as they actively forage with their newly hatched chicks during this interval before they migrate southward in mid- to late September (Erickson and Derrickson, 1981; CWS–USFWS, 2007). Focusing on the June–August period also allowed for the inclusion of measurements from the four nesting wetlands, which were not sampled in September.

Geospatial visualization of the Vulnerability Index scores (Figure 4.8) reveals spatial patterns that align with those observed in the isoscapes (Figure 4.3c), drawdown-scapes (Figure 4.4b) and limnoscapes (Figure 4.6b). Areas with low Vulnerability Index scores correspond with areas where E/I is <0.5, drawdown magnitude is relatively small and water chemistry PCA axis 1 scores are high, and vice versa. Nonetheless, geospatial visualization of the Vulnerability Index provides an effective depiction of vulnerability to desiccation across the landscape. The results reveal waterbodies are most vulnerable to desiccation in the central region of SKNA, which includes areas south of Klewi River and north of Sass River, as well as more localized regions in the northwest and southeast of SKNA, and they are least vulnerable in areas adjacent to the Sass and Klewi rivers and downstream portion of Preble Creek near its confluence with Sass River.

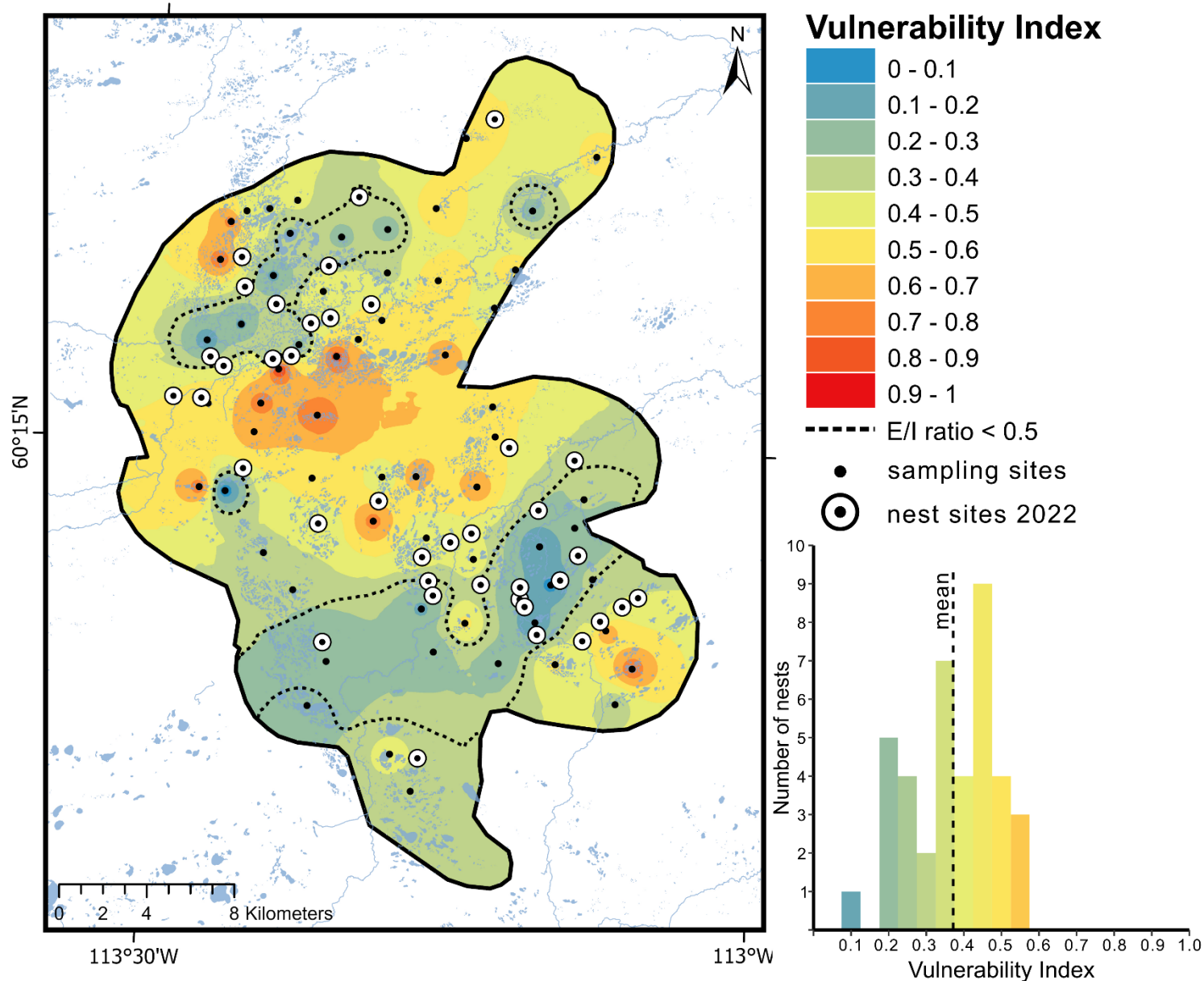


Figure 4.8. Map showing geospatial visualization of the Vulnerability Index scores across Sass-Klewi Nesting Area (Table A2.9), with sampling sites (small black circles) and locations of the 39 active nests in 2022 (large, white-filled circles around small black centers) superimposed. Contours identify areas where E/I ratios are below 0.5 (black-dashed lines) based on August isotope compositions of waterbodies. All nest locations were compiled by John Conkin, Mark Bidwell and Lori Parker. A histogram on the bottom right identifies the frequency distribution of the interpolated Vulnerability Index scores in August 2022 at the 39 active nests, and the vertical black dashed line represents the mean (0.37).

We explored whether there is an association between interpolated Vulnerability Index scores and the location of active whooping crane nests in SKNA in 2022. Of the 39 nests, 21 (54%) are located in areas where interpolated Vulnerability Index scores are below 0.4, which is where moderate and strong groundwater connectivity offsets drawdown of water-levels by

evaporation (Figure 4.8). Computed Vulnerability Index values for the four nesting wetlands ranged from 0.20 to 0.36 and the remaining 35 nests are in areas where interpolated values are <0.6 , suggesting that whooping cranes may select nest sites at wetlands that are less vulnerable to desiccation. However, waterbodies in areas with high Vulnerability Index scores may be unsuitable for nesting due to other ecological factors, such as insufficient access to nesting materials and preferred food or greater risk of predation. Although further testing of the association between interpolated Vulnerability Index scores and nest locations is needed, continued measurements and derivation of the Vulnerability Index over many years will provide opportunity to assess variation over space and time in freshwater availability as climate change continues to unfold.

4.6 Conclusions

Above-average snowpack, delayed freshet and below-average rainfall in 2022 generated conditions conducive to elucidating how complex interactions among hydrological processes influence water levels in critical wetland habitat for endangered whooping crane at Ramsar Site 240 (Whooping Crane Summer Range). Water levels declined across the landscape during the open-water season in 2022, and measurements of water isotope composition and chemistry revealed that spatial variation of the drawdown is due largely to differences in groundwater connectivity across the karst terrane. Drawdown was greatest in waterbodies with weak connectivity to groundwater due mainly to water loss by evaporation. In contrast, drawdown was smallest at waterbodies with strong connectivity to groundwater because sustained discharge of groundwater offsets evaporation. In these waterbodies, drawdown is due to water loss via vertical seepage as the water table declined. Drawdown at waterbodies with moderate connectivity to groundwater falls between that of waterbodies in the other two categories as a consequence of intermediate influence of water loss by evaporation and vertical seepage. Drawdown and desiccation of waterbodies have negative consequences for whooping crane productivity due to loss of aquatic food sources and increased risk of encounters with predators (Kuyt and Barrie, 1992; CWS–USFWS, 2007). Thus, the findings of this study indicate that wetlands located in portions of the landscape where there is weak connectivity to groundwater are most vulnerable to desiccation during

dry climatic intervals, whereas wetlands with strong connectivity to groundwater are least vulnerable to desiccation. This is supported by geospatial interpolation of a multi-metric 'Vulnerability Index', based on isotope-inferred evaporation-to-inflow ratios, water-level drawdown and water chemistry ordination sample scores, which identified that whooping crane nested in 2022 where waterbodies possess low index scores associated with minimal drawdown because of strong connectivity to groundwater.

Similar to other water-rich landscapes in cold regions, waterbodies in WCSR will likely respond rapidly to future climatic changes due to their small surface area and volume (Smol and Douglas, 2007; Williamson, 2009; Smol, 2016; Woolway et al., 2022). Downscaled climate models consistently project accelerated warming, altered precipitation patterns, and more frequent extreme events (Easterling et al., 2000; Bonsal et al., 2001; Prowse et al., 2009; Feng et al., 2014; Mekis and Vincent, 2011; Nelson et al., 2014; Sauchyn et al., 2015; Bush and Lemmen, 2019). A few long-term studies in cold regions of Canada have demonstrated rising influence of rainfall on water balance of shallow lakes and rivers (Wolfe et al., 2011; MacDonald et al., 2012, 2017, 2021; Beel et al., 2021; Mohammed et al., 2023), findings that are consistent with expected trends based on regional climate projections (Bintanja and Andry, 2017; Mekis and Vincent, 2011; Bintanja et al., 2020; Dou et al., 2021, 2022; Boisvert et al., 2023). However, hot, arid summers in 2022 and 2023 fostered extensive forest fires near our study area in 2023 (Jain et al., 2024; Kirchmeier-Young et al., 2024; Wang et al., 2025). Such meteorological variability highlights the need for a more extensive understanding of the hydrological responses of shallow waterbodies that form critical breeding habitat for the endangered Aransas-Wood Buffalo Population of whooping crane. Ongoing measurements of water isotope composition, depth and chemistry in SKNA and throughout the entire whooping crane breeding range is therefore needed to capture a broader range of meteorological conditions and the hydrological responses of waterbodies, which may improve our ability to predict the vulnerability of wetland breeding habitat to possible climates of the future and form a basis for long-term aquatic ecosystem monitoring.

4.7 References

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Chapter 5 | Synthesis of key findings

Aquatic ecosystem monitoring is essential for rapid detection and quantification of environmental changes, informing conservation strategies, and supporting evidence-based stewardship decisions. A wealth of research has defined principles and best practices including how to design and execute sustainable, effective monitoring programs across diverse ecosystems (Gozlan et al., 2017; Hobbie et al., 2003; Hunting et al., 2017; Lee et al., 2011; Lindenmayer et al., 2022a; Lovett et al., 2007; Sparrow et al., 2020). Key contributions emphasize that effective long-term monitoring programs must be developed from a comprehensive understanding of the key processes that regulate an ecosystem – knowledge that is obtained through extensive primary data collection and research (Lindenmayer and Likens, 2010; 2018; Lindenmayer et al., 2008; 2012; 2015; 2020; 2022b). Such an understanding forms the foundation for clearly defined aquatic ecosystem monitoring program goals and objectives, as well as selecting appropriate, standardized methodologies capable of detecting change across a range of scales of space and time.

Developing a comprehensive understanding of the key processes influencing aquatic ecosystem structure and function can be challenging due to the inherent complexity and variability at which the processes operate over space and time (Strobl and Robillard, 2008; Lindenmayer and Likens, 2018). Selecting the number and locations of sampling sites, identifying the most informative and cost-effective variables to measure, and determining the frequency at which measurements should be obtained, for example, require considerable thought, planning and experience to determine the key processes and causes affecting an ecosystem and to detect when they become altered (Rose and Smith, 1992; Lovett et al., 2007; Lindenmayer et al., 2008; Sparrow et al., 2020). Additional challenges typically include limited access to or lack of incorporation of baseline and historical data (Kilgour et al., 2007; Allen et al., 2025).

Overcoming the above obstacles requires that rigorous scientific research is undertaken to generate the knowledge needed to inform the design of a long-term aquatic ecosystem monitoring program, and to continually refine monitoring approaches. As illustrated in Figure 5.1, Lindenmayer and co-authors cited above advocate for a “Research-to-Inform

Monitoring Continuum”, where key phases of scientific research generate the knowledge needed to inform the design and implementation of long-term aquatic ecosystem monitoring. Driven by the need to address concerns about ecosystem degradation, the Continuum begins with *Conceptualization*, when key stressors and processes are identified and initial ideas for informative measurements are discussed. *Collaborative Planning* follows, bringing together researchers, stakeholders, rightsholders and natural resource stewards to define objectives and agree on an effective study design that will generate primary data on key processes influencing aquatic ecosystem structure and function. *Funding Acquisition* typically occurs during the *Collaborative Planning* phase to secure the necessary resources for the research. The *Data Collection* phase involves field-based sampling from which the measurements of key variables are obtained. The *Data Analysis* phase may co-occur with *Data Collection*, especially when data collection occurs over several years, and it includes when important features of the data are recognized and conclusions are drawn. During the *Communication of Results* phase, researchers share the findings with the stakeholders, rightsholders, natural resource stewards and general public in accessible formats as well as with the scientific community through peer-reviewed manuscripts. Finally, during the *Synthesis and Recommendations* phase, researchers summarize key findings from the primary research in ways that inform the design of a long-term aquatic ecosystem monitoring program. This phase is intended to facilitate a smooth transition to stakeholder- or agency-led long-term monitoring, as depicted in Figure 5.1.

Research-to-Inform Monitoring Continuum

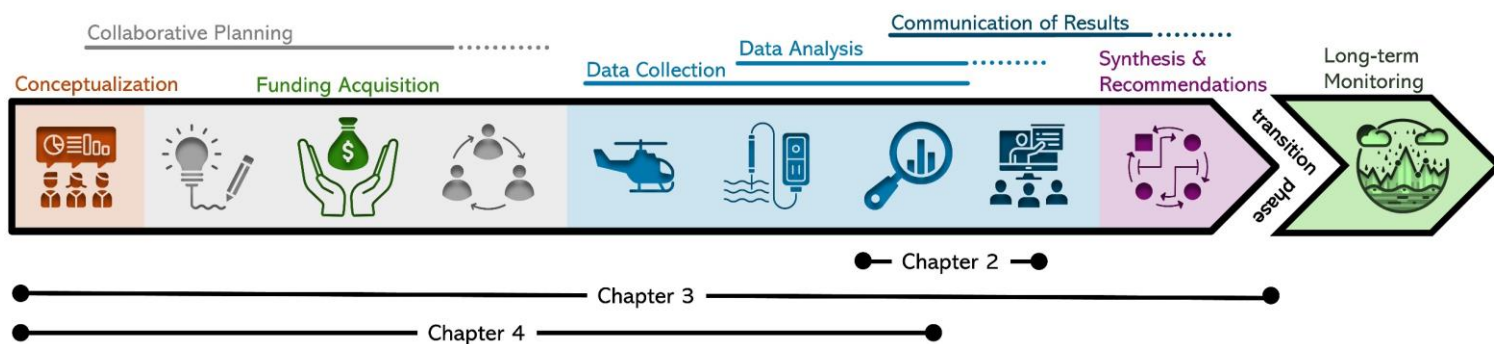


Figure 5.1. Illustration of a proposed ‘Research-to-Inform Monitoring Continuum’ that captures key phases of the research process required to inform and guide implementation of effective and sustainable environmental monitoring programs. Research contained in this dissertation (Chapters 2-4), collectively, spans the Continuum from *Conceptualization* to *Synthesis and Recommendations*.

Effective and sustainable long-term monitoring of shallow water ecosystems is urgently needed for two Ramsar Wetlands of International Importance (the Peace-Athabasca Delta and the Whooping Crane Summer Range) located within Wood Buffalo National Park (WBNP) – Canada’s largest national park – where concerns have long been raised about the potential for ecosystem degradation by climate change and industrial activities. In 2014, the concerns prompted the Mikisew Cree First Nation to petition for the park’s World Heritage Status to be downgraded to ‘In Danger’ (MCFN, 2014), which would be a first for Canada’s 22 World Heritage Sites. In response, national and international organizations and governing bodies conducted investigations to identify the environmental threats and directions of change to the park’s Outstanding Universal Values, resulting in a series of recommendations from UNESCO’s Reactive Monitoring Mission (WHC/IUCN, 2017) and the development of a federal Action Plan (WBNP, 2019). At the time this multi-jurisdictional process was unfolding, my research group had already initiated the *Conceptualization* phase, and ongoing *Collaborative Planning* and *Funding Acquisition* (Figure 5.1) generated a program of research that began in 2015 to inform monitoring of shallow lakes at the Peace-Athabasca Delta (PAD), which built forward from paleolimnological and hydroecological research conducted during the previous 15 years. Interactions with stakeholders during that prior research, and the evidence it generated, had long fostered recognition of an urgent need for aquatic ecosystem monitoring in the PAD to track environmental changes and determine the causes (see Wolfe et al., 2012).

Conducting informative research to guide monitoring frameworks for shallow lakes in the Peace-Athabasca Delta and shallow ponds and wetlands in the Whooping Crane Summer Range has been the overarching goal of my thesis. In these remote northern landscapes, hydrological processes exert dominant influence on physical, chemical and biological characteristics of the shallow waterbodies and they affect the transport and deposition of substances of concern. An emphasis of my PhD research, thus, has been on adopting and applying a suite of informative methods (continuous water-depth measurements, seasonal measurements of water isotope composition and water chemistry) to identify the influential hydrological processes and characterize how they operate over space and time. Collectively,

the research communicated in Chapter 2-4 spans portions to the entire Research-to-Inform Monitoring Continuum, as illustrated at the bottom of Figure 5.1, from *Conceptualization* to *Synthesis and Recommendations*. Below, I situate each of these chapters within this Continuum, highlight key contributions they make to the advancement of knowledge, and offer recommendations for future research and monitoring.

The research I conducted in the Peace-Athabasca Delta (**Chapter 2** and **Chapter 3**) spans the entire Research-to-Inform Monitoring Continuum. I began my research in 2018 (as an MSc student and later fast-tracked to the PhD program), three years after *Data Collection* and *Data Analysis* had been completed. Thus, I was not involved in the *Conceptualization*, *Collaborative Planning* and *Funding Acquisition* phases that had been previously undertaken. At the time of my involvement, seasonal sampling and analysis for water isotope compositions was the primary method used to identify and characterize influential hydrological processes, because the methodology was well-suited for field sampling at numerous sites in remote, lake-rich landscapes, had low analytical costs and enabled quantification of water balance metrics and the distinction of different input source waters (Gibson and Edwards, 2002; Gibson et al., 2005; Brock et al., 2009; Turner et al., 2014; MacDonald et al., 2017, 2021). Additionally, the application of this methodology had been previously demonstrated at the PAD (Wolfe et al., 2007b, 2008, 2012; Yi et al., 2008; Remmer et al., 2018). However, we realized that some potentially important hydrological processes are short-lived, or may be difficult to detect, and therefore may not be adequately captured by the systematic seasonal analysis of water isotope compositions. Opportunity emerged, via funding from the Canada Foundation of Innovation, to deploy depth loggers at the 60 study lakes in the PAD that also served as sampling locations for water isotope composition. These continuous measurements effectively revealed the timing and magnitude of lake-level change across the PAD. Four common water-level patterns, each linked to a dominant hydrological process, formed the basis of a new lake classification for the PAD (Neary et al., 2021b [**Chapter 2**]): drawdown by evaporation, stable levels promoted by rainfall, gradual rise due to backwater flooding from adjacent waterbodies connected to the open-drainage network, and sharp rise from direct input of river floodwaters. A key advancement was the

discovery that open-water flooding, a previously under-recognized hydrological process, is a common and extensive source of input that sustains shallow lakes across the Athabasca sector of the delta. Deployment of depth loggers in 2018 and 2019 also provided critical information on the magnitude of water-level drawdown across the PAD – a key concern because it reduces habitat for aquatic species and can impede navigation by land users.

Recommendations stemming from the UNESCO Reactive Monitoring Mission report (WHC/IUCN, 2017) include to “*Establish adequate baseline hydrological information of the Peace and Athabasca River Basins to enhance the reference for monitoring and assessing current and future hydrological conditions*” (Recommendation 7) and “*Expand the scope of monitoring and project assessments to encompass possible individual and cumulative impacts on the Outstanding Universal Value of the property and in particular the PAD*” (Recommendation 9). Subsequent development of the WBNP Action Plan (WBNP, 2019) created important opportunities to align our research so it could contribute meaningfully towards the design and implementation of a long-term aquatic ecosystem monitoring program. A key contribution was production of a comprehensive Technical Report for WBNP (Remmer et al., 2020b; Appendix 3.1), which I co-authored, that summarized key findings and standard operating procedures from our research program during 2015-2019, marking the first step toward a transition to agency-led long-term monitoring (Figure 5.1). The value of the research program was recognized by Parks Canada, and they contributed funding from the WBNP Action Plan and staff resources to support some of the field sampling and analyses since 2020. Importantly, these contributions sustained the sampling of the lake and river sites during the period of Covid travel restrictions in 2020 and 2021, and annual reports were prepared to communicate key findings (Neary et al., 2021a; Imran et al., 2022). Without the pre-established partnership with WBNP and financial support from Parks Canada, the research program likely would not have continued beyond 2019, and we would have missed out on valuable insights gained from continued sampling in 2020 and 2021. These two years were among the wettest on record at the PAD, resulting in unusually high river discharge, widespread ice-jam and open-water flooding, and the expansion of lakes beyond their normal shorelines (e.g., Desrochers et al., 2023). Continued sampling during

this unusually wet period led to recognition that 1) an increase in relative humidity required re-configuration of parameters for isotope mass balance modelling (Imran et al., 2024) and 2) the role of snowmelt runoff on lake-level rise was substantial, rivalling that of floodwaters, during the spring of 2020 (Imran et al., in review) – key contributions of my colleague Arisha Imran’s MSc thesis (Imran, 2024). The findings presented in Imran et al. (2024) highlight the importance of periodically reassessing approaches and methodologies once long-term monitoring has been established.

Many of the results from analyses of samples collected during the first five years (2015-2019) of the research program in the PAD were communicated in journal articles and theses by other graduate students. These included hydrological research based on measurements of water isotope composition by Remmer et al. (2020a,c) and scientific determination of the extent of enrichment of substances of concern in lake sediment by Owca et al. (2020) and in periphytic biofilm by Savage et al. (2021). Although each of these publications, and including Neary et al. (2021b [**Chapter 2**]), provided some level of synthesis and recommendations, they primarily communicated conclusions stemming from focused and specific research within the *Data Collection* through to *Communication of Results* phases of the Research-to-Inform Monitoring Continuum (Figure 5.1). Once these products were published and the comprehensive Technical Report that summarized our findings was compiled (Remmer et al. 2020b; Appendix 3.1), opportunity emerged to contribute to the penultimate phase of the Continuum – to generate a comprehensive synthesis of the findings from all 7 years (2015-2021) of sample collection and analysis and provide recommendations for a long-term monitoring framework for lakes in the PAD (Neary et al. 2024 [**Chapter 3**]). Ultimately, this contribution spans the entire Research-to-Inform Monitoring Continuum (Figure 5.1). Fortunately, the period 2015-2021 captured marked variation in meteorological and hydrological conditions and their effects on lake water balance, depth and chemistry across the PAD. Correlation of temporal variation in hydrological and limnological metrics with large-scale climatic indices revealed the sensitivity of the measurements for long-term monitoring and their potential for predicting responses of lakes in the delta to climatic shifts. Concentrations of nickel and vanadium (key oil sands indicators) remain within the range of

natural variation in surface sediment and periphytic biofilm. The combination of hydrological, limnological, and contaminant-based approaches synthesized in Chapter 3 (Neary et al., 2024) serves as a guide for an effective long-term aquatic ecosystem monitoring program for PAD lakes, addressing pressing concerns over degradation from upstream industrial development and ongoing climate change. Adoption of the monitoring approaches we advocate for in Chapter 3 (Neary et al., 2024) is especially timely to assess the effectiveness of mitigative actions under active consideration by the WBNP Action Plan (WBNP, 2019), including installation of weirs in channels adjacent to open-drainage lakes and strategic water releases from the WAC Bennett Dam, and to assess for effects of possible releases of treated oil sands process-affected water into the Athabasca River.

When early results of the research project in the PAD were presented to the Warden and Natural Resource Manager of WBNP in 2016, they asked if we would be interested to apply similar sampling methodologies at a pond-rich landscape at the northern end of the park where the endangered whooping crane breed. Soon after, the UNESCO Reactive Monitoring Mission report (WHC/IUCN, 2017) recommended to “*closely monitor the entire used and potential nesting area of the Whooping Crane... so as to be able to respond to possibly changing management requirements*” (Recommendation 16). In 2020, our program was able to act on this invitation and this presented opportunity for me to become involved during the *Conceptualization* phase of the Research-to-Inform Monitoring Continuum, which benefitted from the experience we had gained at the PAD and communicated in **Chapter 2** (Neary et al., 2021b) and **Chapter 3** (Neary et al., 2024). After securing funding from several sources (NSERC Discovery and Northern Research Supplement grants, Northern Water Futures, Polar Continental Shelf Program), we collaborated with conservation ecologists and managers at WBNP and Environment and Climate Change Canada (ECCC) to launch a research program focused on shallow ponds in the Whooping Crane Summer Range (WCSR). By this later stage of my PhD, I was well positioned to take a leadership role in the early phases (*Conceptualization*, *Collaborative Planning* and early *Data Collection*) of the Research-to-Monitoring Continuum (Figure 5.1). During regular meetings with conservation ecologists at WBNP (Lori Parker) and ECCC (Mark Bidwell, John Conkin) we identified key

knowledge gaps, developed a study design, selected the sampling sites, determined the optimal timing for the field-based hydrological and limnological measurements to best inform conservation decisions for the whooping crane, and coordinated the field sampling to avoid helicopter encounters with whooping cranes. Together, we launched a pilot project in 2021 to determine if water levels, water balance and chemistry of ponds that had previously been used as nesting sites by whooping cranes differ from ponds that had not been used since 1966 when nest site locations began to be recorded. The results, however, revealed no differences between these pond types, and we recognized that the paired-samples design we had used was not well-suited for geospatial interpolation of results to determine how hydrological processes vary over space and time (Appendix 3.2).

Findings from the 2021 pilot project prompted further *Collaborative Planning* to refine the sampling design, which led to a decision to concentrate the sample sites within the Sass-Klewi Nesting Area (SKNA) of the WCSR, which is a smaller, more logistically feasible area to sample where whooping crane have nested for the longest and nest density is highest. A geographic grid (3 km x 3 km grid cells) was used to select 60 ponds dispersed at relatively consistent distances across the SKNA landscape, which enabled geospatial interpolation to evaluate the variation of key measurements across space and time — an approach that had proven to be an effective communication tool in the PAD. **Chapter 4** (Neary et al., in review) presents the results obtained during the open-water season of 2022 using this sampling design and the same combination of measurements and methodologies employed at the PAD (hourly measurements of water depth, seasonal measurements of water isotope compositions and water chemistry) to characterize hydrological processes regulating water levels and water balance of ponds and nesting wetlands across SKNA. Fortuitously, meteorological conditions in 2022 generated substantial snowpack but very little rainfall, which provided opportunity to evaluate the influence of input from snowmelt, loss via evaporation and hydrological connectivity to groundwater. Results showed that peak water levels occurred in spring when consistently low isotope compositions and low concentrations of major nutrients and ions reveal widespread, profound influence of snowmelt runoff. Spatial variability of these measurements increased during summer and

fall as connectivity to groundwater waned and influence of evaporation increased at some waterbodies. Although nearly every pond experienced water level drawdown, waterbodies that received near-continuous groundwater discharge underwent less drawdown. Geospatial interpolation of a multi-metric 'Vulnerability Index', based on isotope-inferred evaporation-to-inflow ratios, water-level drawdown and water chemistry ordination sample scores, identified that Whooping Crane nest locations in 2022 cluster within areas where waterbodies retained strong connectivity to groundwater, suggesting that wetland vulnerability to desiccation may be one of many factors influencing nest site selection. Application of the methodology in subsequent years will provide opportunity to evaluate responses of hydrological processes to a greater range of meteorological conditions and inform the design of an effective long-term aquatic ecosystem monitoring program.

A key priority of ongoing research will be to evaluate how the Vulnerability Index varies over space and time to a wider range of meteorological conditions than those observed in 2022. For example, recent dry summers fueled extensive forest fires near our study area in 2023 (Jain et al., 2024; Kirchmeier-Young et al., 2024; Wang et al., 2025). Whooping cranes are known to have high nest site fidelity (Kuyt, 1981; Johns et al., 2005), yet some pairs placed their nests where the Vulnerability Index scores were moderately high in 2022 (i.e., 0.5-0.6). It is possible that the whooping crane may have chosen these nest locations in previous years when rainfall was higher and less drawdown occurred, and, thus, had lower Vulnerability Index scores when the nest sites were selected. Long-term application of the Vulnerability Index could therefore help address remaining knowledge gaps, including: 1) Do whooping cranes select nesting locations where waterbodies are most resilient to water-level fluctuations, or do they select nest sites based on other aspects of habitat quality, such as food availability and shoreline vegetation?, and 2) Is there an association between Vulnerability Index scores and rates of breeding success and failure across the landscape? Ultimately, the findings and framework presented in **Chapter 4** lay the foundation for continued data collection to improve our understanding of hydrological processes regulating pond responses to changes in meteorological conditions as the research progresses along the Continuum (Figure 5.1) in the next few years.

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Appendix 1 | Supplementary material for Chapter 3

The following pages present information on 1) the genesis of the study, 2) the Materials and Methods, including details of site selection, and the field, laboratory and analytical methods used for the hydrology, water chemistry and contaminants in sediment and periphytic biofilm components of the research, and 3) the water isotope framework and stable isotope mass balance modelling of lake water balance in the Peace-Athabasca Delta. All data reported in **Chapter 3** are made available in Supplementary Material 2 of Neary et al. (2024) at <https://cdnsiencepub.com/eprint/VFGEGNYZDAZ2JZWRBTR6/full>.

A1.1 Genesis of the study

The data we report in **Chapter 3** (Neary et al., 2024) extend from 2015-2021, however, the research is an outcome of key observations and developments achieved since 2000 when our program initiated paleolimnological investigations to evaluate the relative roles of Peace River flow regulation and climate on the observed drawdown of the delta's perched lakes during the 1980s and 1990s. During the first field visit in October 2000, we conducted a spatial survey of 60 lakes spanning the hydroecological gradients of the PAD and sites in the major rivers for measurements of water isotope composition, water chemistry and aquatic biota to inform interpretation of the paleoenvironmental data obtained from analyses of lake sediment cores (Wolfe et al., 2007b). Results from the survey revealed profound variability over space in the influence of hydrological processes on lake water balances and water chemistry. Based on that delta-wide water isotope survey, Wolfe et al. (2007b) provided a spatial representation of lake water evaporation-to-inflow (E/I) ratios, which we now refer to as 'isoscares' and displays variation in lake water balances across the delta in map form. This 'snapshot' illustration of hydrological conditions across the vast landscape was viewed as a distinct strength of the approach, because it communicated effectively to a variety of audiences where 'hotspots' of lake drawdown by evaporation occurred and where positive lake water balance occurred due to input of floodwaters (see also Wolfe et al., 2007a). Another early isoscare was used to hypothesize that a recent geomorphic change in the flow path of the Athabasca River had a major influence on the water balance of lakes across the Athabasca sector (Wolfe et al., 2008). Subsequent systematic application of water isotope

tracers to characterize hydrological conditions of shallow lakes across other remote northern water-rich landscapes, including the Slave River Delta (Northwest Territories), Hudson Bay Lowlands (northern Manitoba) and Old Crow Flats (Yukon Territory), was used to establish quantitative relations with drivers including meteorological conditions, river discharge and catchment characteristics, often communicated as isoscapes (Brock et al., 2007, 2008, 2009; Turner et al., 2010, 2014; MacDonald et al., 2017). These collective efforts and experiences ultimately led to motivation and confidence that similar systematic application of water isotope tracers across lakes of the PAD could provide novel opportunity to capture the influence of evaporation and river floodwaters on lake water balances, the two most important and scrutinized hydrological processes, over space and time.

Early during our paleolimnological research in the PAD, we observed layers of black oily sediment in cores retrieved from flood-prone oxbow lakes (Wolfe et al., 2006). Radiocarbon analyses revealed they contained old (i.e., geological) carbon that we hypothesized was sourced from upstream exposures of bitumen and coal deposits that had been eroded, transported, and deposited by episodic flood events for centuries. Since that stratigraphic discovery, concern increased over aquatic ecosystem degradation at the PAD by releases of substances of concern from rapidly expanding oil sands development, which led us to identify a new research opportunity – to use analyses of sediment cores from lakes in the PAD to generate ‘pre-disturbance’ concentration baselines as a valuable reference point for quantifying the enrichment of substances of concern attributable to industrial activities. Indeed, highly critical reviews of regional monitoring programs had identified the absence of knowledge of the range of natural concentrations of substances of concern before onset of large-scale bitumen mining and processing as a key limitation undermining an ability to evaluate the extent of contamination attributable to oil sands development within the Lower Athabasca River and the PAD (Gosselin et al., 2010; Schindler, 2010; Dillon et al., 2011). This recognition led to a recommendation by the Federal Oil Sands Advisory Panel that sediment cores from floodplain lakes be used to establish the “natural, pre-development state” of the river and delta (Dowdeswell et al., 2010 p. 31-32). In response to this recommendation and to address this critical knowledge gap, our foundational contaminant studies in the PAD

have demonstrated the unique value of using sediment core records from upland and floodplain lakes in the PAD to establish pre-disturbance baseline concentrations for substances of concern, and to quantify the degree of enrichment since oil sands development via atmospheric and fluvial pathways, respectively (Hall et al., 2012; Wiklund et al., 2012b, 2014). These studies analysed concentrations of polycyclic aromatic compounds (PACs), trace elements, and total mercury, and established pre-disturbance baselines to characterize the range of variation before industrial development. Their measurement in aquatic sediment is useful because most have low solubility in water and, thus, are mainly adhered onto sediment particles and organic matter (Shotyk et al., 2021).

Based on knowledge gained from several spatial surveys of water isotope composition and chemistry and paleolimnological research conducted between 2000 and 2014, which included recommendations for an aquatic ecosystem monitoring program for the PAD detailed in a synthesis of paleolimnological results (Wolfe et al., 2012), we developed a proposal in February 2014 for research to inform approaches for long-term aquatic ecosystem monitoring. For several years prior to this, early concepts were communicated at multi-stakeholder meetings in Fort Chipewyan and Edmonton (Alberta), organized by Wood Buffalo National Park (WBNP) under the former PAD Ecological Monitoring Program (PADEMP). The meetings were pivotal because they facilitated exchange of ideas and stimulated new collaborative opportunities and partnerships that allowed us to launch the project in 2015. The research received financial support from the Natural Sciences and Engineering Research Council of Canada Collaborative Research and Development partnership program, with financial and logistical support from stakeholders in government (Alberta Environmental Monitoring, Evaluation and Reporting Agency (now within Alberta Environment and Protected Areas)), industry (BC Hydro, Canadian Natural Resources Ltd., Suncor Energy Inc.), and Parks Canada Agency. This support was subsequently used to leverage other grants, including the Global Water Futures / Northern Water Futures program, which extended the duration of research from 2 to 5 years. Multi-agency support for the program provided clear indication of the importance and need for this research. WBNP employees and members of the local CBMs were involved in the fieldwork during 2015-2016,

and they continued critical aspects of the sampling program during the Covid-19 pandemic (2020-2021), funded in part by the WBNP Action Plan, which extended some of the data sets to 7-year duration.

A1.2 Methods and Materials

A1.2.1 Selection and locations of the study lakes and river sites

Lake selection was finalized at an annual PADEMP Forum in February 2015, where the researchers, Wood Buffalo National Park (WBNP) employees, and local stakeholders and knowledge holders convened. Sixty lakes were selected to span much of the geographic extent of the PAD and capture the broad range of hydrological conditions, which includes open-, restricted- and closed-drainage lakes. They include 41 lakes from the initial October 2000 survey reported in Wolfe et al. (2007b) and 19 lakes where local community-based monitoring programs had begun to track changes in muskrat abundance, water depth and ice thickness, based mainly on sampling in winter via snowmobile access (the latter are denoted with the prefix “M” in the site IDs in Figure 3.1 and Supplementary Material 2 Table S1 of Neary et al. (2024)). Twenty-six of the lakes are in the Peace sector and 34 are in the Athabasca sector. Nine river sites were selected to characterize the water isotope composition and water chemistry of the rivers for determining the influence of floodwaters on lakes.

A1.2.2 Field sampling

Twenty-one field sampling campaigns were carried out during 2015-2021. Figure 3.2 provides a timeline for key developments of the lake monitoring framework, and an overview of when sampling occurred for each of the three research components. Due to absence of roads and an abundance of dense terrestrial vegetation and waterlogged terrain in the PAD, travel by river and land was inefficient for accessing the lakes. Almost all the sample collection and deployment of equipment was conducted from a helicopter with fixed floats. Water samples for measurement of isotope composition and chemistry, limnological measurements with a YSI multimeter, photographs and field observations were systematically obtained three times per year (typically end of May, mid-July and mid-September) to characterize the spatial and temporal variation in water balance, hydrological

processes and water chemistry during the open-water season at each of the lake and river sites. This holds true for every year except spring of 2016 when extensive forest fires in the region caused the fieldwork to be delayed until late June. Beginning in 2018, water-depth loggers were deployed at a central location of each lake in May and retrieved in September. Samples of surface sediment and periphytic biofilm for the contaminant component were collected less frequently than for the other two components. This included deployment of artificial-substrate samplers in all lakes between May and September of 2017 and 2018, and collection of lake surface sediment at all lakes in 2017 and at 20 spring-flooded lakes in 2018.

Below, we detail the field, laboratory and analytical methods associated with the hydrology, water chemistry and contaminants in sediment and periphytic biofilm components of the study.

A1.2.3 Hydrology

A3.2.3.1 Measurement of water isotope composition

During each sampling campaign, a water sample was collected from ~10 cm below the water's surface at each site and stored in a 30 mL narrow-mouth HDPE bottle for measurement of water isotope composition. Sample collection in May is intended to capture influence of input from snowmelt and ice-jam induced floodwaters on lake water balance in spring, whereas the July samples inform about the influence of rainfall, open-water flooding and evaporation at the time of peak biological productivity in summer, and the September samples provide an integration of open water season hydrological conditions before ice cover forms in fall. After each sampling episode, the samples were analyzed at the University of Waterloo – Environmental Isotope Laboratory (UW-EIL) using a Los Gatos Research (LGR) Liquid Water Isotope Analyser (LWIA) model T-LWIA-45-EP instrument with precisions (2σ) of $\delta^2\text{H} = \pm 0.8 \text{ ‰}$ and $\delta^{18}\text{O} = \pm 0.2 \text{ ‰}$. A suite of water standards (Vienna Standard Mean Ocean Water, Vienna Standard Light Antarctic Precipitation) from the International Atomic Energy Agency (IAEA) were used as reference materials and duplicate samples were run at a minimum on every tenth sample. Isotope compositions are expressed as δ -values, representing deviations in per mil (‰) from Vienna Standard Mean Ocean Water

(VSMOW). Results of $\delta^{18}\text{O}$ and $\delta^2\text{H}$ analyses are normalized to -55.5‰ and -428‰ , respectively, for Standard Light Antarctic Precipitation (Coplen, 1996).

Lake water isotope compositions were used to estimate the evaporation-to-inflow (E/I) ratio, an informative metric of lake water balance (Craig and Gordon, 1965; Gonfiantini, 1986; Gibson and Edwards, 2002; Gat, 2008). Positive water balance ($E/I < 1.0$) identifies when input exceeds evaporative water loss, whereas negative water balance ($E/I > 1.0$) identifies when evaporation exceeds input. E/I ratios were estimated using a coupled isotope tracer approach (Yi et al., 2008; Supplementary Material 2 Table S2 in Neary et al. (2024)). For more detail on isotope mass-balance equations that were used to estimate E/I ratios, see section 3 below. For samples collected during 2015-2019, we utilize the isotope framework from Remmer et al. (2020a). For samples collected during 2020-2021, when average ice-free season flux-weighted relative humidity rose by 4.6%, we utilize a modified framework (Imran et al., 2024). The parameters that comprise each of these frameworks are recorded in Supplementary Material 2 Table S3 in Neary et al. (2024). Ordinary kriging was used to generate ‘isoscapes’ for each sampling episode, which provide representations of the spatial distribution of lake E/I ratios across the landscape. The isoscapes were generated using ArcMap V.10.8.2, and interpolation by ordinary kriging was justified based on statistically significant Moran’s I values (at $\alpha = 0.05$) for each isoscape (Supplementary Material 2 Table S4 in Neary et al. (2024)). These methods are consistent with Remmer et al. (2020a), with the additional improvement of a masking grid layer (i.e., a shapefile) that provides more accurate delineation of the PAD by excluding Lake Claire, Lake Athabasca and the region to its north, and an elevated area of sand dunes beyond the southern margin of the delta.

A3.2.3.2 Continuous water depth measurements from data loggers

From 2018 to 2021, water depth loggers (onset HOBO Water Level Data Logger (0-4 m); SKU: U20-001-04) were deployed at a central location in each lake between May and September (Neary et al., 2021b [**Chapter 2**]). Before deployment, the loggers were set to record at hourly intervals and attached to a rock-sock anchor with sufficient rope to allow a wooden float to remain at the lake water surface. A variable number of loggers were successfully retrieved from the lakes each year (48 in 2018, 53 in 2019, 45 in 2020 and 31 in 2021). The hourly logger

data obtained from each lake were converted from absolute pressure (psi) to lake depth (m) using the Barometric Compensation Assistant in the HOBOWare Pro software program and ambient air temperature and pressure data recorded by a logger deployed outside WBNP's field house in Fort Chipewyan at the same hourly intervals. This is true for all the years except in 2021 when air temperature and pressure recorded at a local meteorological station (station 3072659) in Fort Chipewyan were used to convert the logged pressure readings to water depth. The lake depth data were compiled using Microsoft Excel (Supplementary Material 2 Table S5 in Neary et al. (2024)) and a 24-hour moving average was computed using the *pracma* package (Borchers, 2023) in RStudio version 2023.06.0 to reduce minor high-frequency variability in the datasets. The amount of drawdown that occurred at each pond was determined as the minimum depth minus initial depth and expressed in centimeters. Lake-depth variations were then sorted into the four lake-level categories (stable, drawdown, gradual rise and sharp rise) using the criteria established by Neary et al. (2021b) to inform about the main hydrological processes influencing lake depth variations (Supplementary Material 2 Table S6 in Neary et al. (2024)).

A1.2.4 Water chemistry

During each field sampling campaign in 2015-2019 and 2021 (samples were not collected in 2020 due to challenges brought on by the Covid-19 pandemic), ~5 litres of water were collected from each lake and river site at ~10 cm below the surface using a 4-L carboy and a 1-L Nalgene bottle. This quantity exceeded the requirement for subsequent water chemistry measurements (~750 mL) but was collected to ensure ample volume in case of errors during sample handling and processing. Also, *in situ* measurements of pH, temperature, turbidity and specific conductivity were obtained at ~10 cm below the water surface using a YSI ProDSS multiparameter water quality meter.

The water samples were stored in a refrigerator at the field base (Fort Chipewyan) until further processed. The processing involved filling a 500 mL bottle with water for measurement of 'total' constituents (i.e., turbidity, pH, and concentrations of total phosphorus (TP), total nitrogen (TN), and major ions), and a 250 mL bottle with water after passing it through a 0.45- μ m pore-size cellulose acetate filter (using a vacuum pump) for

measurement of ‘dissolved’ constituents (i.e., dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), dissolved silica (DSi)). The processed water chemistry samples were promptly shipped (refrigerated) to the Biogeochemical Analytical Service Laboratory (BASL) at University of Alberta (Edmonton, Canada) for analysis. Samples were analyzed for the following: pH, turbidity, and concentrations of TN, TP, conductivity, DSi, DIC, DOC, bicarbonate (HCO_3^-), major anions (Cl^- , SO_4^{2-}) and major cations (Ca^{2+} , K^+ , Mg^{2+} , Na^{2+} ; Supplementary Material 2 Table S7 in Neary et al. (2024)). Concentrations of TN, TP, and DSi were determined by Flow Injection Analysis (TN Method: 353.2; TP Method: 4500-P-G; DSi Method: 370.1) using a Lachat QuickChem QC8500 FIA Automated Ion Analyzer. Concentrations of DIC and DOC were determined using a Shimadzu TOC-5000A Total Organic Carbon Analyzer (Method 415.1). Conductivity and pH in water samples were determined using the PC-titrate instrumentation (pH Electrometric Method 4500-H+B). Turbidity was determined using a Hach 2100N Turbidimeter (Method 180.1). Ionic concentrations were determined using a Thermo ICAP-6300 Inductively Coupled Argon Plasma – Optical Emission Spectrometer (Method 200.7). Note that some variables were measured both *in situ*, using the handheld YSI, and at BASL. We used *in situ* pH and turbidity measurements because of potential effects resulting from sample storage. For instances when the handheld YSI reported nonsensical values due to interference by macrophytes, we used turbidity measurements determined at BASL.

Principal component analysis (PCA), a form of multivariate ordination, was used to explore patterns of variation in water chemistry at 59 of the lakes (PAD 18 was removed because it is elevated above the delta and serves as an index lake for the water isotope framework) and 8 river sites (site R9 was removed because it was non-flowing for the majority of sampling episodes) in the spring, summer and fall of 2015-2019 and 2021. Prior to analysis by PCA, all variables were tested for normality, log-transformed if non-normal, and centred and standardized using *mvn* and *factoextra* packages in RStudio version 2023.06.0 (Korkmaz, et al., 2014; Kassambara and Mundt, 2020). Variables that were log-transformed include TN, TP, Cl^- , SO_4^{2-} , Na^{2+} , K^+ , DSi and turbidity. The PCA was generated using the *prcomp* function in base R (R Core Team, 2019; Supplementary Material 2 Table S8 in Neary et al. (2024)). The

water chemistry data from 2021 were included as supplementary samples (i.e., the data were not ‘active’) in a PCA of the 2015-2019 data to reduce the influence of a year with unusually widespread flooding (Desrochers et al., 2023) on the sample scores and environmental vectors. To visualize relations between gradients of water chemistry variables (defined by PCA axes 1 and 2) and basin hydrology, lakes were colour-coded in the ordination biplot according to three hydrologically distinctive categories: open-drainage lakes, flooded perched (i.e., closed- and restricted-drainage) lakes, and non-flooded perched lakes. Open-drainage lakes include Lake Claire (PAD 62), Mamawi Lake (PAD 45) and Richardson Lake (PAD 38). Sample scores for the river sites were also included in the PCA to assess variation of water chemistry across the full gradient of hydrological connectivity. The distribution of water chemistry variables among the hydrological categories is displayed in Figure A1.1.

Flooded and non-flooded perched lakes were distinguished using water isotope composition, specific conductivity and field observations for 2015-2019 (see data recorded in Tables S1.2-S1.5 of Remmer et al., 2020c). Similar methods were applied to the 2021 water chemistry data; however, specific conductivity was not used in 2021 because it was not measured consistently that year (see Supplementary Material 2 Table S9 in Neary et al. (2024)). We are, therefore, less confident in the designations of flood status in 2021. Our determination of whether a perched lake flooded during the weeks to months before sample collection is potentially influenced by the varying ‘memory’ of the isotope composition and specific conductivity of lake water, which reflects an unknown period of time before sampling took place. For example, perched lakes designated as flooded in spring received river floodwater between late winter and the time of spring sampling. In summer and fall, flooded perched lakes may have either flooded between the spring sampling episode and the summer or fall episodes, or retained the signal of spring flooding beyond the summer or fall sampling episodes. As demonstrated in Figure 3.5, separation of sample scores according to the three hydrological categories was apparent along PCA axis 2, thus we explored the spatial distribution of PCA axis 2 sample scores for each sampling episode (Supplementary Material 2 Table S10 in Neary et al. (2024)). We refer to these geostatistical products (maps) as ‘limnoscapes’, which are presented in Figure 3.4c of **Chapter 3**.

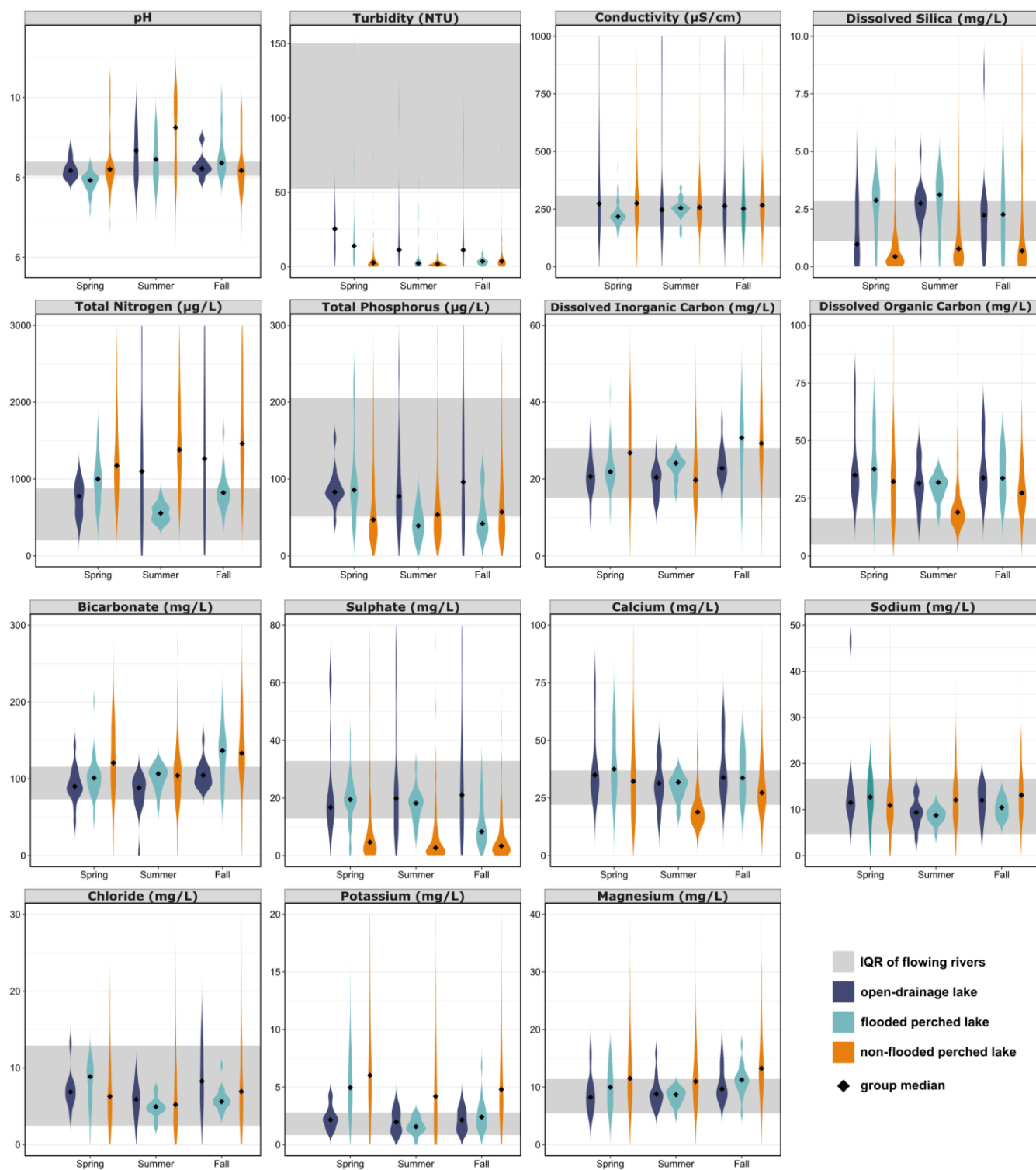


Figure A1.1. Violin plots showing distribution of water chemistry variables among lakes in different hydrological categories during 2015-2019. The range of values for flowing rivers is depicted as grey horizontal bars.

Consistent with methodology used to construct the isoscapes, we used ordinary kriging in ArcMap version 10.8.2 because Moran's I values were statistically significant at $\alpha = 0.05$ for all limnoscapes except Fall 2015 (Supplementary Material 2 Table S4 in Neary et al. (2024)).

A1.2.5 Contaminants in sediment and periphytic biofilm

For the contaminants component, we synthesize results from analysis of samples of periphytic biofilm obtained from 49 lakes in 2017 and 42 lakes in 2018, and from surface sediment collected at 58 lakes in 2017 and from a subset of 20 lakes (8 in the Peace sector, 12 in the Athabasca sector) in 2018 after they received influx of ice-jam induced floodwaters (Owca et al., 2020; Savage et al., 2021). Surface sediment was not collected at PAD 9 and PAD 20 in 2017 because they contained little to no water at that time. The surface sediment samples (uppermost ~1 cm) were collected at the centre of each lake in mid-September 2017 and July 2018 using 3-5 deployments of a mini-Glew corer (Glew, 1991), and stored in a Whirl-Pak bag at 4 °C until prepared for analysis. In both years, artificial substrate samplers (52.5 x 18 cm HDPE sheets), attached via nylon rope between a 45-cm-long wooden float and rock-sock anchor, were deployed in May at the centre of each lake. During collection in September, the samplers and accrued periphytic biofilm were retrieved, placed into large Ziploc bags, and stored frozen. Shortly after collection, the surface sediment and artificial substrate samplers were shipped in coolers with ice packs to the University of Waterloo for analysis.

Biofilms were scraped off the artificial substrate samplers using a plastic spatula, placed into containers and frozen until further processing. Subsamples of well-mixed lake surface sediment and biofilm-sediment mixtures were frozen, freeze-dried, homogenized using a mortar and pestle and subsampled (~1 gram) prior to analysis of elemental concentrations. Lake surface sediment samples were analyzed at ALS Canada Ltd. (Waterloo, Canada) following EPA method 200.2/6020A and the biofilm-sediment mixtures were analyzed at ALS Canada Ltd. (Vancouver, Canada) following EPA Method 200.3/6020A. These analyses provided concentrations of a suite of 34 elements, including the metal(loid)s: aluminum (Al), antimony (Sb), arsenic (As), barium (Ba), beryllium (Be), bismuth (Bi), boron (B), cadmium

(Cd), calcium (Ca), chromium (Cr), cobalt (Co), copper (Cu), iron (Fe), lead (Pb), lithium (Li), magnesium (Mg), manganese (Mn), molybdenum (Mo), nickel (Ni), phosphorus (P), potassium (K), selenium (Se), silver (Ag), sodium (Na), strontium (Sr), sulfur (S), thallium (Tl), tin (Sn), titanium (Ti), tungsten (W), uranium (U), vanadium (V), zinc (Zn) and zirconium (Zr).

Here we focus on analyses performed on concentrations of nickel (Ni) and vanadium (V) because they serve as key indicator metals of emissions from oil sands development due to their elevated concentrations in the mined bitumen ore relative to other geological sources, and in mining wastes (Gosselin et al., 2010; Shotyky et al., 2021), and because their analysis is cost-effective relative to more hazardous compounds (e.g., polycyclic aromatic compounds). We quantify the enrichment of Ni and V by comparing concentrations in the surface sediment and periphytic biofilm samples to sector-specific (Peace, Athabasca) pre-disturbance baselines that have been generated from analyses of river-supplied sediment deposited before ~1920 (see Kay et al., 2020, 2023; Owca et al., 2020; Savage et al., 2021). Our program has continued to improve the pre-disturbance baselines by increasing the number of sites and samples, thus, here, we present the most updated pre-disturbance baselines, established using radiometrically dated-sediment cores from 7 lakes in the Athabasca sector (PAD 26, PAD 30, PAD 31, PAD 32, PAD 71, M2, M7), 3 lakes in the Peace sector (PAD 52, PAD 65, PAD 67) and sediment from riverbank exposures from both rivers (Supplementary Material 2 Table S11 in Neary et al. (2024)). For all samples, concentrations of Ni and V are normalized to aluminum (Al) concentration to account for confounding influence of natural variability in mineralogy and changes in sediment grain size caused by shifting flow energy, as occurs in floodplains (Loring, 1991; Kirsten and Smedes, 2002). Concentrations of Ni and V in the surface sediment samples are expressed as Enrichment Factors (EF) to quantify the degree of enrichment, using the sector-specific baselines (Supplementary Material 2 Table S12 in Neary et al. (2024)). EFs were computed with the same equation used by Owca et al. (2020) and Kay et al. (2020, 2023), which express the Al-normalized concentrations relative to the pre-disturbance baseline, such that an EF of 1.0 indicates no enrichment relative to the baseline whereas an EF of 2.0 indicates the concentration has doubled relative to the baseline:

$$\text{Enrichment Factor (EF)} = \frac{(M/Al)_{\text{sample}}}{(M/Al)_{\text{pre-disturbance}}}$$

$M_{\text{pre-disturbance}}$ and $Al_{\text{pre-disturbance}}$ are the expected values for the metal of interest and Al in sediments deposited before ~1920, derived from the equation of the regression line for the metal-Al relation where $Al_{\text{pre-disturbance}} = Al_{\text{sample}}$. Thresholds and terminology presented in a widely cited comprehensive review by Birch (2017) provide internationally consistent categorization of the degree of enrichment above pre-disturbance baselines, where EFs ≤ 1.5 represent the range of natural variability or ‘pristine conditions’, EFs $> 1.5\text{--}3.0$ represent ‘minimal enrichment’, EFs $> 3.0\text{--}5.0$ represent ‘moderate enrichment’, EFs $> 5.0\text{--}10.0$ represent ‘considerable enrichment’ and EFs > 10.0 represent ‘severe enrichment’. EFs and Birch’s (2017) thresholds have been used to evaluate paleolimnological records in the PAD (Kay et al., 2020, 2022, 2023) and Lower Athabasca River (Klemm et al., 2021).

A1.2.6 Hydrometric and meteorological data

Hydrometric and meteorological data were extracted from publicly available sources to evaluate their influence on the status and trends in metrics derived from the measurements made at the lake and river sites. The hydrometric data include measurements of daily discharge for Peace River at Peace Point (station 07KC001; 59°07'05" N, 112°26'13" W) and Athabasca River at Embarras Airport (station 07DD001; 58°18'45" N, 111°30'54" W), and daily measurements of open-drainage network water levels at Lake Athabasca (station 07MD001; 58°42'47" N, 111°07'20" W) and Mamawi Lake (station 07KF003; 58°37'54" N, 111°20'03" W). Hydrometric data were obtained from: <https://wateroffice.ec.gc.ca/> and <https://rivers.alberta.ca/>. Meteorological data (air temperature, precipitation, snow depth) recorded at the Fort Chipewyan airport (station 3072659; 58°46'02" N, 111°07'02" W) were obtained from: https://climate.weather.gc.ca/historical_data/search_historic_data_e.html. Data from the meteorological station at Fort Smith (station 2202202; 60°01'34" N, 111°55'46" W), NWT, ~140 km north of Fort Chipewyan, were used to fill gaps in the Fort Chipewyan meteorological record for daily precipitation in 2016 and snow on the ground in 2018 and 2019. To contextualize hydrometric and meteorological data during 2015-2021, we compare the data with variable-specific long-term averages and interquartile ranges (IQR).

We based these on a 30-year interval spanning 1981-2010, when possible. For the variables with substantial missing data (e.g., >1 month) during 1981-2010, we derived the long-term averages and IQRs using data that were available. This included 1971-1984 for Athabasca River discharge measurements and 1968-1984 for snow depth measurements at Fort Chipewyan. Notably, the range of hydrological conditions during the research program (2015-2021) is representative of the historical (1931-present) range of water levels for Lake Athabasca (Figure A1.2). Aside from period when the Williston Reservoir was filled (1968-1972), Lake Athabasca water levels during 2015-2021 span a similar range (204.1-211.2 m) as the pre- and post-regulation time periods (205.6-211.3 m and 206.4-210.8 m, respectively).

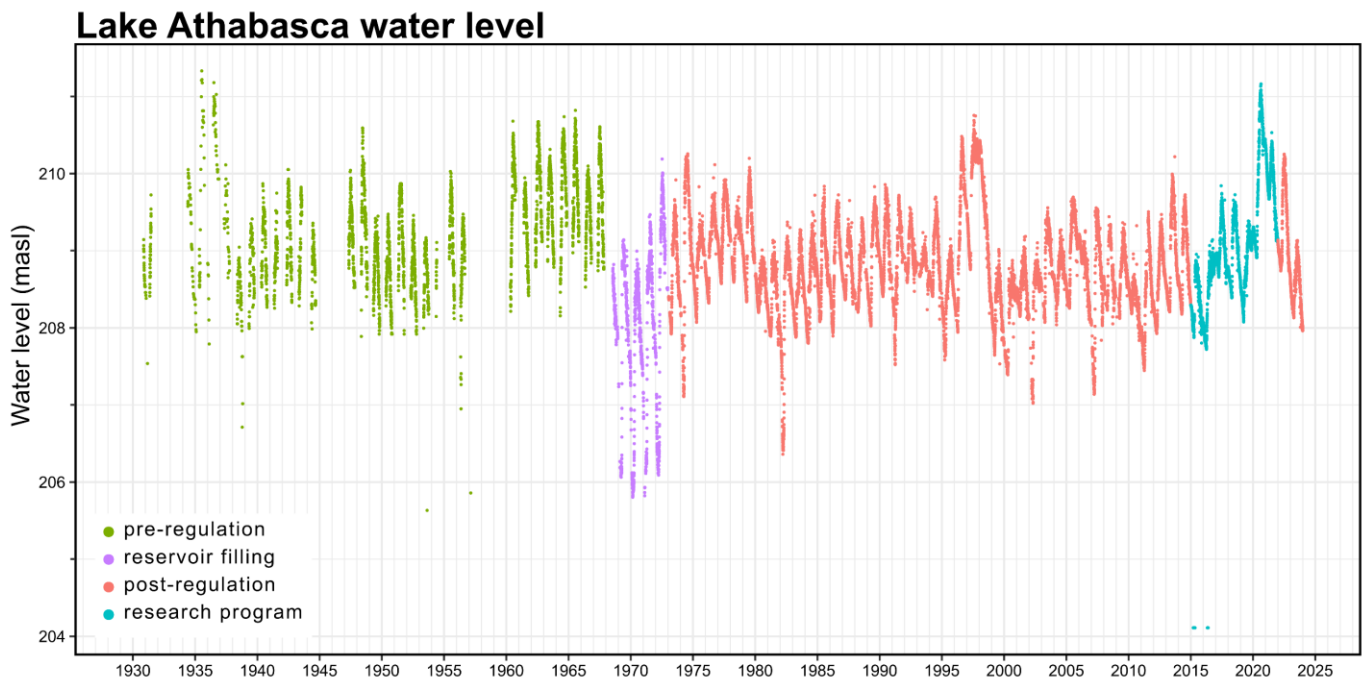


Figure A1.2. Water level time series of Lake Athabasca from 1931–present (station 07MD001). Coloured points correspond to major time periods (green = pre-regulation, purple = reservoir filling, red = post-regulation, blue = research program (2015-2021)).

A1.3 Isotope framework and stable isotope mass balance modelling of lake water balance in the Peace-Athabasca Delta

Below we detail a series of 4 steps used to estimate the lake evaporation-to-inflow (E/I) ratios reported in the **Chapter 3**.

Step 1: Calculate isotope fractionation (α^*) and separation (ϵ^* , ϵ_k) factors using local meteorological data

The Peace-Athabasca Delta (PAD) experiences marked seasonal variation in meteorological conditions, where surface water bodies (lakes) interact with the atmosphere during the open-water season (May to September). Consequently, we use PET flux-weighted relative humidity (h_{FW}) and temperature (T_{FW}) to estimate α^* (equilibrium fractionation factor), ε^* (equilibrium separation factor) and ε_k (kinetic separation factor), following recommendations by Gibson (2002).

$$h_{FW} = \sum_{i=1}^n \frac{(h_i \times E_i)}{E_i}$$

$$T_{FW} = \sum_{i=1}^n \frac{(T_i \times E_i)}{E_i}$$

In the above equations, h_i and T_i are hourly relative humidity and temperature measurements, respectively, E_i is monthly potential evapotranspiration and n represents the number of months. Hourly relative humidity and temperature data at Fort Chipewyan were obtained at <https://climate.weather.gc.ca>.

Potential evapotranspiration for each month was estimated using the equation by Thornthwaite (1948):

$$E = 1.6 \times \frac{L}{12} \times \frac{N}{30} \left[\frac{10 \times T}{I} \right]^a$$

where L is the average day length for the month expressed in hours, N is the number of days in the month, I is the ice-free season heat index, and a is a calculated coefficient.

$$L = \frac{\text{sun}}{N}$$

$$I = \sum_{i=1}^n \left[\frac{T}{5} \right]^{1.5}$$

$$a = 0.49 + 0.0179 \times I - 7.71 \times 10^{-5} \times I^2 + 6.75 \times 10^{-7} \times I^3$$

We obtained the total hours of sunshine (sun) for each month using National Research Council Canada's sunrise/sunset calculator (<http://www.nrc-cnrc.gc.ca/eng/services/sunrise/>).

α^* and ε^* were determined for oxygen and hydrogen independently using equations by Horita and Wesolowski (1994):

$$1000 \times \ln(\alpha_{\text{oxygen}}^*) = -7.685 + 6.7123 \left(\frac{10^3}{T_{FW}} \right) - 1.6664 \left(\frac{10^6}{T_{FW}^2} \right) + 0.35041 \left(\frac{10^9}{T_{FW}^3} \right)$$

$$1000 \times \ln(\alpha_{\text{hydrogen}}^*) = 1115.8 \left(\frac{T_{FW}^3}{10^9} \right) - 1620.1 \left(\frac{T_{FW}^2}{10^6} \right) + 794.84 \left(\frac{T_{FW}}{10^3} \right) - 161.04 + 2.9992 \left(\frac{10^9}{T_{FW}^3} \right)$$

$$\varepsilon^* = \alpha^* - 1$$

The kinetic separation factors (ε_k) were determined using equations provided by Gonfiantini (1986):

$$\varepsilon_{k-oxygen} = 0.0142(1 - h_{FW})$$

$$\varepsilon_{k-hydrogen} = 0.0125(1 - h_{FW})$$

Step 2: Develop an isotope framework for the Peace-Athabasca Delta

An isotope ‘framework’ is best represented by two linear relationships that reflect mass-dependent partitioning of stable isotopes contained within the water molecule as it passes through the hydrological cycle (Edwards et al., 2004), and which serve as important references for subsequent isotope mass-balance equations outlined in Steps 3 and 4 below. One line is the Local Meteoric Water Line (LMWL), which represents a range of isotope values for precipitation in a given area. For the PAD, the LMWL was derived using a least squares regression through amount-weighted isotope compositions of precipitation collected in Fort Smith (Northwest Territories) as part of the Global Network for Isotopes in Precipitation (GNIP) in 1962-1969 (as first reported in Wolfe et al., 2007b). The other line is the Local Evaporation Line (LEL), which represents the trajectory of isotope composition in a surface water body as it undergoes progressive evaporative enrichment and is anchored by two points: δ_p and δ^* . δ_p represents the average annual isotope composition of precipitation and was obtained from the Online Isotopes in Precipitation Calculator (OIPC; Bowen, 2023). δ^* represents the limiting isotope composition of a lake as it approaches desiccation. This value was derived using the equation provided by Gonfiantini (1986):

$$\delta^* = \frac{h_{FW} \times \delta_{AS} + \varepsilon_k + \frac{\varepsilon^*}{\alpha^*}}{h_{FW} - \varepsilon_k - \frac{\varepsilon^*}{\alpha^*}}$$

where δ_{AS} is the isotope composition of atmospheric moisture for the ice-free season (Gibson et al., 1999; Gibson and Edwards, 2002).

$$\delta_{AS} = \frac{\left[\frac{(\delta_{SSL} - \varepsilon^*)}{\alpha^* - \varepsilon_k - \delta_p(1 - h_{FW} + \varepsilon_k)} \right]}{h_{FW}}$$

A third point that falls along the LEL, δ_{SSL} , represents the isotope composition of a terminal lake in hydrological “steady state” (i.e., evaporation = inflow). In the PAD, an index lake (PAD 18) has been used to represent this parameter given its near-constant isotope composition over ~20 years (Remmer et al., 2020a,c). Recently, the isotope composition of PAD 18 shifted in response to a 4.6% increase in ice-free season flux-weighted relative humidity which

called for new parameterization of δ_{SSL} (Imran et al., 2024). See Supplementary Material 2 Table S3 in Neary et al. (2024) for parameters associated with each isotope framework.

Step 3: Estimate the isotope composition of evaporation (δ_E) and input (δ_I) water for each lake

A coupled isotope tracer approach (Yi et al., 2008) was used to estimate the isotope composition of input water (δ_I) to each lake using the intersection of lake-specific evaporation lines ($\delta_E - \delta^*$) with the Local Meteoric Water Line (LMWL). δ_E , the isotope composition of evaporative flux at each lake, was estimated using the equation (Craig and Gordon, 1965):

$$\delta_E = \frac{\left[\frac{(\delta_L - \varepsilon^*)}{\alpha^* - h_{FW} \times \delta_{AS} - \varepsilon_k} \right]}{(1 - h_{FW} + \varepsilon_k)}$$

where δ_L is the measured lake water isotope composition. The intersection between $\delta_E - \delta^*$ and the LMWL (δ_I) can be determined using the following equation:

$$\delta_I = \frac{-m_{LMWL} + m_{\delta_E - \delta^*}}{\beta_{LMWL} - \beta_{\delta_E - \delta^*}}$$

where m and β are the slope and intercept of the LMWL and $\delta_E - \delta^*$ lines, respectively.

Note isotope equations listed above (steps 1-3) are expressed in decimal notation.

Step 4: Calculate lake evaporation-to-inflow ratios

Evaporation-to-inflow (E/I) ratios were determined using the equation reported by Gibson and Edwards (2002):

$$E/I = \frac{(\delta_I - \delta_L)}{(\delta_E - \delta_L)}$$

Consistent with Remmer et al. (2020a), we set E/I ratios to a maximum of 1.5 for lakes experiencing strong non-steady-state conditions.

Appendix 2 | Supplementary material for Chapter 4

This appendix contains isotope mass-balance equations associated with **Chapter 4**, as well as supplementary tables and figures for the manuscript in preparation.

A2.1 Isotope framework and stable isotope mass balance modelling of pond water balance in the Whooping Crane Summer Range

A series of isotope mass-balance equations were used to estimate evaporation-to-inflow ratios at waterbodies sampled across the Sass-Klewi Nesting Area in May, August and September 2022. The isotope mass-balance model parameters are reported in Table A2.2, and the equations are listed below. Note, equations A4.7-A4.17 are formulated for parameters expressed in decimal notation.

Shallow waterbodies at high latitudes are typically ice-covered for at least half of the year. Because of this, influential meteorological parameters used in isotope mass-balance modelling (i.e., temperature and relative humidity) must be flux-weighted to reflect the open-water period, when waterbodies interact with the atmosphere (Gibson, 2002). Hourly relative humidity and temperature data were obtained at <https://climate.weather.gc.ca> to determine the flux-weighted relative humidity and temperature for 2022.

EQ.A2.1

$$h_{FW} = \sum_{i=1}^n \frac{(h_i \times E_i)}{E_i}$$

EQ.A2.2

$$T_{FW} = \sum_{i=1}^n \frac{(T_i \times E_i)}{E_i}$$

where h_i and T_i are hourly relative humidity and temperature measurements, respectively, E_i is monthly potential evapotranspiration and n represents the number of months.

For each month during the open-water season (May–September), potential evapotranspiration was estimated using the equation by Thornthwaite (1948):

EQ.A2.3

$$E_i = 1.6 \times \frac{L}{12} \times \frac{N}{30} \left[\frac{10 \times T}{I} \right]^a$$

where L is the average day length for the month expressed in hours, N is the number of days in the month, T is the average monthly temperature, I is the ice-free season heat index, and a is a coefficient calculated using **EQ.A4.6**.

EQ.A2.4

$$L = \frac{sun}{N}$$

EQ.A2.5

$$I = \sum_{i=1}^n \left[\frac{T}{5} \right]^{1.5}$$

EQ.A2.6

$$a = 0.49 + 0.0179 \times I - 7.71 \times 10^{-5} \times I^2 + 6.75 \times 10^{-7} \times I^3$$

We obtained the total hours of sunshine (*sun*) for each month using National Research Council of Canada's sunrise/sunset calculator (<http://www.nrc-cnrc.gc.ca/eng/services/sunrise/>) using the following inputs: Civil Twilight (sunrise/sunset, full year); Lat and Long: 60°12' N, -113°11 W; Mountain Standard Time (Canada & US) (UTC-7:00).

The equilibrium fractionation (α^*), equilibrium separation (ϵ^*) and kinetic separation (ϵ_k) factors were determined for oxygen and hydrogen independently using equations by Horita and Wesolowski (1994) and Gonfiantini (1986):

$$\text{EQ.A2.7} \quad 1000 \times \ln(\alpha_{oxygen}^*) = -7.685 + 6.7123 \left(\frac{10^3}{T_{FW}} \right) - 1.6664 \left(\frac{10^6}{T_{FW}^2} \right) + 0.35041 \left(\frac{10^9}{T_{FW}^3} \right)$$

$$\text{EQ.A2.8} \quad 1000 \times \ln(\alpha_{hydrogen}^*) = 1115.8 \left(\frac{T_{FW}^3}{10^9} \right) - 1620.1 \left(\frac{T_{FW}^2}{10^6} \right) + 794.84 \left(\frac{T_{FW}}{10^3} \right) - 161.04 + 2.9992 \left(\frac{10^9}{T_{FW}^3} \right)$$

EQ.A2.9

$$\epsilon^* = \alpha^* - 1$$

EQ.A2.10

$$\epsilon_{k-oxygen} = 0.0142$$

EQ.A2.11

$$\epsilon_{k-hydrogen} = 0.0125$$

where T_{FW} represents the evaporation flux-weighted temperature expressed in Kelvin and h_{FW} represents the evaporation flux-weighted relative humidity.

The isotope composition of ice-free season atmospheric moisture (δ_{AS}) is difficult to measure directly but can be estimated using various approaches. An evaporation pan was not feasible, so initially we modelled this parameter under the assumption of isotopic equilibrium with evaporation-flux-weighted local precipitation (δ_{PS}).

EQ.A2.12

$$\delta_{AS} = \frac{(\delta_{PS} - \epsilon^*)}{\alpha^*}$$

where δ_{PS} ($\delta^{18}O$: -15.8 ‰, δ^2H : -126.5 ‰) represents the average isotope composition of summer precipitation (May-September) collected at Fort Smith, NWT from 1960-1969 (IAEA/WMO, n.d.). The resulting value of δ_{AS} derived from assuming equilibrium with δ_{PS} generated a Local Evaporation Line (LEL) with ‘poor fit’ to the measured waterbody isotope compositions. Therefore, we further constrained the δ_{AS} parameter until it resulted in an LEL slope that was near equivalent to the slope of a linear regression through water isotope data ($m = 4.3$) collected at the 63 ponds in May, August and September 2022. To achieve this, we adjusted δ_{AS} derived using **EQ.A4.12** by 0.95‰ for $\delta^{18}O$ and 7.6‰ for δ^2H . A similar approach has been used elsewhere (Wolfe et al., 2011; Gibson and Reid, 2014; Wan et al., 2020).

The isotope composition of a terminal lake in “steady state” (δ_{SSL}) was determined using the classic linear resistance model of Craig and Gordon (1965) and the equation provided by Gonfiantini (1986):

$$\mathbf{EQ.A2.1} \quad \delta_{SSL} = \alpha^* \delta_P (1 - h_{FW} + \varepsilon_k) + (\alpha^* \times h \delta_{AS}) + (\alpha^* \times \varepsilon_k) + \varepsilon^*$$

where δ_P represents the mean annual isotope composition of precipitation collected in Fort Smith, NWT from 1960-1969 (IAEA/WMO, n.d.).

The limiting isotope composition (δ^*) on a LEL represents the theoretical endpoint of isotopic enrichment of a waterbody before desiccation, and was determined using Gonfiantini (1986):

$$\mathbf{EQ.A2.14} \quad \delta^* = \frac{h_{FW} \times \delta_{AS} + \varepsilon_k + \frac{\varepsilon^*}{\alpha^*}}{h_{FW} - \varepsilon_k - \frac{\varepsilon^*}{\alpha^*}}$$

The LEL ($\delta^2H = 4.3 \times \delta^{18}O - 69.6$) for the study region was then constructed between δ_P and δ^* , with δ_{SSL} positioned along the LEL (see Figure 4.3a in **Chapter 4**).

A coupled-isotope tracer approach (Yi et al., 2008) was used to estimate the isotope composition of input water, δ_i , for each pond for each sampling episode in 2022. This value was estimated from the intersection of each pond’s evaporation line (δ_E through δ_L to δ^* , where δ_L is the pond water isotope composition) with the Local Meteoric Water Line (LMWL),

assuming input water is primarily meteoric water that has not undergone evaporation. δ_E was determined using the Craig and Gordon (1965) model and equation provided by Gonfiantini (1986):

EQ.A2.15
$$\delta_E = \frac{\left[\frac{(\delta_L - \varepsilon^*)}{\alpha^* - (h_{FW} \times \delta_{AS}) - \varepsilon_k} \right]}{(1 - h_{FW} + \varepsilon_k)}$$

The intersection between $\delta_E - \delta^*$ and the LMWL (δ_I) was determined using the following equation:

EQ.A2.16
$$\delta_I = \frac{-m_{LMWL} + m_{\delta_E - \delta^*}}{\beta_{LMWL} - \beta_{\delta_E - \delta^*}}$$

where m and β are the slope and intercept of the LMWL and $\delta_E - \delta^*$ lines, respectively. The LMWL was defined using the isotope composition of precipitation collected at Fort Smith, NWT during 1960-1969 ($\delta^2\text{H} = 6.7 \times \delta^{18}\text{O} - 19.2$) by the Global Network for Isotopes in Precipitation (IAEA/WMO, n.d.; available at <https://www.iaea.org/services/networks/gnip>).

Evaporation-to-inflow (E/I) ratios were then estimated for each pond and nesting wetland for each sampling episode (Gibson and Edwards, 2002):

EQ.A2.17
$$E/I = \frac{(\delta_I - \delta_L)}{(\delta_E - \delta_L)}$$

We assign a maximum E/I ratio of 1.5 for ponds experiencing strong, non-steady-state evaporation because of increasing uncertainty of model estimates beyond this value.

A2.2 Chapter 4 supplementary tables

Table A2.1. Coordinates, elevation and surface area of ponds (SK prefix) and river (R prefix) sampling sites in the Sass-Klewi Nesting Area. Metrics that aided in site selection, including pond colour, relative fetch and surface area, were determined using satellite imagery acquired in 2021.

Site ID	Coordinates (DMS)		Coordinates (Decimal)		Pond colour	Estimated fetch (m)	^a Elevation (masl)	^b Surface Area (m ²)
	Latitude	Longitude	Latitude	Longitude				
SK1	60° 22' 37.0"	-113° 14' 60.0"	60.377	-113.250	medium brown	110	205.38	6974
SK2	60° 22' 16.9"	-113° 08' 30.5"	60.371	-113.141	light green	224	203.65	31750
SK3	60° 20' 54.1"	-113° 11' 35.5"	60.348	-113.193	dark green	166	205.4	20972
SK4	60° 22' 43.1"	-113° 14' 58.7"	60.348	-113.273	dark green	105	207.61	7430
SK5	60° 20' 19.7"	-113° 18' 44.6"	60.339	-113.312	medium green	173	209.04	15863
SK6	60° 20' 06.9"	-113° 21' 00.8"	60.335	-113.350	medium green	111	209.89	13203
SK7	60° 20' 08.8"	-113° 23' 34.8"	60.336	-113.393	medium green	872	210.17	771979
SK8	60° 20' 57.2"	-113° 23' 16.7"	60.349	-113.388	green brown	51	210.65	4268
SK9	60° 20' 32.9"	-113° 24' 31.0"	60.345	-113.410	green brown	674	210.04	1198339
SK10	60° 20' 39.1"	-113° 25' 44.6"	60.344	-113.429	dark brown	114	209.98	20571
SK11	60° 20' 23.1"	-113° 26' 31.3"	60.340	-113.442	light green	161	210.92	18397
SK12	60° 19' 26.4"	-113° 26' 58.0"	60.324	-113.450	light green	149	210.99	13527
SK13	60° 19' 05.6"	-113° 24' 20.2"	60.318	-113.406	dark green	137	209.53	10076
SK14	60° 18' 44.3"	-113° 21' 51.6"	60.312	-113.364	light green	158	210.3	19639
SK15	60° 19' 16.1"	-113° 18' 42.5"	60.321	-113.311	light green	209	209.58	11296
SK16	60° 19' 06.2"	-113° 16' 08.1"	60.318	-113.269	medium brown	97	208.68	5497
SK17	60° 19' 25.3"	-113° 12' 20.2"	60.324	-113.206	cream	742	207.64	539600
SK18	60° 18' 28.1"	-113° 13' 18.8"	60.308	-113.222	medium brown	399	209.09	140716
SK19	60° 17' 18.6"	-113° 15' 33.9"	60.288	-113.261	dark brown	195	211.49	22155
SK20	60° 18' 02.9"	-113° 18' 46.3"	60.301	-113.314	medium green	161	209.26	22216
SK21	60° 17' 37.9"	-113° 20' 01.8"	60.293	-113.333	light green	282	208.67	66879
SK22	60° 18' 03.7"	-113° 21' 15.7"	60.301	-113.354	dark brown	159	209.83	11229
SK23	60° 17' 08.7"	-113° 21' 03.7"	60.286	-113.351	green brown	687	210.18	262903
SK24	60° 17' 24.9"	-113° 22' 57.1"	60.290	-113.382	light green	116	210.63	9607
SK25	60° 17' 58.7"	-113° 26' 05.1"	60.298	-113.431	dark green	65	211.58	4169
SK26	60° 17' 26.7"	-113° 27' 29.7"	60.291	-113.458	dark green	72	211.12	4393
SK27	60° 16' 44.2"	-113° 23' 42.7"	60.280	-113.398	cream	36	210.64	1248
SK28	60° 15' 56.2"	-113° 24' 44.9"	60.266	-113.412	dark green	82	215.23	4327
SK29	60° 13' 43.9"	-113° 26' 30.1"	60.229	-113.439	cream	112	224.83	5590
SK30	60° 15' 53.2"	-113° 27' 21.0"	60.265	-113.455	green brown	71	214.17	1950
SK31	60° 15' 15.4"	-113° 24' 59.3"	60.254	-113.417	green brown	262	215.16	37863
SK32	60° 15' 43.4"	-113° 21' 41.7"	60.261	-113.365	medium green	70	215.5	5209
SK33	60° 14' 14.3"	-113° 18' 32.3"	60.237	-113.310	medium brown	220	216.26	25878
SK34	60° 14' 16.1"	-113° 16' 54.5"	60.238	-113.282	green brown	400	214.87	57861
SK35	60° 16' 03.2"	-113° 13' 11.2"	60.267	-113.221	medium brown	133	211.96	12206
SK36	60° 15' 18.7"	-113° 13' 02.6"	60.255	-113.218	green brown	46	212.51	2869
SK37	60° 14' 04.2"	-113° 13' 53.4"	60.234	-113.231	medium brown	112	213.29	14266
SK38	60° 12' 45.8"	-113° 16' 16.4"	60.213	-113.272	medium brown	138	215.13	9679
SK39	60° 12' 53.4"	-113° 21' 32.6"	60.216	-113.357	green brown	28	224.31	5269
SK40	60° 12' 05.5"	-113° 24' 09.2"	60.204	-113.405	green brown	269	226.86	46035
SK41	60° 11' 23.3"	-113° 22' 50.1"	60.190	-113.380	dark brown	108	229.48	9167
SK42	60° 09' 38.3"	-113° 21' 02.6"	60.161	-113.351	medium green	262	230.44	69632
SK43	60° 08' 33.3"	-113° 21' 53.1"	60.142	-113.365	dark brown	89	235.13	5540
SK44	60° 11' 01.7"	-113° 16' 27.7"	60.184	-113.274	light green	159	218.29	20638
SK45	60° 09' 58.1"	-113° 15' 46.6"	60.166	-113.263	medium green	114	221.52	5962
SK46	60° 10' 42.8"	-113° 14' 14.3"	60.178	-113.238	light green	103	217.16	6370
SK47	60° 12' 14.5"	-113° 13' 46.9"	60.205	-113.232	green brown	87	215.02	6907
SK48	60° 10' 53.5"	-113° 10' 41.7"	60.180	-113.180	medium green	660	216.41	189796
SK49	60° 11' 38.5"	-113° 10' 00.3"	60.195	-113.168	medium green	245	215.88	20909
SK50	60° 12' 23.3"	-113° 10' 34.6"	60.211	-113.178	medium green	82	214.62	6885
SK51	60° 13' 07.6"	-113° 08' 58.7"	60.219	-113.150	dark brown	142	214.34	16947
SK52	60° 13' 50.9"	-113° 08' 29.9"	60.231	-113.143	medium brown	170	212.65	16867
SK53	60° 11' 52.0"	-113° 08' 00.6"	60.198	-113.134	medium brown	233	216.23	22625
SK54	60° 10' 38.0"	-113° 07' 17.7"	60.177	-113.121	medium green	111	217.9	8518
SK55	60° 09' 43.6"	-113° 05' 55.4"	60.162	-113.099	medium green	83	220.08	5303
SK56	60° 08' 49.3"	-113° 06' 40.6"	60.147	-113.112	medium brown	149	221.78	16309

SK57	60° 09' 45.3"	-113° 09' 43.2"	60.163	-113.162	medium green	311	218.7	40661
SK58	60° 09' 44.6"	-113° 12' 32.4"	60.162	-113.209	green brown	125	222.67	5409
SK59	60° 07' 27.8"	-113° 17' 49.8"	60.124	-113.296	green brown	715	238.42	340847
SK60	60° 06' 32.0"	-113° 16' 41.3"	60.109	-113.278	green brown	200	239.99	13298
SK61	60° 14' 09.8"	-113° 22' 06.0"	60.236	-113.367	cream	162	225.62	14257
SK62	60° 13' 50.2"	-113° 27' 37.6"	60.230	-113.460	light green	33	224.62	1144
SK63	60° 13' 08.4"	-113° 18' 57.7"	60.219	-113.316	green brown	37	217.61	1317
RP	60° 14' 19.9"	-113° 10' 20.8"	60.228	-113.198				
RKU	60° 18' 13.6"	-113° 22' 14.8"	60.304	-113.371				
RKD	60° 22' 16.1"	-113° 09' 02.9"	60.375	-113.140				
RSD	60° 14' 20.1"	-113° 10' 20.3"	60.241	-113.172				

^aElevation for each site was extracted from Canada's High Resolution Digital Elevation Model (HRDEM) available online at: <https://open.canada.ca/data/en/dataset/957782bf-847c-4644-a757-e383c0057995>.

^bSurface area of each site was calculated as polygon areas from individual catchments that were delineated using a high-resolution satellite image of the WCNR acquired in 2021 provided by John Conkin (ECCC-CWS) in ArcMap V10.8.2.

Table A2.2. Isotope mass-balance model parameters for 2022.

Parameter	Units	Value	
T_{flux}	(°C)	15.87	
rh_{flux}	(%)	60.39	
		¹⁸ O	² H
α^*		1.0106	1.0893
ε^*	(‰)	10.16	89.33
ε_K	(‰)	5.62	4.95
δ_{AS}	(‰)	-25.6	-199
δ_{AS}^a	(‰)	-26.5	-205.7
δ_{SSL}	(‰)	-8.5	-108.7
δ^*	(‰)	-0.6	-72.1
δ_P	(‰)	-20.1	-155.9
δ_{PS}	(‰)	-15.7	-126.5
LEL Slope		4.3	
LEL Intercept		-69.6	

Table A2.3. Measured water isotope compositions for sites sampled in the Sass-Klewi Nesting Area in May, August and September 2022.

Site ID	May		August		September	
	$\delta^{18}\text{O}$ (‰)	$\delta^2\text{H}$ (‰)	$\delta^{18}\text{O}$ (‰)	$\delta^2\text{H}$ (‰)	$\delta^{18}\text{O}$ (‰)	$\delta^2\text{H}$ (‰)
SK1	-17.79	-148.75	-11.20	-121.33	-11.41	-120.26
SK2	-14.50	-130.18	-8.23	-103.37	-6.99	-98.17
SK3	-17.98	-149.32	-12.86	-125.83	-11.76	-121.92
SK4	-17.44	-145.38	-9.81	-113.75	-8.80	-108.43
SK5	-17.50	-146.99	-12.48	-123.42	-11.08	-118.19
SK6	-17.81	-147.38	-11.97	-121.75	-10.72	-117.68
SK7	-18.40	-150.77	-12.60	-123.45	-11.19	-117.81
SK8	-16.91	-143.96	-10.62	-116.87	-8.77	-110.60
SK9	-18.40	-150.90	-11.08	-118.27	-9.93	-112.69
SK10	-18.62	-151.76	-11.11	-119.66	-9.64	-112.67
SK11	-14.30	-131.27	-6.24	-99.81	-6.52	-99.35
SK12	-15.94	-140.53	-7.49	-105.96	-6.23	-99.83
SK13	-18.76	-151.58	-13.88	-130.12	-13.49	-129.02
SK14	-18.46	-150.88	-10.57	-116.86	-9.50	-111.76
SK15	-18.52	-151.39	-11.52	-121.12	-10.29	-113.04
SK16	-17.87	-148.55	-9.49	-114.86	-7.99	-106.22
SK17	-15.91	-137.73	-8.62	-110.91	-8.76	-104.92
SK18	-16.82	-142.59	-10.86	-118.84	-8.21	-103.41
SK19	-16.94	-144.09	-8.19	-108.31	-7.02	-100.17
SK20	-18.19	-150.67	-9.42	-111.32	-9.61	-110.56
SK21	-17.70	-147.92	-10.44	-115.85	-9.12	-106.52
SK22	-17.64	-147.62	-10.62	-118.52	-9.17	-110.70
SK23	-17.79	-148.21	-7.43	-104.74	-6.62	-97.66
SK24	-18.64	-151.18	-15.44	-138.83	-15.42	-137.20
SK25	-18.69	-152.23	-15.73	-140.03	-15.55	-136.67
SK26	-18.68	-152.06	-16.25	-141.95	-16.23	-139.21

SK27	-14.35	-133.43	-7.96	-108.53	no water to sample	
SK28	-17.31	-147.71	-8.08	-108.30	-6.33	-96.00
SK29	-19.78	-158.00	-19.79	-158.54	-20.07	-157.05
SK30	-17.52	-145.30	-11.17	-121.52	-10.82	-117.65
SK31	-16.53	-143.27	-8.39	-109.66	-6.94	-101.73
SK32	-15.38	-137.02	-7.27	-105.72	-5.89	-98.26
SK33	-17.98	-149.16	-10.10	-116.67	-8.69	-110.45
SK34	-17.73	-147.98	-9.41	-112.90	-7.75	-102.81
SK35	-16.29	-139.99	-8.05	-106.36	-6.69	-98.66
SK36	-17.14	-143.56	-9.82	-115.19	-8.37	-106.65
SK37	-17.05	-144.34	-8.20	-109.10	-7.13	-99.71
SK38	-16.27	-139.76	-9.55	-113.74	-8.56	-105.57
SK39	-17.37	-146.42	-10.84	-119.79	-9.49	-110.20
SK40	-17.18	-144.81	-10.77	-118.07	-9.69	-111.07
SK41	-16.50	-141.05	-10.07	-114.77	-8.17	-101.86
SK42	-18.84	-153.73	-15.80	-140.07	-15.61	-136.92
SK43	-14.87	-132.12	-9.97	-112.30	-9.33	-109.30
SK44	-18.38	-149.33	-15.45	-136.91	-14.41	-133.02
SK45	-18.46	-150.52	-14.06	-131.76	-11.56	-121.84
SK46	-14.62	-133.37	-9.83	-114.00	-8.72	-111.32
SK47	-17.08	-145.76	-11.96	-124.61	-10.44	-119.77
SK48	-18.04	-148.67	-15.68	-136.35	-14.70	-132.81
SK49	-17.82	-147.92	-14.71	-132.58	-13.69	-127.25
SK50	-17.56	-146.54	-13.84	-129.35	-13.20	-126.37
SK51	-18.31	-151.87	-13.21	-126.87	-11.42	-118.44
SK52	-17.86	-149.50	-11.56	-119.79	-9.76	-111.76
SK53	-18.33	-149.92	-11.60	-121.20	-9.18	-109.78
SK54	-16.61	-143.77	-7.53	-105.35	-6.37	-97.42
SK55	-14.38	-132.17	-7.41	-104.39	-6.38	-96.76
SK56	-18.58	-149.59	-10.59	-117.38	-8.91	-106.68
SK57	-16.35	-140.99	-8.75	-108.69	-7.59	-100.29
SK58	-18.49	-151.00	-14.34	-133.63	-10.93	-118.06
SK59	-14.61	-128.79	-9.90	-111.61	-8.74	-104.43
SK60	-17.10	-142.97	-10.21	-115.74	-8.43	-105.39
SK61	-16.99	-145.91	-10.42	-120.27	-9.39	-110.54
SK62	-16.11	-139.64	-8.19	-110.68	-8.26	-108.97
SK63	-17.46	-146.97	-9.66	-116.97	-7.08	-102.95
RP	-18.28	-149.94	-13.74	-128.14	-13.37	-126.27
RKU	-18.49	-150.22	-13.96	-131.48	-12.95	-124.96
RSD	-18.40	-148.86	-14.31	-131.61	-12.38	-121.83
RKD	-18.34	-149.33	-12.90	-125.64		
Nest04	-20.06	-157.60	-11.84	-119.05		
Nest05	-18.73	-151.31	-14.07	-130.64		
Nest07	-19.27	-155.54	-11.93	-123.40		
Nest10	-19.40	-154.35	-15.78	-139.03		

Table A2.4. Calculated δ_i (input isotope composition) and E/I ratios for sampling sites in the Sass-Klewi Nesting Area in May, August and September 2022. Also included is the E/I-defined groundwater connectivity category.

Site ID	May-22			Aug-22			Sep-22			E/I Category
	δ_i (^{18}O)	δ_i (^2H)	E/I	δ_i (^{18}O)	δ_i (^2H)	E/I	δ_i (^{18}O)	δ_i (^2H)	E/I	
SK1	-21.51	-163.35	0.15	-21.53	-163.48	0.66	-20.24	-154.81	0.56	moderate
SK2	-19.07	-146.95	0.22	-17.58	-136.99	0.83	-17.28	-134.97	1.10	weak
SK3	-21.43	-162.81	0.14	-20.04	-153.48	0.40	-20.37	-155.68	0.53	strong
SK4	-20.69	-157.79	0.13	-20.37	-155.67	0.78	-19.56	-150.29	0.89	moderate
SK5	-21.26	-161.63	0.15	-19.57	-150.33	0.41	-19.80	-151.88	0.57	strong
SK6	-20.91	-159.30	0.12	-19.77	-151.68	0.47	-20.43	-156.08	0.65	strong
SK7	-21.35	-162.21	0.11	-19.34	-148.79	0.38	-19.31	-148.60	0.52	strong
SK8	-20.96	-159.66	0.17	-20.17	-154.34	0.65	-21.50	-163.24	1.06	weak
SK9	-21.40	-162.57	0.11	-19.85	-152.21	0.57	-19.29	-148.43	0.68	moderate
SK10	-21.39	-162.54	0.11	-20.66	-157.60	0.62	-20.10	-153.89	0.79	moderate
SK11	-19.90	-152.51	0.28	-22.45	-169.61	1.96	-20.20	-154.56	1.57	weak

SK12	-21.17	-161.03	0.23	-22.80	-171.95	1.51	-22.55	-170.27	1.97	weak
SK13	-21.10	-160.60	0.09	-20.14	-154.17	0.32	-20.38	-155.76	0.36	strong
SK14	-21.29	-161.87	0.11	-20.30	-155.23	0.67	-19.87	-152.32	0.79	moderate
SK15	-21.40	-162.61	0.11	-20.49	-156.49	0.56	-18.61	-143.92	0.59	moderate
SK16	-21.30	-161.88	0.14	-22.44	-169.53	0.99	-20.63	-157.42	1.17	weak
SK17	-19.98	-153.06	0.18	-22.46	-169.69	1.18	-17.28	-135.00	0.71	moderate
SK18	-20.53	-156.77	0.16	-20.82	-158.67	0.66	-17.66	-137.52	0.85	moderate
SK19	-20.97	-159.69	0.17	-21.81	-165.32	1.22	-18.86	-145.56	1.26	weak
SK20	-21.65	-164.26	0.13	-19.79	-151.80	0.80	-18.74	-144.78	0.69	moderate
SK21	-21.31	-162.01	0.14	-19.99	-153.13	0.66	-17.43	-135.98	0.66	moderate
SK22	-21.29	-161.83	0.15	-21.32	-162.05	0.73	-20.11	-153.97	0.87	moderate
SK23	-21.29	-161.82	0.14	-21.69	-164.49	1.42	-18.07	-140.28	1.30	weak
SK24	-21.13	-160.78	0.09	-21.33	-162.10	0.27	-20.59	-157.16	0.24	strong
SK25	-21.47	-163.06	0.10	-21.34	-162.15	0.25	-20.12	-154.03	0.21	strong
SK26	-21.42	-162.71	0.10	-21.25	-161.55	0.22	-20.08	-153.74	0.17	strong
SK27	-20.85	-158.92	0.32	-23.23	-174.85	1.41				weak
SK28	-21.91	-166.02	0.19	-22.33	-168.80	1.30	-17.62	-137.28	1.34	weak
SK29	-22.05	-166.94	0.08	-22.25	-168.29	0.09	-21.25	-161.58	0.04	strong
SK30	-20.53	-156.74	0.12	-21.76	-165.02	0.68	-20.14	-154.11	0.62	moderate
SK31	-21.33	-162.13	0.21	-22.26	-168.35	1.21	-20.79	-158.47	1.49	weak
SK32	-20.58	-157.10	0.24	-23.86	-179.06	1.69	-22.62	-170.75	2.15	weak
SK33	-21.37	-162.36	0.13	-21.65	-164.23	0.83	-21.69	-164.55	1.10	weak
SK34	-21.29	-161.84	0.14	-21.04	-160.19	0.90	-18.55	-143.52	1.03	weak
SK35	-20.31	-155.28	0.17	-20.52	-156.65	1.14	-18.73	-144.69	1.35	weak
SK36	-20.42	-156.02	0.14	-21.45	-162.90	0.86	-19.58	-150.39	0.98	moderate
SK37	-20.89	-159.16	0.16	-22.60	-170.60	1.29	-18.07	-140.27	1.14	weak
SK38	-20.25	-154.85	0.17	-21.23	-161.45	0.89	-18.20	-141.13	0.82	moderate
SK39	-21.24	-161.48	0.16	-21.55	-163.60	0.71	-18.82	-145.30	0.72	moderate
SK40	-20.87	-159.04	0.15	-20.55	-156.90	0.66	-18.87	-145.60	0.69	moderate
SK41	-20.41	-155.96	0.17	-20.31	-155.28	0.74	-16.76	-131.48	0.77	moderate
SK42	-21.84	-165.54	0.11	-21.22	-161.36	0.24	-20.13	-154.08	0.21	strong
SK43	-19.29	-148.47	0.21	-18.94	-146.07	0.65	-18.65	-144.16	0.73	moderate
SK44	-20.80	-158.58	0.09	-20.41	-155.91	0.23	-20.53	-156.73	0.30	strong
SK45	-21.15	-160.90	0.10	-20.60	-157.24	0.33	-20.84	-158.83	0.58	strong
SK46	-20.29	-155.14	0.28	-20.49	-156.48	0.79	-22.41	-169.36	1.15	weak
SK47	-21.45	-162.95	0.18	-21.54	-163.49	0.57	-22.96	-173.02	0.87	moderate
SK48	-21.07	-160.36	0.12	-19.77	-151.67	0.19	-19.88	-152.42	0.25	strong
SK49	-21.12	-160.68	0.13	-19.76	-151.61	0.24	-19.14	-147.41	0.28	strong
SK50	-20.97	-159.69	0.14	-19.85	-152.17	0.31	-19.62	-150.67	0.35	strong
SK51	-21.96	-166.31	0.14	-19.85	-152.22	0.36	-19.15	-147.53	0.49	strong
SK52	-21.72	-164.70	0.15	-19.60	-150.51	0.50	-19.13	-147.38	0.70	moderate
SK53	-21.11	-160.66	0.11	-20.33	-155.44	0.54	-19.40	-149.16	0.81	moderate
SK54	-21.41	-162.67	0.20	-21.86	-165.65	1.41	-18.82	-145.31	1.47	weak
SK55	-20.17	-154.36	0.29	-21.40	-162.56	1.40	-18.14	-140.72	1.38	weak
SK56	-20.61	-157.27	0.08	-20.60	-157.22	0.68	-18.02	-139.96	0.75	moderate
SK57	-20.64	-157.47	0.19	-19.93	-152.75	0.93	-17.15	-134.10	0.93	moderate
SK58	-21.29	-161.87	0.11	-20.98	-159.75	0.33	-20.11	-153.91	0.61	strong
SK59	-18.33	-142.02	0.18	-18.68	-144.35	0.64	-17.04	-133.34	0.69	moderate
SK60	-20.25	-154.84	0.13	-20.58	-157.09	0.74	-18.43	-142.71	0.87	moderate
SK61	-21.69	-164.50	0.20	-23.48	-176.54	0.91	-19.32	-148.67	0.77	moderate
SK62	-20.46	-156.30	0.19	-24.53	-183.58	1.47	-22.16	-167.68	1.24	weak
SK63	-21.32	-162.03	0.16	-23.71	-178.06	1.06	-21.46	-162.99	1.51	weak
Nest04	-21.47	-163.04	0.05	-18.61	-143.89	0.41				strong
Nest05	-21.04	-160.20	0.09	-20.03	-153.39	0.30				strong
Nest07	-21.88	-165.79	0.10	-20.84	-158.84	0.54				moderate
Nest10	-21.21	-161.32	0.07	-20.77	-158.37	0.22				strong
RP	-21.20	-161.24	0.11	-19.46	-149.56	0.30	-19.25	-148.19	0.31	River
RKU	-20.98	-159.79	0.10	-20.67	-157.68	0.34	-19.41	-149.25	0.36	River
RSD	-20.59	-157.17	0.08	-20.04	-153.46	0.28	-18.96	-146.23	0.38	River
RKD	-20.86	-158.99	0.10	-19.86	-152.24	0.39				River

Table A2.5. Water depth and drawdown metrics for sampling sites in Sass-Klewi Nesting Area in 2022.

Site ID	Mean (m)	Initial (m)	Minimum (m)	Maximum (m)	^a Drawdown (cm)		^b Drawdown magnitude (%)	
					June-August	June-September	June-August	June-September
SK1	0.49	0.62	0.40	0.62	-16.20	-22.1	26.1	35.6
SK2	0.64	0.82	0.48	0.82	-23.7	-34.2	28.9	41.7
SK3	0.95	1.09	0.80	1.10	-19	-29.0	17.4	26.6
SK4	0.85	0.99	0.71	1.00	-20.8	-28.1	21.0	28.4
SK5	0.95	1.08	0.81	1.09	-17.8	-27.5	16.5	25.5
SK6	0.84	0.95	0.76	0.97	-14.5	-19.0	15.3	20.0
SK7	0.77	0.88	0.64	0.89	-15.4	-19.3	17.5	21.9
SK8	0.73	0.89	0.60	0.89	-20.4	-29.1	22.9	32.7
SK9	0.83	1.03	0.67	1.03	-25.4	-35.8	24.7	34.8
SK10	0.55	0.76	0.38	0.76	-28	-38.0	36.8	50.0
SK11	0.35	0.48	0.23	0.48	-16.7	-24.6	34.8	51.3
SK12	0.42	0.57	0.29	0.57	-19.9	-28.5	34.9	50.0
SK13	1.37	1.51	1.22	1.52	-19.5	-29.1	12.9	19.3
SK14	0.65	0.93	0.51	0.94	-36.1	-42.7	38.8	45.9
SK15	0.77	0.95	0.62	0.95	-25.3	-28.9	26.6	30.4
SK16	0.69	0.89	0.57	0.89	-23.3	-31.7	26.2	35.6
SK17	0.89	1.07	0.74	1.07	-21.6	-33.0	20.2	30.8
SK18	0.60	0.76	0.47	0.76	-19.8	-28.6	26.1	37.6
SK19	0.48	0.61	0.34	0.62	-17.6	-26.7	28.9	43.8
SK20	0.61	0.84	0.48	0.84	-28.2	-35.6	33.6	42.4
SK21	0.69	0.88	0.55	0.88	-25.5	-33.0	29.0	37.5
SK22	0.86	1.03	0.73	1.03	-21.9	-30.4	21.3	29.5
SK23	0.42	0.63	0.27	0.63	-26.4	-35.8	41.9	56.8
SK24	0.55	0.66	0.47	0.67	-13.3	-18.4	20.2	27.9
SK25	0.95	1.11	0.84	1.12	-21.1	-27.5	19.0	24.8
SK26	0.72	0.82	0.63	0.82	-13.3	-19.1	16.2	23.3
SK27	0.25	0.42	0.00	0.42	-29.1	-41.6	69.3	99.0
SK28	0.47	0.66	0.30	0.66	-24.5	-36.0	37.1	54.5
SK29	0.48	0.50	0.47	0.51	-1.4	-3.1	2.8	6.2
SK30	0.28	0.40	0.19	0.41	-17.4	-20.8	43.5	52.0
SK31	0.70	0.84	0.57	0.84	-19.4	-26.6	23.1	31.7
SK32	0.56	0.73	0.42	0.74	-22.6	-31.6	31.0	43.3
SK33	0.65	0.72	0.56	0.74	-9.6	-16.4	13.3	22.8
SK34		did not retrieve logger						
SK35		did not retrieve logger						
SK36	0.63	0.78	0.51	0.79	-19.8	-27.7	25.4	35.5
SK37	0.54	0.71	0.40	0.72	-22.3	-31.7	31.4	44.6
SK38		did not retrieve logger						
SK39	0.57	0.71	0.47	0.71	-16.8	-24.1	23.7	33.9
SK40	0.60	0.68	0.51	0.69	-9.1	-17.2	13.4	25.3
SK41	0.63	0.72	0.48	0.74	-12.8	-24.0	17.8	33.3
SK42	0.45	0.58	0.36	0.58	-12.8	-21.7	22.1	37.4
SK43	0.89	1.03	0.81	1.03	-16	-22.6	15.5	21.9
SK44	0.58	0.64	0.50	0.64	-6.8	-13.9	10.6	21.7
SK45	0.56	0.63	0.47	0.63	-8.3	-15.4	13.2	24.4
SK46*	0.47	0.50	0.40	0.50				
SK47	0.33	0.41	0.26	0.41	-10.1	-15.3	24.6	37.3
SK48	0.62	0.69	0.54	0.71	-9.9	-15.3	14.3	22.2
SK49	0.75	0.82	0.69	0.82	-8.5	-13.5	10.4	16.5
SK50	0.86	0.94	0.80	0.95	-10.8	-13.6	11.5	14.5
SK51	0.73	0.84	0.63	0.85	-14.3	-20.9	17.0	24.9
SK52	0.56	0.70	0.46	0.70	-17.5	-23.5	25.0	33.6
SK53	0.67	0.81	0.57	0.82	-18.5	-24.9	22.8	30.7
SK54	0.48	0.63	0.38	0.63	-20	-24.6	31.7	39.0
SK55	0.50	0.67	0.39	0.67	-21.8	-27.2	32.5	40.6
SK56	0.46	0.60	0.35	0.60	-18.6	-25.6	31.0	42.7
SK57	0.46	0.54	0.39	0.55	-12.1	-15.4	22.4	28.5
SK58	0.53	0.58	0.38	0.63	-6.3	-19.9	10.9	34.3
SK59	0.69	0.81	0.58	0.83	-14.4	-22.4	17.8	27.7

SK60	0.56	0.68	0.42	0.70	-15.5	-26.1	22.8	38.4
SK61	0.28	0.29	0.24	0.33	-1.3	-4.7	4.5	16.2
SK62	0.39	0.55	0.28	0.55	-21.8	-27.2	39.6	49.5
SK63	0.23	0.36	0.10	0.36	-16.9	-26.4	46.9	73.3
RKD	0.41	0.59	0.27	0.60	-25.3	-33.4	42.9	56.6
Nest04	0.09	0.24	0.00	0.24	-23.1			
Nest05	0.01	0.13	0.00	0.13	-11.7			
Nest07	0.04	0.22	0.02	0.23	-20.1			
Nest10	0.08	0.24	0.01	0.24	-13			

Table A2.6. Water chemistry measurements for sampling sites in the Sass-Klewi Nesting Area in May, August and September 2022.

Site ID	[TP] (µg/L)	[TN] (µg/L)	DSi (mg/L)	[DIC] (mg/L)	[DOC] (mg/L)	[SO ₄ ²⁻] (mg/L)	[Ca ²⁺] (mg/L)	[K ⁺] (mg/L)	[Mg ²⁺] (mg/L)	[Na ⁺] (mg/L)	[HCO ₃] (mg/L)	[Cl ⁻] (mg/L)
Sampling on May 28-29, 2022 (note: "Nest" sites were sampled May 17-19).												
SK1	17	1080	1.7	44.1	21.5	533.8	165.9	3	56.3	108.3	242.6	121.4
SK2	11	1270	2.5	19.9	25.3	274.3	91	2.6	38.9	8.5	109.7	2.9
SK3	9	580	0.6	25.4	14.3	531	205.1	2.5	28.7	23.4	148.6	30.9
SK4	6	624	1.2	24.3	16.8	387.1	146.8	2.3	26.4	12.3	129.7	9.9
SK5	5	632	1.5	24.7	15	396.6	165.2	2.4	24.6	8	138.6	5.2
SK6	8	683	1.8	26	16.9	420.4	169.6	2.4	27.4	10.4	142.7	7.2
SK7	7	709	1.4	25.4	16.5	367.5	156.1	2.4	24.9	9.6	139.3	6
SK8	8	978	1.4	29.3	19.7	406.9	144.4	2.8	40.6	29.8	158.2	20
SK9	8	806	0.7	23.5	17.3	354.1	143.8	2.6	31.5	19.5	141	11.4
SK10	17	1120	0.7	31.3	25.1	143.6	71.2	1.4	30.5	20.9	174.6	15.7
SK11	15	2120	5.7	24.6	28.2	1332.4	170.5	4	196.8	236.6	157.5	254
SK12	12	1340	1.2	17	16.2	553.7	163.5	2.5	47	34.9	94.7	20.8
SK13	17	619	0.6	26.2	16.1	374.2	169.2	2.2	23.2	7.1	155	3.7
SK14	4	590	0.9	27.4	14.5	412.1	195.2	2.4	18.7	5.9	148.8	4.1
SK15	6	587	1	25.9	13.8	405.1	189	2.5	20.2	6.6	150.2	4.6
SK16	17	1010	1.7	28.8	21.4	626.6	244.6	3	34.9	23.5	157.7	27.8
SK17	11	1870	1.1	24.5	35.5	304.5	92.4	5.3	47.4	31.6	154.8	31.3
SK18	17	1400	0.5	21	29.7	155.8	72.6	3.1	27.3	20.8	133.3	21.3
SK19	23	1690	0.2	33.1	32.4	180.8	96.3	5.7	33.6	17.3	199.8	24.7
SK20	8	824	0.8	28.4	16.8	596.8	232.9	2.5	36.5	51.7	163	73.6
SK21	13	1290	2.9	28.2	21.3	884.1	297.3	4.8	65.3	103.9	164.3	163.4
SK22	13	965	3	26.2	21.3	681.8	264.8	2.9	37.3	26.1	147.4	31.1
SK23	12	895	2.3	25.3	17.4	637.7	238.4	2.2	40	95.6	139.7	151
SK24	7	526	0.6	31	14.2	613.4	256.6	2.3	29.6	16.7	173.7	17.8
SK25	6	544	0.4	28.1	15.9	477.1	217.6	2	19.1	5.3	159.3	3.7
SK26	4	595	0.3	24.7	16.1	350.6	150.3	1.8	22.1	6	145.2	2.9
SK27	10	1380	9	16.8	18.9	895.6	319.5	2.1	59.3	153.4	97.9	257.2
SK28	11	864	1	13.3	17.1	515.3	179.6	2.4	34.5	81	84.5	116.6
SK29	8	342	2.2	35.2	10.2	1092.1	445.5	1.8	34.2	10.8	204	10.3
SK30	11	790	1.3	31.2	17.2	908.4	350.4	2.1	45.2	24.6	196.7	23.6
SK31	17	1460	1.9	24.1	25.7	538.5	174.7	2	52.4	75.5	142.7	108.6
SK32	15	1460	0.6	16	30.8	591.7	147.2	3	73	107.8	105.4	170
SK33	10	826	1.7	27.2	17.4	228.5	109.9	1.5	28.7	36.5	176.6	50.1
SK34	8	761	0.6	18.7	17	437.3	161.5	2.2	30.2	48.3	122.6	71
SK35	20	1650	0.3	27.9	29.8	235.6	88.6	4.9	43.4	33.5	187.1	45.5
SK36	10	1120	0.8	24.5	23.8	353.7	150.7	4.7	32.1	46.3	160.2	62.8
SK37	13	801	0.8	21.6	19.1	546	202.6	2.8	28	19.8	129.1	23.6
SK38	19	1190	1.5	23.7	27	542.9	186.1	3.8	38.9	28.9	133.7	34.4
SK39	8	1020	0.6	28.3	25.6	311.1	115.7	2.4	33.9	26.3	167.6	34.4
SK40	10	981	0.6	25	19	471	174.5	1.9	30.8	16.1	136.4	18
SK41	32	790	0.2	8.8	19.2	90.3	46.2	1.1	5.3	2.1	47.2	1.4
SK42	10	464	1.4	29.1	12.4	1071.6	429	2.1	24.1	5.1	174.6	4.2
SK43	20	2400	2.6	17.2	29.4	159.2	95.1	1.2	4.9	3.2	102.5	2
SK44	5	401	4.3	28	10.6	1394.9	540.5	2.1	36.4	10.4	170.5	9.8
SK45	6	593	2.5	29.4	16.6	613.3	246.4	1.9	26.3	7.4	163.1	7
SK46	14	1670	11.9	25.8	21.8	1158.5	331.9	3	84.9	33	143.5	34.3

SK47	19	1100	0.6	32.1	21.1	520.6	205.9	2.7	30.2	24.9	187.4	32.5
SK48	7	710	0.3	33	17.6	245	128.6	2.6	16.2	4.7	185.2	3.7
SK49	7	631	0.5	32.1	17.8	239.2	127.6	2.3	14.4	3.7	183.4	3.5
SK50	9	652	0.7	30.8	16.4	247.5	127.6	2.5	14.9	3.9	181.3	3
SK51	11	1020	0.2	34.4	26	137.9	81.2	3	20.4	8.8	191.5	8.3
SK52	10	1090	0.3	28.7	26.4	112.4	62.8	3.9	20.2	7.6	175.6	6.8
SK53	11	1010	0.4	35.7	24.2	158.5	88.8	2.8	20.2	8.2	201.8	7.5
SK54	28	2160	1.7	29.2	28.5	275.5	85.4	4.3	42.5	23.8	163.3	25.6
SK55	18	1290	1.3	17	21	510.5	118.6	3.4	58.6	42.6	105.6	48.2
SK56	15	1390	0.9	39.3	30.9	67.2	61.7	3.8	21.1	7.8	259.8	7
SK57	15	1250	0.6	24.7	20	230.3	76.8	1.9	32.8	15.7	142.1	17.5
SK58	16	1110	2	34.7	23.9	328	164.1	1.7	18.3	4	197.5	3.6
SK59	18	1880	3.5	24.1	31.1	683.6	198.7	4.7	61.6	24.5	142.1	22.5
SK60	18	838	1.9	18.6	19.1	345.8	148.6	2	17.8	6.1	103.4	6.8
SK61	14	1360	3.7	30.9	17.7	877.6	334.3	2.1	51	134	179.8	216.6
SK62	Sample lost in transit											
SK63	13	1080	0.6	22.4	15.6	504.1	198.2	3	25.2	17.8	129.7	22.2
RP	9	616	0.3	27.5	14.6	281	140.9	2.2	17.2	4.7	151.5	3.1
RKU	3	566	0.7	26.9	14.8	445.5	205.5	2.2	18.6	5.6	157.6	4.5
RSD	7	613	0.3	26.8	40.1	296.3	146.5	2.2	18	4.9	156.4	3.2
RKD	8	638	0.6	27.2	15.6	504.9	202.8	2.5	28.3	21.7	151.7	26.1
Nest04	21	994	1.5	30.3	24.4	74.6	60.3	3.1	18.7	8.5	171.8	5.8
Nest05	18	582	3.5	25.1	13.2	745.4	320.4	1.2	18.7	4.9	141.6	3
Nest07	12	573	2.5	24.8	14.8	349.8	179.1	2	17.5	7.1	134.5	5.4
Nest10	22	536	1.8	29.2	14.9	548.8	232.1	2	26.2	15.3	161.3	18

Sampling on August 9-10, 2022 (note: "Nest" sites were sampled August 7-8).

SK1	99	7280	5.1	24.2	51	1644.1	324	4.6	175.9	376	145.1	386.1
SK2	26	3680	11.5	13.4	58.6	796.7	190.8	7.8	98	26.4	86.4	5.8
SK3	9	1120	10	24.4	22.4	957.8	360.2	2.4	51.2	35.3	144	38.5
SK4	25	3560	8.4	13.2	51.5	1406.3	317.3	5.9	150.9	312.9	72.8	316.5
SK5	11	1480	11.3	21.9	29.2	686.2	252.9	2.7	47.9	21.3	134.8	12.8
SK6	14	1630	10.8	23.2	31	740.5	280.9	3.3	51.6	24.1	140.9	15.4
SK7	12	1600	10.5	14.7	29.2	675.9	248.9	2.8	45.8	21.3	84.2	12.5
SK8	18	2320	9.7	21.9	46.3	779.3	231.5	3	82.7	65.7	140.8	45.1
SK9	17	2060	11.7	16.1	35.8	765.7	252.5	3.5	69.3	50.1	98	31.9
SK10	20	2510	15.5	34.1	49.1	520.5	144.8	2.2	86.7	73.5	223.7	63.6
SK11	111	11700	13.4	35.1	82.2	4416.8	461.3	10.8	666.6	742.1	239.9	795.4
SK12	24	3920	8.2	17.2	53.2	1464.9	390.1	6	124.4	92.5	107.1	60.8
SK13	9	1130	9.8	28.2	26.2	822.9	337.7	1.4	44.3	14.9	172.2	9.1
SK14	13	2010	9.5	14.9	34	1069.5	395.1	3.5	51.1	20.4	94.1	16.5
SK15	11	1500	9.7	15.7	28	859.5	325.2	3.2	45	18.2	94.3	13.7
SK16	22	2960	7.1	13.8	51.6	1131.8	386	6.2	79.5	61.9	87.9	67.4
SK17	59	4710	10.6	31.6	61.8	438.3	150.5	7.1	72.5	61	197.6	60.6
SK18	20	3550	6.8	22.8	64.5	475	132.9	8.2	80.7	66.6	142.7	67.1
SK19	70	5140	4.4	22	71.7	478.3	129.5	14.1	87.4	52.8	143.8	68.6
SK20	13	1960	13.3	14.9	32	1176.7	391.1	5.2	83.2	109.9	89.5	174.6
SK21	15	2620	6.7	10.2	40.5	1648.8	486.7	9.8	130.8	209.6	62.2	330
SK22	11	1950	9.7	12.3	40.8	1175.9	402.9	4.3	73.6	58	75.4	65.7
SK23	26	4130	16	14.7	48.3	2174.6	723.9	5.6	136	316	93.9	534
SK24	6	932	10.5	27.8	18.9	1370.8	537.3	1.5	63.4	33.4	169.9	38.1
SK25	21	887	10.1	33.4	19.7	1039.5	452	1.2	43.3	11.1	203	7
SK26	10	845	11.2	38.8	23.1	878.5	397.8	0.9	46	13.3	240.6	7.9
SK27	not enough water to collect sufficient sample											
SK28	30	3840	2.3	12.2	52.1	1285.8	431.1	5.3	90.8	203.1	80.3	337
SK29	6	452	5.1	37.2	10.3	1290	543.7	2	44	10.2	235.5	8.7
SK30	36	3330	9.1	17.9	39.8	1864.8	637.2	2.1	98.4	60.5	112.2	71.8
SK31	21	3800	3.6	20.6	64.2	1384.5	400.9	4.9	132.6	188.2	135.8	294.5
SK32	28	3840	0.3	22.8	64.7	1379.1	331.4	7.2	166.2	272.9	147.8	431.3
SK33	14	2260	6.9	22.1	34.4	688.2	230.6	2.7	86.4	99.5	144	160.9
SK34	14	2620	11.5	16.8	40.5	1100.4	365.5	4.5	90.6	127.1	105.5	206
SK35	34	4770	3.5	20.9	75.6	667.6	136.7	12.3	110.4	91.4	130.9	122
SK36	16	2770	11.6	27.5	45	950.4	308.2	7	88.5	108.5	172.2	183.6
SK37	27	3230	16.6	16.9	45.6	1335.9	464.3	5.7	82.6	66.2	106.1	73.9
SK38	20	2540	13	18.1	44.5	893.1	303.2	7.6	84.2	60.8	116.1	62.6

SK39	18	2670	13.4	27.9	49.6	740.4	251.3	2.9	90.5	89	182.6	121.3
SK40	17	2510	11.3	21.5	40.4	879.5	308	2.1	73.2	48	136.8	55.1
SK41	45	3070	4.6	11	41.3	261.9	109.7	2.4	16	5.6	71.6	1.5
SK42	6	916	7.2	25.7	18.4	1443.9	646.2	1.4	41.9	8.9	155.1	5.8
SK43	15	2510	5.1	18.4	50.3	259.7	152.6	1.8	8.6	5.1	120.1	2.5
SK44	5	831	5.9	22.8	14.8	1579.6	663.8	2.4	48.2	15.3	141	16.1
SK45	6	1320	6	17.9	23.3	1610.4	646.2	1.6	66.1	16.3	114.6	10.7
SK46	13	2930	14.5	11.9	42	2499.3	818.4	5.6	179.6	66.4	82.8	56.3
SK47	13	1780	10.3	19.5	28.9	1550.4	570.2	3.6	96.3	98.2	132	129.9
SK48	6	1020	8.6	36	23.3	401.4	206.2	1.6	31.2	10.8	236.7	6.3
SK49	7	1140	9.9	32.6	24.4	399	205.6	1.2	26.6	7.1	218.4	2.5
SK50	8	1220	9.5	30.3	26	422.8	199.2	1.5	27.3	7.5	195.9	2.8
SK51	17	2410	9.7	37.5	47.3	327	147.8	2.6	46.9	23	262	16.5
SK52	16	2990	10.2	31.6	54	358	119.1	4.6	58.2	24.7	213.1	16.6
SK53	12	2360	11.8	24.2	42.8	467.2	157.3	3.5	50.3	22	153.6	16.4
SK54	30	5640	8.6	17.3	79.6	1085.2	210.2	10	163.5	91.8	124.2	105.6
SK55	27	4220	8.7	14.9	57.2	1408.2	311.8	9.3	173.4	118.6	94.5	129.9
SK56	16	3240	12.1	38.1	59.7	172.6	86.5	5.9	48.7	21.4	244.1	15.6
SK57	58	5660	9.2	18.7	54.4	753.6	182.3	4.9	107.2	57.8	128.7	58
SK58	30	2220	10.4	16.1	36.9	1244.7	490.4	0.4	57.1	12.1	113.5	7.7
SK59	11	2620	8.4	13.3	48.7	1050.6	321	7.6	100.1	43.1	92.8	32.5
SK60	13	2360	11.2	14.9	37.3	699.5	293.6	3.4	38.7	14	97.8	11.2
SK61	8	2260	5.6	9.7	27.3	2127.5	811.4	4.1	115.2	317	64.5	458.6
SK62	70	5890	10.6	17.9	37.4	1936.9	698.2	2	107.4	35.8	119.2	39.5
SK63	81	7360	5.4	19.8	48.2	1794.3	622.6	3.6	97.2	92.7	132.3	167.1
RP	59	1450	10.8	34.5	26.1	652.7	317.4	2.4	39.9	13.9	233.4	9.4
RKU	5	988	9.9	23	20.7	1115.6	486.6	1.6	48.7	19.3	147.4	16.6
RSD	36	932	11.5	32.2	20	794.8	380.2	2.4	37	9.4	208	4.4
RKD	45	2000	15.9	32.9	35.1	812.7	367.4	2.7	60.8	41	232.7	39.5
Nest04	215	7000	20.8	46.9	54.4	396.6	181.1	4.4	65.4	36.5	320.9	33.3
Nest05	92	3160	9.4	30.4	23.7	1511.4	671.4	1.6	49.6	9.8	181.4	4.8
Nest07	27	1470	9.9	17.4	20.7	1210.8	484.6	2.1	56.2	26.5	99.8	26.3
Nest10	15	930	9.7	33.6	17	1324.8	576.4	1.3	63.2	28.4	201.4	34.1

Sampling on September 15-17, 2022 (note: "Nest" sites were not sampled in September).

SK1	530	42100	3.7	38.7	60.6	1855	394.2	5.3	201	432.2	232.4	510.8
SK2	62	6320	15.5	25.7	77.6	1161.5	269.8	9.7	137.2	34.9	156.2	8.6
SK3	10	1230	11.9	28.3	22.5	1273	462	2.9	66.2	41.1	158.2	46.7
SK4	21	3680	8.5	24.6	53	2025.6	426.4	7.8	195.2	405.4	139.1	485.1
SK5	13	1720	10.7	22.7	30.7	846.2	293.4	3.6	59.8	27.4	135.4	17.4
SK	11	1770	10.1	24.9	31.1	933.3	314.8	4.1	66.2	31.3	141.2	19.8
SK7	14	2030	12	16.8	32.3	901.8	297.4	3.7	59.8	28.1	105.1	17.3
SK8	56	4240	10	24.6	50.3	1060.1	281.2	4.1	100.6	85.2	146	59.2
SK9	17	2610	10.5	18.9	41.5	1095.5	323	4.4	92.1	66	119.8	44.5
SK10	29	3520	15.9	44.3	57.1	746.3	183.8	3.3	110.8	97	261.7	90.8
SK11					not enough water to collect sufficient sample							
SK12	181	19100	12.8	35.9	58.8	2289.6	608.6	8.9	197.2	142.3	208.8	90.2
SK13	9	1180	10.9	29.2	24.1	1048.1	400.8	2.3	55.4	19	168.5	11.5
SK14	33	4110	10.9	19.2	37.8	1440.5	518	4.9	72.2	29.1	120.2	23.8
SK15	15	1930	11.6	18.8	30.4	1150.2	416	4	59.4	24	109.8	19.3
SK16	26	3560	9.2	17.2	55.9	1500.5	491.4	7.6	99.6	75.6	112.2	89.5
SK17	21	4310	11	29.8	70.5	733.3	173.4	12.2	102.5	75.4	182	72.7
SK18	16	4340	8.2	27.6	76.4	722.2	180.3	10.3	97.9	85.3	175.7	93.1
SK19	56	5860	8.1	49.7	83.1	745.9	223.8	20.5	113.5	70.2	307.7	101
SK20	20	2520	9.9	19	31.7	1490.1	492.8	7	94.1	146.9	116.4	219.6
SK21	21	3330	7.9	12.6	49.2	2041.2	599.6	13.1	160.9	273	82.7	411.9
SK22	10	2290	5.7	9.8	43.5	1433.1	474	5.5	86.7	70.9	58	79.7
SK23	16	3960	15.2	21.5	55.2	3173.3	1038.4	8.9	216.4	513.6	131.3	830.2
SK24	6	899	10.7	31.3	17.6	1550.7	591	3.3	55.9	28.7	179.3	46.2
SK25	11	951	11.4	36.9	21.1	1234.7	503.6	2.7	41.7	12.6	214.7	12
SK26	11	837	10.2	42.6	19.3	1124.4	467.8	3	44.7	12.9	246.4	11.3
SK27												
SK28	60	6830	1.6	28.1	67.6	2100.8	724.2	7.6	137.4	328.2	166.6	538.2
SK29	14	726	4.9	43.1	8.8	1374.9	585	2.3	35.7	8.5	253.7	10.9
SK30	43	5060	5.6	28.7	41.2	2271.1	777.2	2.7	121.1	52.7	168.5	90.5

SK31	29	5490	1.6	27.7	68.3	1963	566.6	6.7	190.3	280.8	165.7	414.1
SK32	18	4610	0.2	26.8	74.8	1913	448.8	10.1	236.6	399.4	166.3	584.6
SK33	14	2720	5.8	25.9	40.1	958.5	289.2	4.3	83.2	133.1	153.6	211.8
SK34	18	3890	9.4	23.5	48.8	1547.9	518.4	7.2	111.5	184	142.3	281.6
SK35	Sample lost in transit											
SK36	16	3250	8.7	30.4	52.9	1241.8	419.6	9.1	102.4	161.3	185.9	235.5
SK37	23	3840	18.9	23	58.3	1869	645.6	7.7	101.6	90.3	143	110.5
SK38	29	3350	11.7	25	50.7	1207.9	384.8	9.5	89.8	75.8	152.6	78.7
SK39	27	3420	14.3	29.7	57.1	1080.4	334.2	4.2	102.1	122.7	193.1	185.2
SK40	20	3400	14.4	29.1	47.6	1257.8	428.6	2.7	88.8	70.1	182.1	85.4
SK41	27	2710	9.8	20.5	49.8	367.8	159.5	3	22.2	7.7	126	2.5
SK42	11	1000	7	29.6	17	1612.7	662.4	1.6	37.7	7.6	171.5	6.6
SK43	22	4670	6.9	27.1	56.2	318.1	176.4	1.8	8.8	5.1	161.8	3.7
SK44	5	795	5.8	29.1	14.1	1611.7	646.2	3.2	49.8	16	174.9	20
SK45	7	1850	5.7	18.6	25.7	1964.8	726.2	2.4	65.7	18.1	108.9	15.7
SK46	19	3270	17.7	19.7	43	2811	857.8	6	198.6	53.4	127	67
SK47	19	2760	6.4	17.1	33.7	1939.5	636.2	5.9	114.6	136.9	102.8	170.2
SK48	8	1050	9.5	43.3	23.2	480.7	224.8	3.8	33.5	12.1	257	9.6
SK49	8	1180	10.1	35.3	25.6	434.1	206.8	3.2	29.7	8.8	212.5	4.8
SK50	8	1100	9.6	34	25.8	459.5	211.6	2.9	29.4	8.6	201.2	4.5
SK51	23	2710	10.5	45.6	52.3	381.5	164.3	4.3	51	25	272.5	20.9
SK52	18	3380	11	33.7	63.9	463.3	142.5	6.7	67.5	30	217.1	23.3
SK53	14	2950	12.7	26.1	52.4	619.6	196.8	5.5	63.2	28.3	164.2	23
SK54	21	6800	6	29.7	112.6	1676.9	299.2	13.9	251.2	156.5	199.9	158.9
SK55	34	4610	12.6	29.3	70	1996.3	417	12.5	236.2	175.4	184.7	186.2
SK56	37	4450	11.8	46	73.7	280.8	95.6	8.5	58.3	26.4	291.1	23
SK57	38	5350	9.8	28.8	74.6	1125.4	246.6	7	151.4	69.6	188.7	83.2
SK58	27	2860	9.9	24.1	45	1599.1	596.2	1.8	60.8	13.4	151.1	10.7
SK59	15	3060	10.5	15.6	56.1	1401.5	383	9.8	123.1	49.3	103.2	40.8
SK60	36	3970	14.4	22.1	45.3	1031	385.8	5.2	50.8	18.4	142.2	15.2
SK61	7	2200	4.8	14.6	27.2	2692.9	938.4	9.7	141.6	454.4	89.9	732.9
SK62	35	4590	8.7	27.2	42.3	2361.6	791.6	4	138.3	19.7	168.6	49.4
SK63	504	30700	3.6	33.3	69.1	2520.7	915.6	7.2	161.1	179.4	206.9	277.2
RP	12	1000	11.4	35	24.1	919.5	377.4	4.3	32.5	12.3	216.6	12.8
RKU	11	1290	9.7	19.5	24.9	1290	493.4	3.3	39.7	15.8	117.1	16.5
RSD	27	1380	11.4	25.2	24.6	1273.8	475.8	3.8	40.8	26.7	160.1	44.3

Table A2.7. Correlation coefficients for water chemistry variables in the Principal Component Analysis (PCA) presented in Figure 4.6a.

Variable	PCA Axis 1	PCA Axis 2
TP	-0.46	-0.40
TN	-0.73	-0.51
Bicarb	0.45	-0.38
DIC	0.47	-0.34
Cl	-0.90	-0.01
SO4	-0.66	0.65
DOC	-0.64	-0.70
Ca	-0.37	0.84
K	-0.72	-0.41
Mg	-0.92	-0.03
Na	-0.92	-0.08
Conductivity	-0.76	0.53
DSi	0.10	-0.12
pH	-0.40	-0.27

Table A2.8. Principal Component Analysis (PCA) sample scores for sampling sites across the Sass-Klewi Nesting Area in August and September 2022.

Site ID	PCA Axis 1	PCA Axis 2	PCA Axis 1	PCA Axis 2
	Aug-22		Sep-22	
SK1	-3.84	-0.53	-4.50	-1.80
SK2	-0.73	-1.34	-1.39	-2.08
SK3	1.39	1.25	0.90	1.52
SK4	-3.74	0.32	-3.70	0.33
SK5	1.79	0.12	1.11	0.01
SK6	1.44	0.08	0.84	0.27
SK7	1.15	0.35	0.54	0.34
SK8	-0.09	-0.68	-1.09	-1.16
SK9	-0.05	-0.09	-0.74	0.07
SK10	0.81	-2.12	0.06	-2.42
SK11				
SK12	-2.11	0.22	-3.47	-1.06
SK13	2.76	1.01	2.19	1.11
SK14	0.37	1.03	-0.76	0.54
SK15	1.17	0.99	0.37	0.91
SK16	-1.71	0.24	-2.21	0.21
SK17	-0.03	-2.97	-1.07	-2.54
SK18	-0.53	-2.58	-1.24	-2.38
SK19	-1.36	-3.35	-1.57	-3.55
SK20	-1.46	1.01	-2.04	0.95
SK21	-3.77	1.37	-4.39	1.08
SK22	-1.21	1.02	-2.23	1.45
SK23	-3.99	1.38	-4.60	1.59
SK24	1.50	2.40	1.27	2.29
SK25	2.95	1.55	2.45	1.52
SK26	3.36	1.22	2.74	1.35
SK27				
SK28	-3.00	0.90	-3.79	0.12
SK29	3.43	2.84	3.10	2.45
SK30	-1.57	1.50	-1.68	1.23
SK31	-2.71	0.06	-3.70	0.18
SK32	-3.43	-0.41	-4.22	-0.10
SK33	-0.18	-0.22	-1.03	-0.21
SK34	-1.58	0.46	-2.58	0.31
SK35	-2.22	-2.73		
SK36	-0.95	-0.53	-1.78	-0.27
SK37	-1.47	0.30	-2.19	0.24
SK38	-0.81	-0.37	-1.34	-0.41
SK39	-0.29	-0.98	-1.37	-0.94
SK40	0.17	0.08	-0.52	0.01
SK41	2.53	-2.24	2.78	-1.90
SK42	2.60	2.83	2.47	2.71
SK43	4.18	-1.58	3.83	-2.01
SK44	1.75	3.05	1.69	2.76
SK45	1.14	2.85	0.37	2.88
SK46	-2.71	2.03	-2.43	1.65
SK47	-1.05	1.69	-2.30	1.80
SK48	4.20	-0.45	3.41	-0.78
SK49	4.61	-0.37	3.74	-0.74
SK50	4.27	-0.45	3.83	-0.45
SK51	2.74	-2.38	1.99	-2.73
SK52	1.74	-3.01	0.90	-2.93
SK53	1.79	-1.65	0.76	-1.64
SK54	-2.82	-1.90	-3.63	-1.85
SK55	-3.18	-0.52	-3.39	-0.82
SK56	2.96	-4.81	1.34	-4.63
SK57	-1.32	-1.93	-1.78	-1.81
SK58	0.93	1.42	0.43	0.99
SK59	-1.23	0.03	-1.93	-0.03

SK60	1.24	0.13	0.14	-0.56
SK61	-3.50	3.30	-3.86	2.98
SK62	-1.46	1.30	-1.24	1.06
SK63	-2.64	0.42	-4.57	-0.82
Nest04	1.47	-4.02		
Nest05	1.73	1.43		
Nest07	1.10	2.23		
Nest10	1.98	2.26		

Table A2.9. Original and scaled parameters used to develop the Vulnerability Index.

Site ID	Original values			Scaled values			Vulnerability Index
	August E/I	Drawdown (%) June-August	Aug PCA1	August E/I	Drawdown (%) June-August	August PCA1	
SK1	0.66	26.13	-3.84	0.40	0.35	0.98	0.58
SK2	0.83	28.90	-0.73	0.52	0.39	0.62	0.51
SK3	0.40	17.43	1.39	0.22	0.22	0.37	0.27
SK4	0.78	21.01	-3.74	0.49	0.27	0.97	0.58
SK5	0.41	16.48	1.79	0.23	0.21	0.33	0.25
SK6	0.47	15.26	1.44	0.27	0.19	0.37	0.28
SK7	0.38	17.50	1.15	0.21	0.22	0.40	0.28
SK8	0.65	22.92	-0.09	0.40	0.30	0.55	0.42
SK9	0.57	24.66	-0.05	0.34	0.33	0.54	0.40
SK10	0.62	36.84	0.81	0.38	0.51	0.44	0.44
SK11	1.50	34.79		1.00	0.48		0.74
SK12	1.50	34.91	-2.11	1.00	0.48	0.78	0.75
SK13	0.32	12.91	2.76	0.16	0.15	0.22	0.18
SK14	0.67	38.82	0.37	0.41	0.54	0.49	0.48
SK15	0.56	26.63	1.17	0.33	0.36	0.40	0.36
SK16	0.99	26.18	-1.71	0.64	0.35	0.73	0.57
SK17	1.18	20.19	-0.03	0.77	0.26	0.54	0.52
SK18	0.66	26.05	-0.53	0.40	0.35	0.60	0.45
SK19	1.22	28.85	-1.36	0.80	0.39	0.69	0.63
SK20	0.80	33.57	-1.46	0.50	0.46	0.71	0.56
SK21	0.66	28.98	-3.77	0.40	0.39	0.97	0.59
SK22	0.73	21.26	-1.21	0.45	0.28	0.68	0.47
SK23	1.42	41.90	-3.99	0.94	0.59	1.00	0.84
SK24	0.27	20.15	1.50	0.13	0.26	0.36	0.25
SK25	0.25	19.01	2.95	0.11	0.24	0.19	0.18
SK26	0.22	16.22	3.36	0.09	0.20	0.14	0.15
SK27	1.41	69.29		0.94	1.00		0.97
SK28	1.30	37.12	-3.00	0.86	0.52	0.89	0.75
SK29	0.09	2.80	3.43	0.00	0.00	0.14	0.05
SK30	0.68	43.50	-1.57	0.42	0.61	0.72	0.58
SK31	1.21	23.10	-2.71	0.79	0.31	0.85	0.65
SK32	1.50	30.96	-3.43	1.00	0.42	0.94	0.79
SK33	0.83	13.33	-0.18	0.52	0.16	0.56	0.41
SK34	0.90		-1.58	0.57		0.72	0.65
SK35	1.14		-2.22	0.74		0.79	0.51
SK36	0.86	25.38	-0.95	0.55	0.34	0.65	0.51
SK37	1.29	31.41	-1.47	0.85	0.43	0.71	0.66
SK38	0.89		-0.81	0.57		0.63	0.40
SK39	0.71	23.66	-0.29	0.44	0.31	0.57	0.44
SK40	0.66	13.38	0.17	0.40	0.16	0.52	0.36
SK41	0.74	17.78	2.53	0.46	0.23	0.24	0.31
SK42	0.24	22.07	2.60	0.11	0.29	0.23	0.21
SK43	0.65	15.53	4.18	0.40	0.19	0.05	0.21
SK44	0.23	10.63	1.75	0.10	0.12	0.33	0.18
SK45	0.33	13.17	1.14	0.17	0.16	0.40	0.24
SK46	0.79		-2.71	0.50		0.85	0.67
SK47	0.57	24.63	-1.05	0.34	0.33	0.66	0.44
SK48	0.19	14.35	4.20	0.07	0.17	0.05	0.10
SK49	0.24	10.37	4.61	0.11	0.11	0.00	0.07
SK50	0.31	11.49	4.27	0.16	0.13	0.04	0.11

SK51	0.36	17.02	2.74	0.19	0.21	0.22	0.21
SK52	0.50	25.00	1.74	0.29	0.33	0.33	0.32
SK53	0.54	22.84	1.79	0.32	0.30	0.33	0.32
SK54	1.41	31.75	-2.82	0.94	0.44	0.86	0.75
SK55	1.40	32.54	-3.18	0.93	0.45	0.91	0.76
SK56	0.68	31.00	2.96	0.42	0.42	0.19	0.34
SK57	0.93	22.41	-1.32	0.60	0.29	0.69	0.53
SK58	0.33	10.86	0.93	0.17	0.12	0.43	0.24
SK59	0.64	17.78	-1.23	0.39	0.23	0.68	0.43
SK60	0.74	22.79	1.24	0.46	0.30	0.39	0.38
SK61	0.91	4.48	-3.50	0.58	0.03	0.94	0.52
SK62	1.47	39.64	-1.46	0.98	0.55	0.71	0.75
SK63	1.06	46.94	-2.64	0.69	0.66	0.84	0.73
Nest04	0.41		1.47	0.23		0.36	0.30
Nest05	0.30		1.73	0.15		0.33	0.24
Nest07	0.54		1.10	0.32		0.41	0.36
Nest10	0.22		1.98	0.09		0.31	0.20

A2.3 Chapter 4 supplementary figures

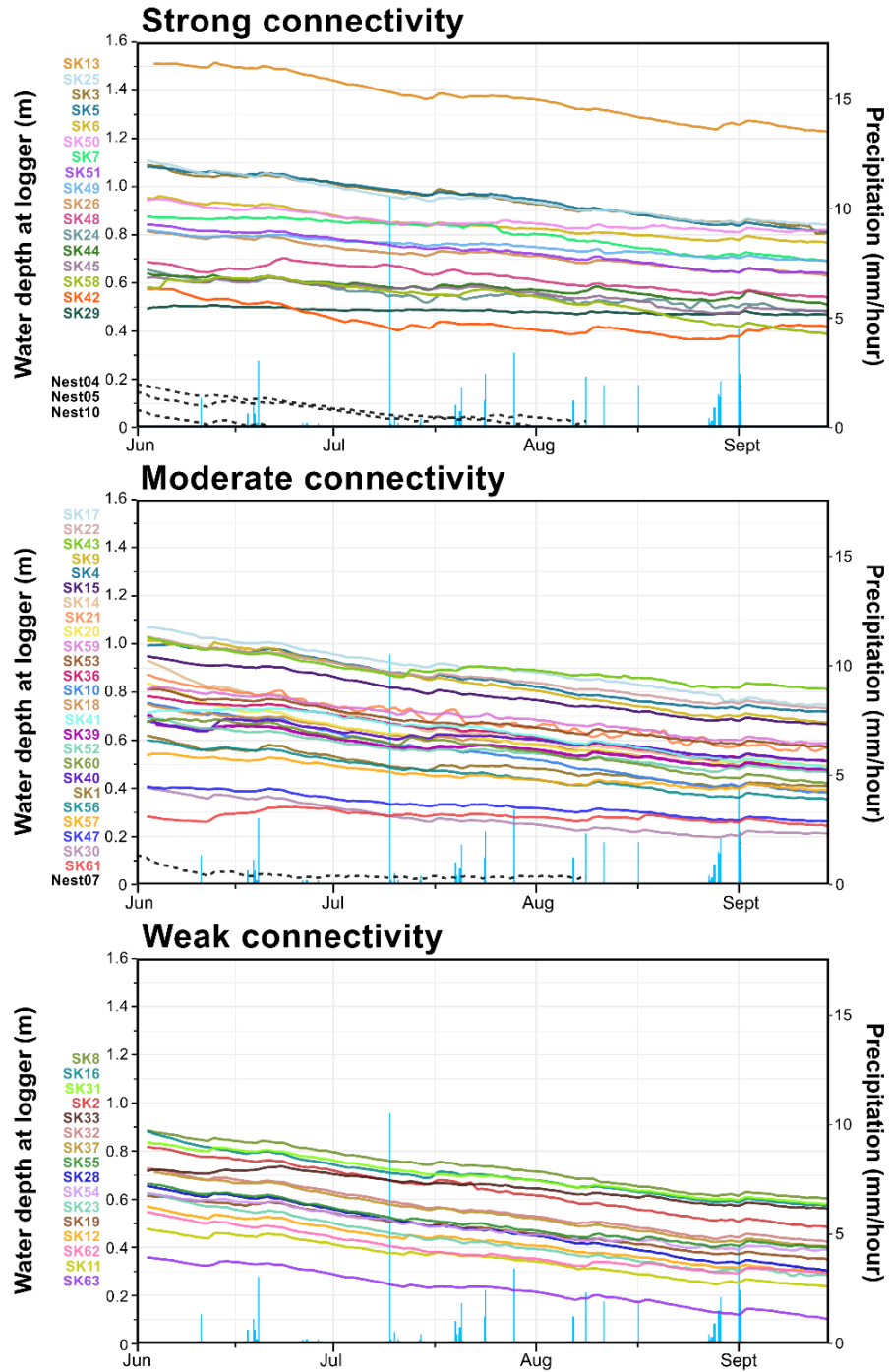


Figure A2.1. Temporal patterns of pond depth variation in the Sass-Klewi Nesting Area from June to mid-September 2022. Panels separate ponds that have been classified into three evaporation-to-inflow (E/I) ratio-defined categories that reflect a gradient of groundwater connectivity: strong, moderate and weak. Also shown (as vertical light-blue bars) is daily precipitation (mm) measured at Fort Smith Airport (ID: 2202202).

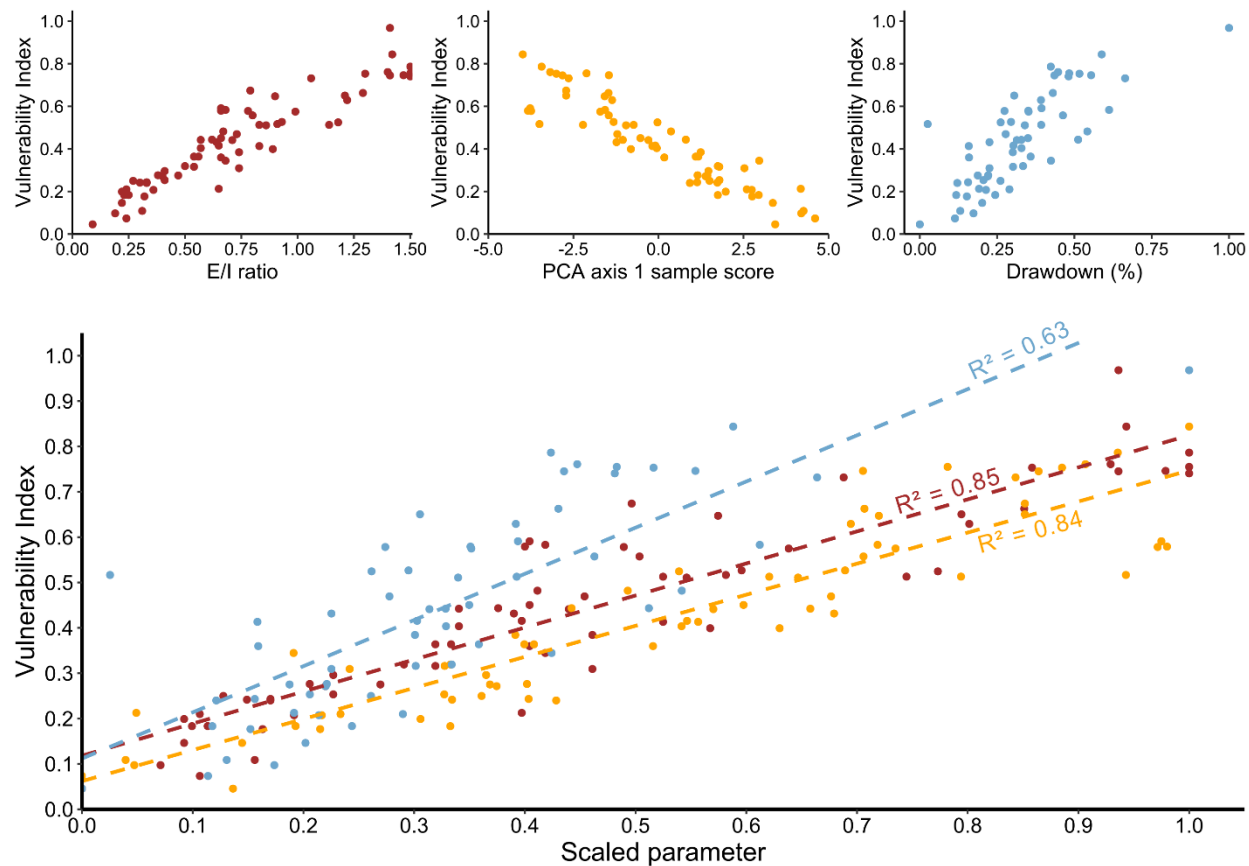


Figure A2.2. Crossplots showing the relations between a key variable (evaporation-to-inflow (E/I) ratio, water chemistry PCA axis 1 sample score, drawdown percent) from each of the three types of measurement in the study for August 2022 and the Vulnerability Index. Original and scaled variables and the Vulnerability Index scores are reported in Table A2.9.

Technical Report for Monitoring Lakes in the Peace-Athabasca Delta, Alberta

Prepared for:

Parks Canada Agency (Wood Buffalo National Park)

Prepared by:

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May 11, 2020



A3.1.1 Executive Summary

The Peace-Athabasca Delta (PAD) is threatened by multiple potential stressors including climate change, hydroelectric development and oil sands production, yet no comprehensive aquatic ecosystem monitoring program is currently in place for the abundant shallow lakes in this landscape. Thus, it has been difficult, if not impossible, to track how the status and trends of aquatic ecosystem conditions in the PAD may be responding (and will continue to respond) to these stressors. This is a critical knowledge gap, because many of the pressing concerns of environmental degradation by human activities center on the delta's aquatic ecosystems, including the perched basins [WHC/IUCN 2017, WBNP 2019]. To address this, we conducted research during 2015-2019, supported mainly by an NSERC Collaborative and Research Development (CRD) grant (in partnership with Wood Buffalo National Park (WBNP) and other agencies), to establish the foundation of a state-of-the-art monitoring framework for ongoing assessment of hydro-ecological conditions and contaminant concentrations in lakes of the PAD. This technical report serves to transition the research and knowledge gained towards agency-based operation. The Report outlines: 1) key results generated by our research program, 2) methods for effective sample collection and processing, and 3) step-by-step instruction (Standard Operating Procedures) for Parks Canada and community based programs to continue integrative monitoring for changes in Hydrology, Water Quality, and Contaminants in the format of a lake sampling plan for 2020 and beyond that builds upon the 5-yr database established for 2015-2019.

The components of the proposed hydro-ecological monitoring program are capable of:

- Detecting ice-jam and open-water flooding of lakes (including timing, spatial extent and magnitude of influence on lake water balance)
- Identifying evaporative hotspots where lakes are vulnerable to desiccation
- Discerning effects of flooding and evaporation on water quality
- Detecting enrichment of contaminants of concern (metals) in lakes conveyed by recently-deposited river sediment

Key results generated through this research-to-monitoring program include:

- Water balance of lakes in the PAD is variably affected by several influential hydrological processes (flooding, snowmelt, rainfall, evaporation) which are dynamic over space and time. Lakes in the relic Peace Delta are more vulnerable to water loss by evaporation.
- Flooding occurred in 2017 - 2019. Ice-jam flooded areas were mainly restricted to the central Athabasca Delta and lakes located near to distributary channels of the Peace River. Open-water flooding occurred only in the central Athabasca Delta, not in the Peace Delta.
- Water quality parameters associated with flooding (elevated turbidity, and total phosphorous, sulfate, dissolved silica concentrations) were not sustained in 2015, one year after a major ice-jam flood in spring 2014. The effects of flooding on lake water quality were short-lived, providing further insight into how strongly lakes are influenced by ongoing climate change and unlikely to be altered by a single isolated flood event.
- Widespread ice-jam flooding in 2018 allowed for opportunistic sampling of recently deposited river-supplied surface sediment from flooded lakes for analysis of metals of concern. Metals concentrations were not enriched above pre-1920 baseline reference concentrations measured in lake sediment cores, indicating current values remain within the range of natural variation. This includes for metals associated with oil sands development (vanadium and nickel).

The hydro-ecological monitoring program described in this report directly addresses Recommendation 9 from the 2017 WHC/IUCN Report on the Reactive Monitoring Mission to Wood Buffalo National Park: *“Expand the scope of monitoring and project assessments to encompass possible individual and cumulative impacts on the Outstanding Universal Value of the property and in particular the PAD”* [WHC/IUCN 2017, p.4]. It aims to contribute to Wood Buffalo National Park’s Action Plan [WBNP 2019].

A3.1.2 Rationale

Growing concerns about deterioration of the Peace-Athabasca Delta (PAD) by cumulative effects of multiple anthropogenic stressors on this internationally recognized ecosystem has prompted a Federal Action Plan [WBNP 2019]. Among the recommendations for action is continued development and implementation of an aquatic ecosystem monitoring program for the PAD capable of tracking changes in water supply, water levels and water quality, and the processes driving changes. Since October 2000, university-based research has included assessment of water balance, water quality, ecological conditions and contaminant deposition, and the hydrological processes regulating these lake characteristics via sampling from a large set of lake and river locations. The lakes span the hydrological gradients of the delta, including open-, restricted- and closed-drainage basins. The river sites were sampled to characterize the chemistry of input water to lakes during flood events. Frequent sampling occurred 2000-2003, sampling was less regular during 2003-2006, and no sampling occurred during 2007-2014. The findings identified marked spatial and temporal variability and highlighted the need for innovative and sustainable monitoring techniques in this dynamic, difficult-to-access landscape. Certainly, consistent, long-term monitoring of a representative set of lakes and rivers is required to track the influence of episodic flood events, given their importance for hydro-ecological conditions of lakes across the delta. Recognizing this need, a 5-year research program was launched in 2015, supported by a research grant from the Natural Sciences and Engineering Research Council (NSERC) and contributions from Parks Canada, Alberta Environment and Parks, BC Hydro, Suncor and CNRL. The short-term objectives were to develop and apply novel combinations of research and monitoring methods to 1) track relations among climate, river flow and lake hydrological conditions in lakes of the PAD, and 2) link the spatial distribution of contaminants (PACs, metals) and their toxicity characteristics in the delta with hydrological pathways (i.e., Athabasca River floodwaters, Peace River floodwaters, snowmelt, rainfall) and limnological conditions. The long-term objectives were to use this knowledge to establish the foundation of a state-of-the-art monitoring framework for the delta, which will provide metrics to ensure industrial activities are following best

environmental practices to limit downstream cumulative effects. Here, we present the foundation of knowledge obtained during those 5 years of sampling at 60 lakes and 9 river sites (Table A3.1.1) and propose plans and protocols for how Parks Canada can continue to monitor water balance, water quality and contaminant concentrations in the delta's water bodies.

The monitoring research aimed to integrate sampling of key components capable of addressing concerns of lake drying, ecological degradation, and spread of pollutants to the delta's water courses; namely, 1) Hydrology, 2) Water quality, and 3) Contaminants. Monitoring Component 1 (Hydrology) focuses on complementary information provided by sampling and analysis of water isotope composition and deployment of water-level loggers, which together can distinguish lake water input sources, magnitude of evaporative loss and drawdown and water-level fluctuations, and the timing, duration and spatial extent of flood events. Monitoring Component 2 (Water Quality) focuses on sampling of water chemistry parameters that inform the link between hydrological processes and abiotic and biotic processes in lakes that affect spatial and temporal variation in water quality. Monitoring Component 3 (Contaminants) focuses on collection of lake surface sediment soon after flood events, as it provides opportunity to assess for enrichment of metals of concern delivered via the Athabasca River from oil sands mining activities 200 km upstream, and via the Peace River which contributes higher concentrations of some metals of concern than the Athabasca River. The following sections outline individual objectives, prior results and key information provided by each of the three monitoring components.

Table A3.1.1. Inventory of samples and datasets obtained from lakes (60) and rivers (9) in the Peace-Athabasca Delta during 2015-2019. Timing of deployment (d) and retrieval (r) of water-level data loggers are indicated, as well as the number of lakes with a water-level data logger.

SAMPLES	2015				2016			2017			2018			2019		
	May	June	July	Sept	May	July	Sept	May	July	Sept	May	July	Sept	May	July	Sept
HYDROLOGY																
Water isotopes	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
Water-level loggers		d-10		r		d-12	r	d-20		r	d-60		r	d-60		r
WATER QUALITY																
Water chemistry	x		x	x	x	x	x	x	x	x	x	x	x	x	x	x
CONTAMINANTS																
Surface sediment	x									x		x				

A3.1.3 Site Selection

A total of 60 lakes and 9 river sites that capture the hydro-ecological gradients of the PAD were selected for the monitoring research. Forty-two of the study lakes are a subset of sites sampled previously [Wolfe et al. 2007b]. The remaining sites were selected in consultation with Parks Canada and a local community-based monitoring program coordinator to align with sites sampled by the Peace-Athabasca Delta Ecological Monitoring Program (PADEMP) annual muskrat surveys. The lake sites span the full hydrologic spectrum of the PAD and cover the spatial extent of both the Peace and Athabasca sectors (Figure A3.1.1). They do not extend to the Birch River Delta area. They do extend to the southeastern corner of Lake Claire, where concerns existed about potential future effects of the proposed Frontier Mine (Teck Resources). The river sites are sampled to characterize water quality and isotope composition of flood waters that enter lakes episodically. Sites were accessed primarily by helicopter and sampling occurred over two to four consecutive days during each sampling campaign. Sampling campaigns typically occurred 3 times during the ice-free season in May, July and September of 2015–2019. Sampling also occurred in June of 2015 to deploy semi-permeable membrane devices measuring concentrations of dissolved PACs. Samples were not collected in May 2016 due to wildfire.

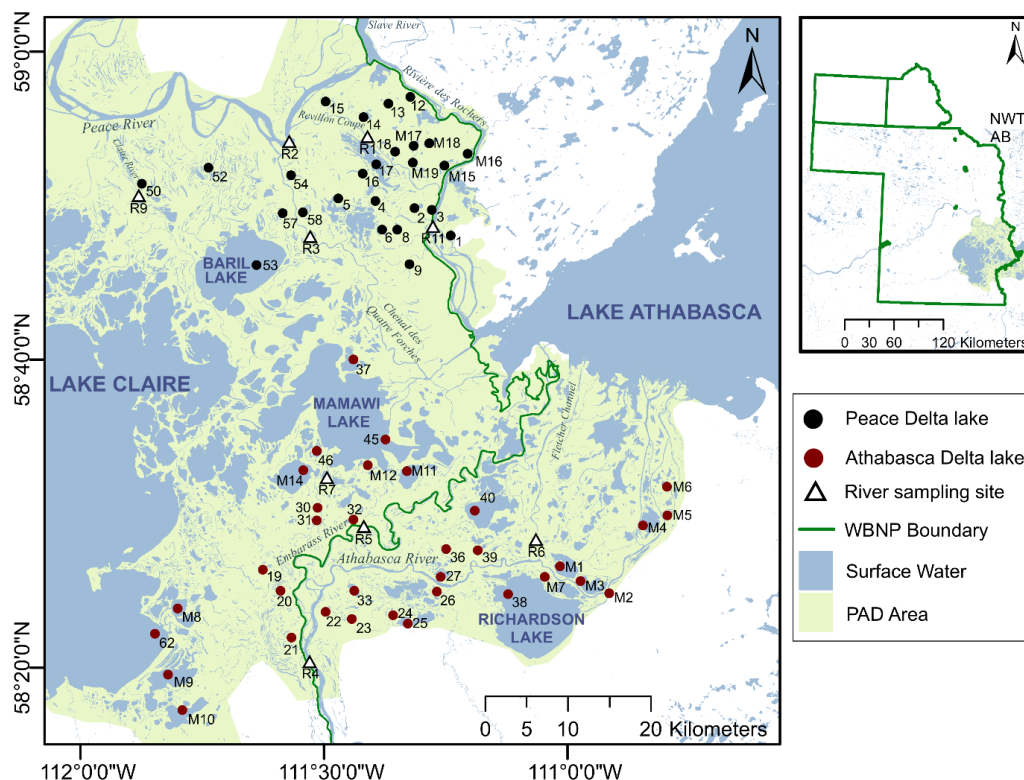


Figure A3.1.1. Map showing locations of the sampling sites in the Peace-Athabasca Delta, Alberta. Lake sites in the Athabasca Delta are solid red circles, lake sites in the Peace Delta are solid black circles and river sampling sites are triangles outlined in black. The boundary of Wood Buffalo National Park (WBNP) is outlined in dark green and the area of the entire Peace-Athabasca Delta is light green. Surface water bodies are filled in blue.

A3.1.4 Components of the monitoring program

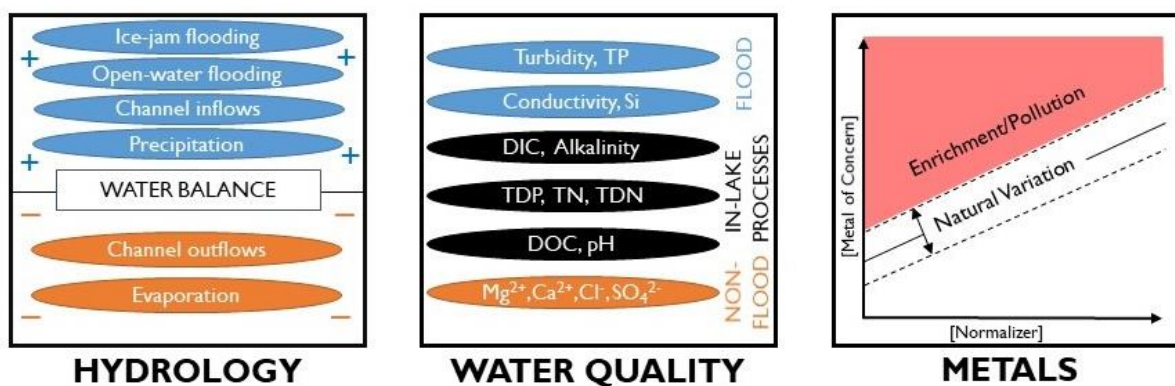


Figure A3.1.2. Schematic depicting the associated knowledge generated, including processes, variables and drivers, for each of the 3 monitoring components.

A3.1.4.1 Monitoring Component 1: Hydrological processes that control lake water balances

Objective: To utilize water isotope compositions and water level loggers, collected and deployed across the PAD, to track hydrological conditions of lakes and identify hydrological processes (river floodwater, snowmelt, rainfall, evaporation) controlling individual lake water balances and water levels, and their landscape-scale spatial patterns and trends over time.

Information Provided: Water isotope compositions delineate the influence of hydrological processes on lakes, allowing, for example, the identification of hotspots of evaporation, as well as the spatial extent of flood events and magnitude of influence on lake water balance. Water isotope compositions reflect an integrated signal of recent weeks, providing the opportunity to track hydrological events after they occur (e.g., spring flooding). Water-level loggers allow precise, hourly, measurement of lake water levels during the ice-free season. These measurements can identify the timing and duration of hydrological events and processes, including open-water flooding, precipitation and evaporative drawdown, and can be used to track and evaluate lake water-level responses to these hydrological events and processes.

Results to Date

[1] *Identification of spatio-temporal patterns of lake water balances using ‘isoscapes’*

Utilizing an isotope-mass balance model, Evaporation-to-Inflow (E/I) ratio ‘isoscapes’ of the PAD have been prepared for the 15 sampling events during ice-free seasons of 2015-2019 (Figure A3.1.3). These isoscapes provide an unprecedented perspective on intra- and inter-annual variation of water balance conditions and forms a hydrological foundation of a monitoring framework for the delta. E/I ratios show marked temporal and spatial variation across the landscape. Summer generally contains the greatest proportion of lakes with E/I ratios >1 , indicating water-level drawdown by evaporative water loss. Exceptions were 2018 and 2019 when a substantial portion of the Athabasca sector contained lakes with lower E/I ratios due to the strong influence of open-water flooding. Strong influence of evaporation

was also detected in spring of 2016, coinciding with regional wildfires, and fall of 2017, which align with field observations of lakes desiccating during these periods. Regions where E/I ratios consistently exceeded 1 include the north and central Peace sector and the southwestern Athabasca sector (summer 2015, spring and summer 2016, summer and fall 2017, summer 2019). Across the 15 sampling episodes, we detected some degree of flooding for 8 of the sampling episodes. These include spring of 2017-2019, summer of 2017-2019 and fall of 2018 and 2019. These flood events varied substantially in spatial extent. The flooding was primarily restricted to the central Athabasca sector along Embarras River and Mamawi Creek, as well as areas adjacent to the Athabasca River, but did not extend to the terminus of the Athabasca Delta.

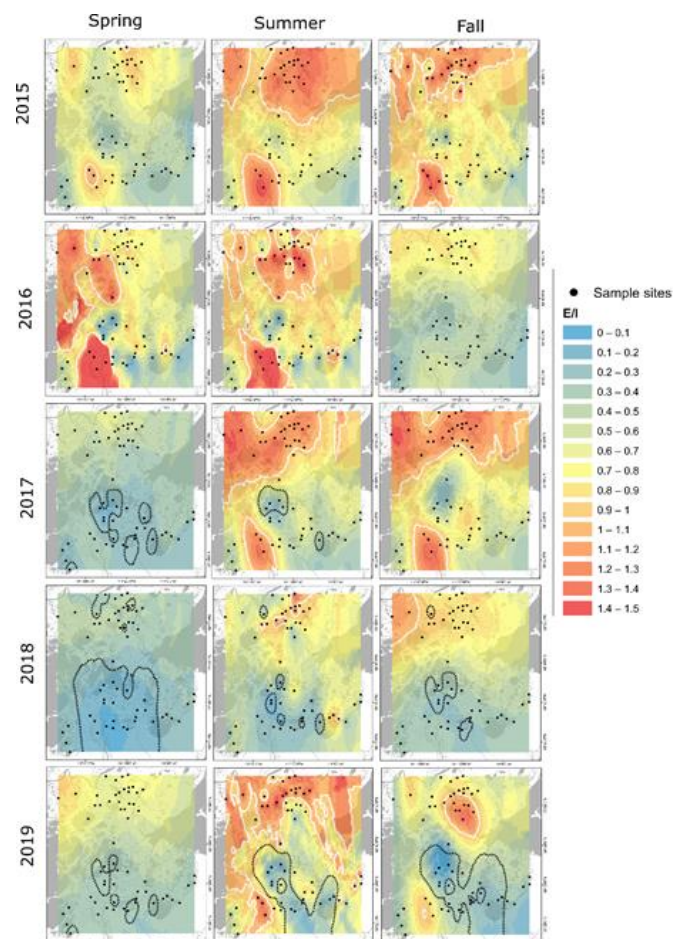


Figure A3.1.3. ‘Isoscapes’ displaying interpolated evaporation-to-inflow (E/I) ratios across the Peace-Athabasca Delta for each spring, summer and fall of 2015-2019. E/I ratios were determined from water isotope analysis. Black dashed lines represent flood extents while white dashed lines represent areas with E/I ratios greater than 1. Warmer colours represent stronger evaporation. Colder colours represent more inflow.

[2] Delineating the magnitude and extent of ice-jam flooding

In May 2018, a widespread ice-jam flood event occurred mainly in the southern Athabasca Delta, as well as limited areas of the north-central Peace Delta. Based on water samples collected ~3 weeks after flooding and analysis of water isotope compositions, lake-specific estimates of input water isotope composition were modelled after accounting for influence of evaporative isotopic enrichment. Then, using the distinct isotopic signature of input water sources, we developed a set of binary mixing models and estimated the proportion of input to flooded lakes attributable to river floodwater and precipitation [snow or rain; Remmer et al. 2020c]. This approach allowed delineation of the spatial extent of flooding and magnitude of influence on lake water balance (Figure A3.1.4), which were not captured by other methods including direct observations from flyovers. Nearly half of the lakes sampled received river floodwaters in spring 2018, and the estimated proportion of lake input provided by river floodwater to the flooded lakes averaged 87.2%. The flood event provided an excellent opportunity to demonstrate the efficiency and effectiveness of water isotope tracers as a hydrological monitoring tool for quantifying spatial extent and magnitude of river floodwater influence on delta lakes and is a method less prone to logistical constraints (i.e., sampling can occur after a flood occurs, not at the exact moment of flooding). These results are published in Remmer et al. [2020c; open access at <https://doi.org/10.1002/hyp.13585>].

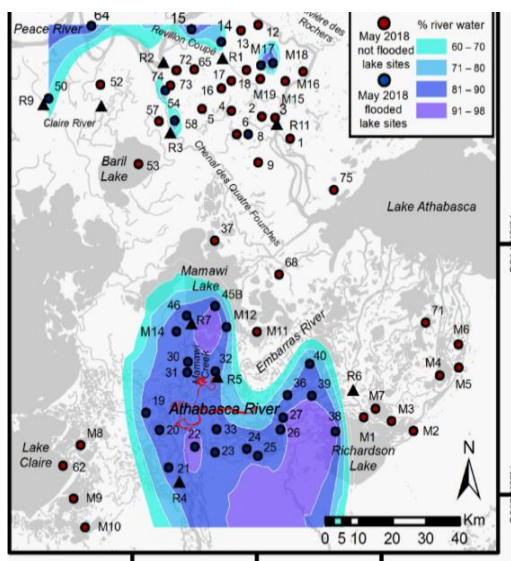


Figure A3.1.4. Extent and magnitude of river floodwaters in May 2018 derived from lake water isotope data [from Remmer et al. 2020c].

[3] *Lake water-level behaviours and responses to flooding and evaporation*

Water-level loggers were installed in 60 lakes across the Peace-Athabasca Delta and recorded hourly water-level measurements during the ice-free seasons of 2018 and 2019. Lake water-level patterns and behaviours align with knowledge concerning the connectivity of open-drainage lakes and the influence of evaporation on perched basins. Most notably, the loggers detected open-water flooding in the summers of 2018 and 2019 in 2 regions within the Athabasca Delta (Figure A3.1.5). This hydrological process has received relatively little recognition as an important lake recharge mechanism, because spring ice-jam flooding has been a primary focus at the PAD.

Of the water-level loggers installed in 30 lakes across the Athabasca Delta, observed temporal patterns of lake water-level variation fell into four main categories: 1) “Open-water flood” includes lakes that received floodwater from the Athabasca River during the open-water, or summer period, as detected by similar rises of water-level in July and August as the Athabasca River, 2) “Water-level drawdown” includes lakes that experienced steady water-level drawdown, 3) “Steady water-levels” includes lakes with relatively stable water levels during the open-water season, and 4) lakes “inundated by open-water network” received input water from large open-drainage lakes that expanded when increased river flows raised their water levels in June - August (Figure A3.1.5). Alignment between the timing of flood pulses in the summer-flooded lakes with rises in the Athabasca River water level confirmed the source of input water into the lakes to be Athabasca River floodwater. Lake water-levels also responded to evaporative processes and declines were most evident in lakes that did not experience open-water flooding. Input of precipitation was detected during large rain events, especially in mid-August 2019 and mid-July 2018, as small, sharp rises in water levels of several lakes in the categories “Water-level drawdown” and “Steady water-level”. Ability to detect these precipitation events in the “Summer Flood” lakes was likely masked by input of floodwater which occurred at the same time. Analysis of water-level time series plots also revealed unique instances where previously abandoned channels reconnected with nearby lakes. For instance, in 2018, a channel carried water from Lake Claire to lakes M9 and M10, due to sustained high water-levels on Lake Claire. This observation is

supported by mirrored water-level patterns for M9 and M10 in comparison to Lake Claire during the open-water season. Spatial patterns of lake water-level behaviours are currently being explored to identify the spatial coverage of the open-water flooding in 2018 and 2019. Together, lake water isotope measurements and water-level loggers provide a powerful approach for hydrological monitoring of lakes in the PAD, capable of identifying the timing and duration of major hydrological processes and their relative influence on lake levels and water balance across the delta.

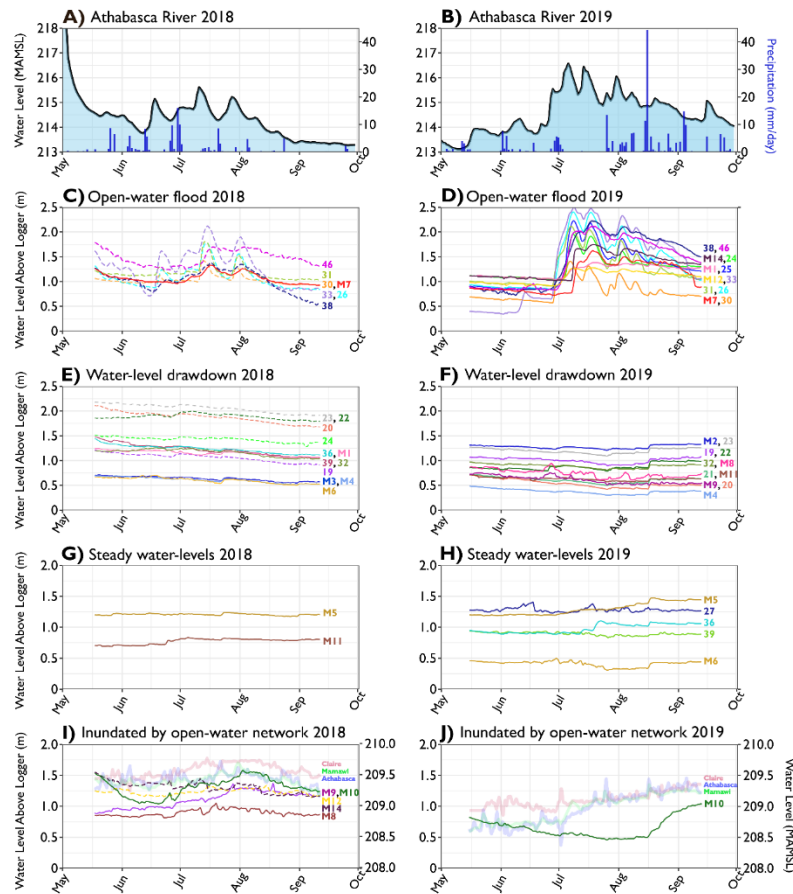


Figure A3.1.5. Time-series water-level plots showing changes in Athabasca River and water-levels of lakes located in the Athabasca Delta from May to October during the open-water seasons of 2018 (left column; panels A, C, E, G, I) and 2019 (right column; B, D, F, H, J). Line colours identify individual lakes located in the Athabasca Delta. Dashed lines identify lakes that received river floodwater in April 2018 after widespread ice-jam flooding. Panels **A** and **B** show the daily water level of the Athabasca River (MAMSL) in 2018 and 2019, measured at Embarras Airport (Station number: 07DD001), and the daily precipitation (mm) measured at Fort Chipewyan Airport (Climate ID number: 3072659). Panels **C** and **D** show water levels (m) for lakes that received river floodwater during the summer seasons. Panels **E** and **F** show water levels (m) for lakes that experienced water-level drawdown during the open-water seasons. Panel **G** and **H** show water levels (m) for lakes that did not experience open-water flooding or marked water-level drawdown. Panels **I** and **J** show water levels (m) for lakes that were inundated by the rising open-water network, including lakes Claire, Mamawi and Athabasca (Station numbers: 07KF002, 07KF003, 07MD001).

A3.1.4.2 Monitoring Component 2: Influence of hydrological processes on water quality

Objective: To utilize seasonal (spring, summer, fall) measurements of water chemistry variables to characterize lake water quality and identify linkages to hydrological processes, including river flooding and evaporative lake-level drawdown.

Information Provided: Water chemistry variables measured include conductivity, pH, turbidity and concentrations of total dissolved nitrogen (TDN), total nitrogen (TN), total phosphorus (TP), total dissolved phosphorus (TDP), dissolved organic carbon (DOC), dissolved inorganic carbon (DIC), chlorine (Cl), sulfate (SO_4), sodium (Na), potassium (K), calcium (Ca), magnesium (Mg), dissolved silica (SiO_2), carbonate (CaCO_3). Multivariate statistical approaches consistently identify three significant gradients across the sampled lakes: conductivity and ionic content, dissolved nutrients and DOC, and flood indicators (including TP, silica, turbidity). These findings were recently reported in Remmer et al. [2018], and previously in Wolfe et al. [2007b] and Wiklund et al. [2012a]. River flood events are expected to reduce conductivity and nutrient concentration, while immediately increasing turbidity and concentrations of TP and dissolved silica. These effects, however, can be short-lived as a result of settling out of suspended river sediment and evaporative drawdown of lake water levels after flooding. In the absence of flooding, concentrations of ions and nutrients increase with increasing evaporative influence.

Results to Date

[1] *Effects of spring ice-jam flooding on lake water quality*

In light of widespread ice-jam flooding in 2014 and subsequent arid conditions during summer of 2015, Remmer et al. [2018] integrated spatial and temporal data to assess current hydrological and water-quality conditions in lakes of the PAD. Although water chemistry from 2014 was not available to verify the effects of flooding on limnological conditions in spring 2014, previous research by Wiklund et al. [2012a] has shown that flooded lakes have similar water chemistry characteristics as river water. Analysis of water chemistry measured at 61 PAD lakes in May, July and September 2015 (one-year post-flooding) across the delta showed that effects of the large-scale ice-jam flood event of 2014

failed to persist beyond the early ice-free season of 2015. By May 2015, water chemistry conditions did not differ significantly between lakes that flooded and did not flood in 2014. The short-lived effects of flooding on limnological conditions on lakes that flooded in 2014 was surprising, because Wiklund et al. [2012a] observed that infrequently flooded lakes in the Peace and Athabasca sectors of the delta took several years after a flood to return to water chemistry values typical closed-drainage conditions characterized by high concentrations of DOC, TKN, bio-available nutrients and ions, and low concentrations of suspended sediments and SO_4 . In contrast, we found that the magnitude of evaporative water loss during the remainder of the 2014 post-flood ice-free season and early 2015 ice-free season leading up to our sampling in May 2015 resulted in a more rapid than predicted return to limnological conditions typical of those found during long periods (many years to decades) without flooding. This is likely the result of unprecedented drying during previous decades, resulting in a limnological trajectory that one rare flood event did not substantially alter.

[2] Spatial and temporal assessment of lake water quality

Ongoing work is combining multivariate statistical analysis and geospatial approaches to generate ‘limnoscapes’ that identify the primary water-quality gradients, their spatial patterns and their linkages to hydrological processes. This will provide fundamental information for individual lakes and regions in the delta, as well as the knowledge needed to detect and track shifts or perturbations of ecosystem condition. Limnoscapes have been generated for each ice-free season (spring, summer, fall) during 2015-2019 (Figure A3.1.6). Results demonstrate marked gradients of conductivity and concentrations of dissolved nutrients and DOC, and flood indicators [e.g., TP, silica, turbidity, based on previous research; Remmer et al. 2018; Wiklund et al. 2012a]. Spatial patterns occur along hydrological gradients across years and seasons. River flooding causes decline in concentrations of ions and dissolved nutrients, and increase of turbidity and TP and silica content. This is most notable in the central Athabasca Delta (Figure A3.1.6). Within one season after flooding and in lakes where flooding has not occurred, evaporative drawdown results in increasing concentrations of ions and nutrients, with consistently high nutrient

content noted in the central Peace and southwestern Athabasca sectors. The linkages between hydrologic processes and limnological conditions are readily apparent when the isoscapes and limnoscapes are compared side-by-side, as shown in Figure A3.1.7, where areas of flooding have low ionic concentrations and dissolved nutrients and high turbidity, TP and silica. Areas where drying is most prevalent, indicated by red zones on the isoscapes, correspond to high concentrations of ions and nutrients and low turbidity, TP and silica, likely due to evaporative concentration and absence of river floodwater.

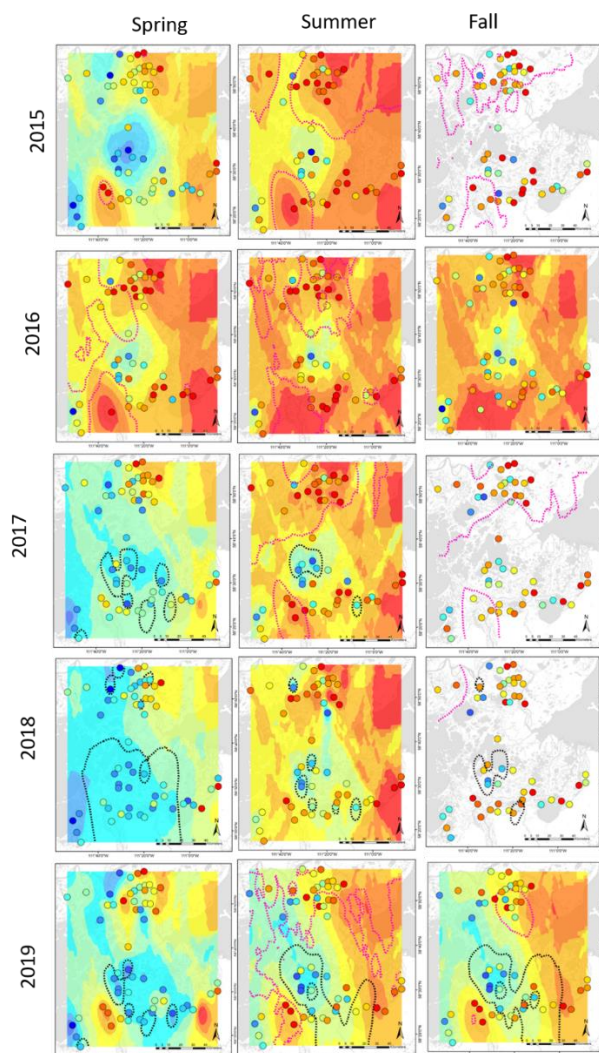


Figure A3.1.6. ‘Limnoscapes’ showing the interpolated values of PCA axis scores, which capture gradients of dissolved nutrients and DOC. Warmer colours indicate high scores (more nutrients/DOC) while darker blue colours indicate low scores (less nutrients/DOC). Pink lines delineate areas of high

evaporative enrichment and black lines delineate areas of river flooding. Significant spatial autocorrelation to justify interpolation was not detected in the fall of 2015, 2017 and 2018.

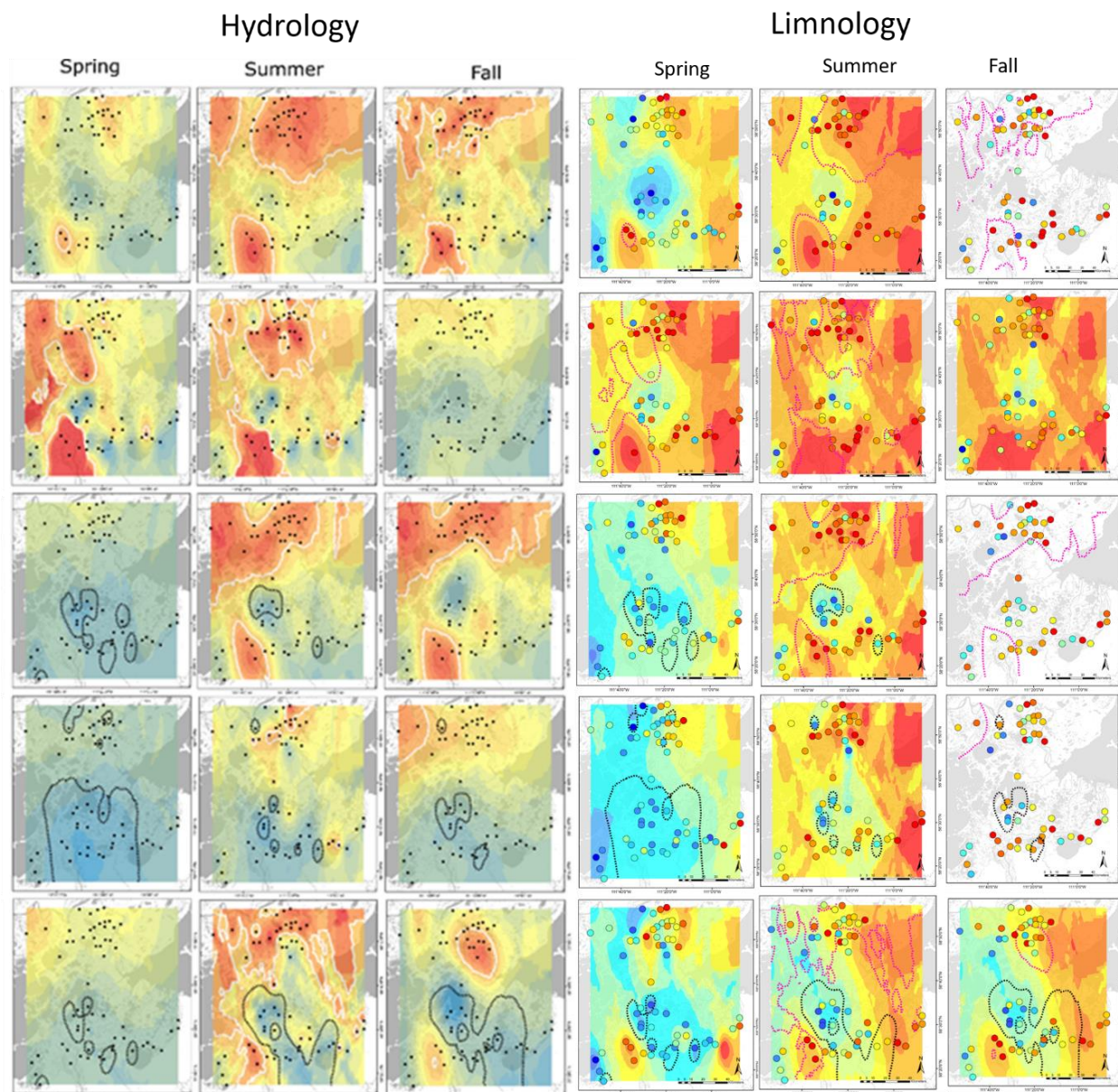


Figure A3.1.7. ‘Isoscapes’ and ‘limnoscapes’ and presented side by side showing the interpolated values of PCA axis scores for limnological variables and evaporation-to-inflow (E/I) ratios across the Peace-Athabasca Delta. Warmer colours in the limnoscapes indicate more nutrients/DOC and evaporation, while darker blue colours indicate less nutrients/DOC and flooding, respectively. White and pink lines delineate areas of high evaporative enrichment and black lines delineate areas of river flooding. Significant spatial autocorrelation of limnological variables to justify interpolation was not detected in the fall of 2015, 2017 and 2018.

A3.1.4.3 Monitoring Component 3: Concentration of metals contaminants in lake surface sediments

Objective: River floodwaters convey contaminants of concern that adhere to sediments suspended by the flowing river water. After flood events, measurements on lake surface sediment can be used to identify the concentration, origin and spatial distribution, and potential enrichment of metals of concern in the PAD. Here, we focus on assessing concentrations of a suite of metals analyzed using ICP-MS (inductively-coupled-plasma mass spectrometry) following aqua-regia digestion of sediment samples to measure the fraction of sediment-bound metals that are most available to aquatic organisms. Such measurement of metals is more cost-effective than for many other classes of contaminants. If elevated metals concentrations are detected, this may indicate a need to also assess other classes of contaminants such as polycyclic aromatic compounds (PACs) or petroleum biomarkers.

Information Provided: Metals concentrations in lake surface sediments are compared to pre-1920 baselines established from analysis of lake sediment cores to assess the degree of enrichment since industrialization. This approach addresses concerns that have been raised over the potential transport of industry-supplied metals to the PAD via the Athabasca and Peace rivers. Metals of concern include Be, Cd, Cr, Cu, Ni, Pb, V, and Zn. Key indicator metals for upstream oil sands development include Ni and V [Wiklund et al. 2014; Kay et al. 2020; Owca et al. 2020].

Results to Date

[1] *Collection of samples of lake surface sediments*

In late April – early May 2018, widespread ice-jam flooding across the Athabasca Delta provided opportunity to sample surface sediment from lakes that were supplied by river floodwaters. Consequently, collection and analysis of the surficial flood layer allowed assessment of the degree of enrichment of contaminants in suspended river sediment, an informative source of river quality assessment. The surficial flood layer of sediment was collected from 20 flooded lakes in July 2018, two months after river flooding along the

Athabasca River. These samples were compared to lake surface sediment collected from 61 lakes spanning the Peace-Athabasca Delta in the previous year (September 2017) when limited flooding occurred. Sector-specific metals concentration baselines (Athabasca and Peace) were derived using sediment deposited in lakes before 1920 (Figure A3.1.8), which includes the period prior to detectable enrichment via atmospheric transport and characterizes the range of natural variation.

To assess for enrichment of metals in recently-deposited sediments, concentrations in lake surface sediment samples were normalized to aluminum, a natural lithogenic element, and then compared to the aluminum-normalized pre-1920 baselines for the Athabasca and Peace sectors. Lake sediment metals concentrations collected in 2017 (Figure A3.1.9) and 2018 (Figure A3.1.10) were not enriched above the pre-1920 baselines, indicating metals concentrations have not increased relative to the range of natural variation. Enrichment factors, expressed as a ratio of the measured Al-normalized metals concentrations to the expected metals concentrations based on the pre-1920 relationship with the normalizing agent (Al), were calculated for metals of concern (Be, Cd, Cr, Cu, Ni, Pb, V, and Zn; Figure A3.1.11).

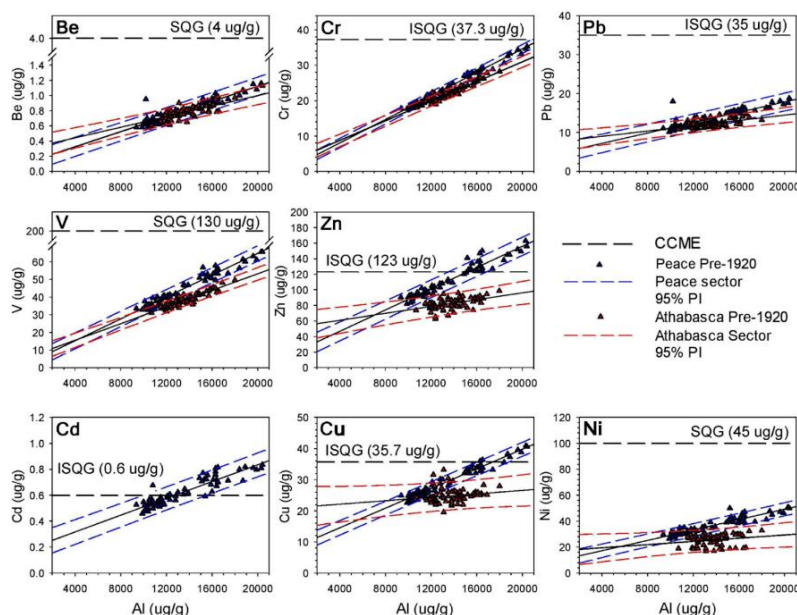


Figure A3.1.8. Cross-plots demonstrating the linear relations between pre-1920 metal concentrations and the normalizing agent (Al). The Peace River 95% PI (blue dashed lines) and regression lines are based on the pre-1920 measurements of metals from PAD 65 and PAD 67 (blue triangles). The Athabasca River 95% PI (red dashed lines) and regression lines are based on the pre-1920 measurements of metals concentrations from PAD 30, PAD 31, PAD 32, and M7 (red triangles). Canadian Council of Ministers of the

Environment (CCME). Interim Sediment Quality Guidelines (ISQG) and Soil Quality Guidelines (SQG) plotted on y-axis denote the guideline concentrations for the metal of concern [from Owca et al. 2020].

The enrichment factors for each metal in the 2017 and 2018 sediment samples and pre-1920 baselines were compared using ANOVA tests to assess for significant differences from the baselines. Numerical analysis demonstrated no evidence for significant enrichment of these metal concentrations in either 2017 or 2018 with respect to their pre-1920 baselines. Collectively, the spatially comprehensive 2017 dataset along with the opportunistic sampling of the 2018 spring-flood deposits provide strong evidence that oil sands development has not yet caused marked enrichment of metals in recently-deposited river-supplied sediment relative to the pre-1920 baselines. These methods provide a foundation for ongoing monitoring of metals in river-derived sediment, and although no enrichment has yet been detected, numerical analyses presented here possess a strong ability (>99% power) to detect minor enrichment of metals in lake surface sediment.

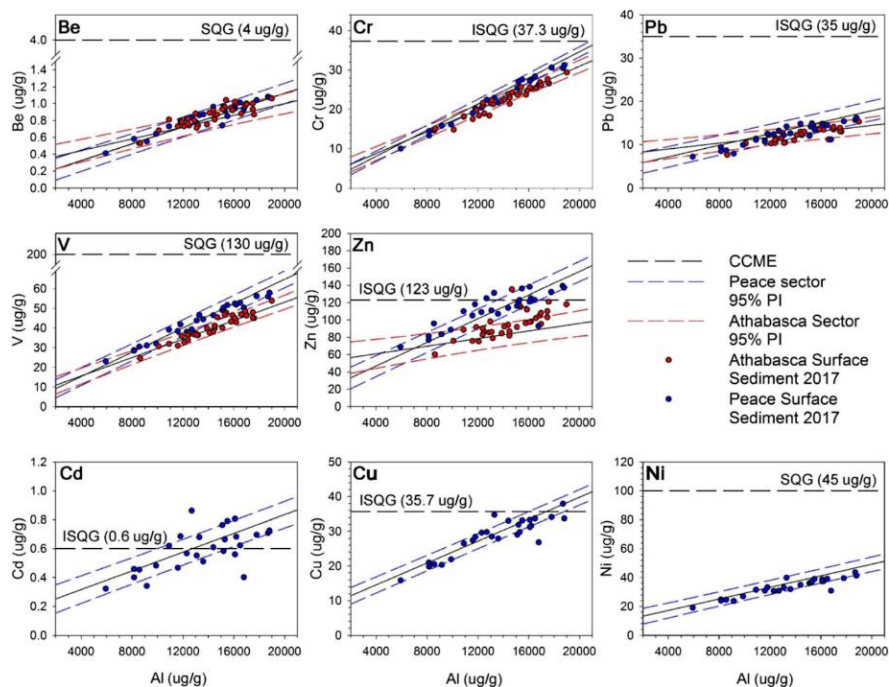


Figure A3.1.9. Metals concentrations from surface sediments (upper ~1 cm) collected in September 2017 from Peace sector lakes (blue circles) and Athabasca sector lakes (red circles) plotted on the pre- 1920 linear regressions and 95% PIs. Canadian Council of Ministers of the Environment (CCME). Interim Sediment Quality Guidelines (ISQG) and Soil Quality Guidelines (SQG) plotted on y-axis denote the guideline concentrations for the metal of concern [from Owca et al. 2020].

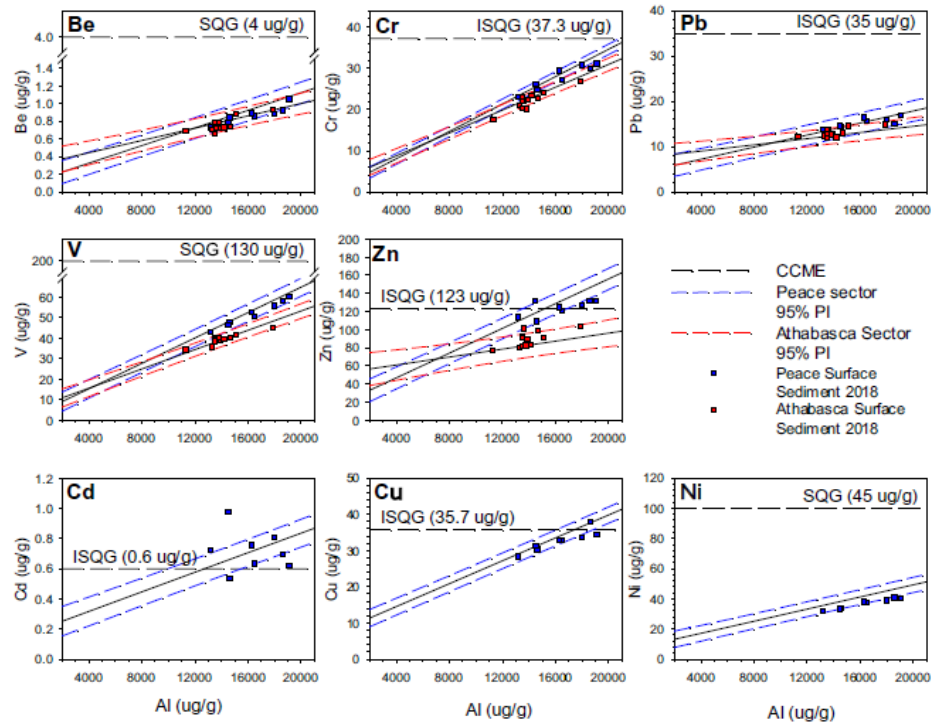


Figure A3.1.10. Metals concentrations from surficial flood-layer sediments collected in July 2018 after spring flooding from Peace sector lakes (blue squares) and Athabasca sector lakes (red squares) plotted on the pre-1920 linear regressions and 95% PIs. Canadian Council of Ministers of the Environment (CCME). Interim Sediment Quality Guidelines (ISQG) and Soil Quality Guidelines (SQG) plotted on y-axis denote the guideline concentrations for the metal of concern [from Owca et al. 2020].

We propose that opportunistic lake surface sediment collection be conducted soon after flood events, coupled with routine water isotope monitoring to track hydrological conditions, to capture freshly deposited river-supplied sediment, as was done in July 2018. This provides a snapshot of river sediment metals concentrations at the time of the spring freshet, which carries contaminants accumulated in the winter snowpack prior to melt in the spring. These results are published in Owca et al. [2020; open access at <https://doi.org/10.1007/s10661-020-8067-y>]. Another publication by Kay et al. (2020) employed a comparable approach to assess temporal patterns of metals enrichment in eight flood-prone lakes in the Athabasca sector (open access at <https://doi.org/10.1016/j.scitotenv.2019.135309>).

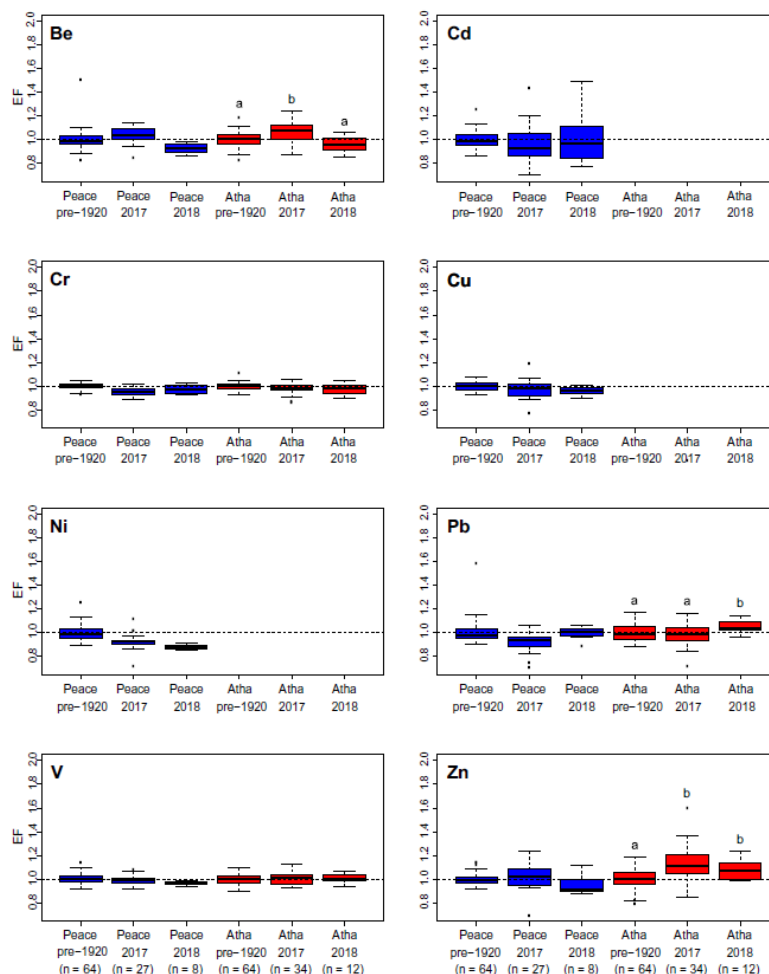


Figure A3.1.11. Boxplots showing the distribution of enrichment factors (EF) for metals (Be, Cd, Cr, Cu, Ni, Pb, V, and Zn) relative to sector-specific pre-1920 baseline concentrations. Enrichment factors were determined for Peace sector (blue) and Athabasca sector (red) surface sediment from 2017 and 2018 sampling campaigns. Black dashed line at an EF value of 1.0 (indicating no enrichment). Letters denote significant ($\alpha = 0.05$) increase in mean EF relative to pre-1920 baseline, as determined from one-way ANOVA and post hoc tests [from Owca et al. 2020].

A3.1.5 Conclusions and Recommendations

Presented here are three components comprising a foundation for monitoring lakes in the Peace-Athabasca Delta, Alberta. A summary of the information provided by metrics measured within each component is outlined within alongside key results generated through the development of this hydro-ecological monitoring framework. Continued monitoring of water balance, water quality and contaminants within lakes in the Peace-Athabasca Delta is essential to track changes across the landscape and over time, which can both provide early warning signs of ecosystem degradation and further protect the ecological integrity of the Peace-Athabasca Delta.

A3.1.6 Standard Operating Procedures

A3.1.6.1 Standard Operations for Component #1: Field & laboratory procedures for conducting hydrological monitoring in the Peace-Athabasca Delta, Alberta

This S.O.P. gives step-by-step instructions for collection of lake water isotope samples and deployment of water-level loggers in 60 lakes across the Peace-Athabasca Delta, Alberta. Water isotope samples should be shipped (unrefrigerated) to University of Waterloo, where they will be prepared by a student, under supervision of Dr. Roland Hall, and analyzed at the University of Waterloo Environmental Isotope Laboratory (UW – EIL) for measurement of $\delta^{18}\text{O}$ and $\delta^2\text{H}$.

This S.O.P. describes:

SOP.1.1 The field and laboratory equipment and forms required.

SOP.1.2 The timing and sequence of sample collection and equipment deployment/retrieval during the fieldwork.

SOP.1.3 Detailed methods on pre-field preparation, safety, field protocols and lab protocols.

SOP.1.4 Post-fieldwork tasks.

A list of routine monitoring site locations, including site names and GPS coordinates are reported in Table SOP.1.1. An additional Excel file was sent with this Technical Report as a template for recording field observations and values obtained during sample collection and processing.

SOP.1.1. Required equipment and forms

SOP.1.1.1. Hydrological monitoring field equipment [for each sampling episode]

- ☐ 75 [per sampling episode] x 30mL high density polyethylene (HDPE) bottles
- ☐ YSI Multi Meter
- ☐ 62 Onset HOBO water level data loggers (SKU: U20-001-04; 0-4 m)
- ☐ 62 pre-assembled rock-sock anchors attached to a wooden float via nylon rope and zip-ties (~300 feet of rope, ~300 regular zip-ties, ~35 pairs of socks)
- ☐ Black sharpie markers (6-10)
- ☐ Colourful electrical tape/ masking tape (2-3 rolls)
- ☐ Clipboard with field sampling notes
- ☐ Pens

- ☐ Camera
- ☐ GPS loaded with lake and river locations (Table SOP1.2)
- ☐ ~6 large Ziploc bags for separating empty and full water-isotope samples
- ☐ Booklet with map of ponds and photos for identification
- ☐ Extra batteries for GPS (AA)

Table SOP.1.1. Recommended helicopter sampling route ('water loops') for effective sample collection.

Loop 1 (12)	Loop 2 (10)	Loop 3 (9)	Loop 4 (12)	Loop 5 (13)	Loop 6 (14)
PAD 1	PAD 9	PAD 53	PAD 37	M11	PAD 40
PAD R11	PAD 8	PAD R9	PAD 45B	M12	PAD 39
PAD 3	PAD 6	PAD 50	M14	PAD 32	PAD 36
PAD 2	PAD 4	PAD 52	R7	R5	PAD 27
M19	PAD 5	PAD R2	PAD 46/M13	PAD 19	PAD 26
M15	PAD 17	PAD 54	PAD 30	PAD 20	PAD 38
M16	PAD 16	PAD 57	PAD 31	PAD 21	M7
M18	PAD R1	PAD 58	M8	R4	PAD R6
M17	PAD 14	PAD R3	PAD 62	PAD 22	M1
PAD 18	PAD 15		M9	PAD 23	M3
PAD 13			M10	PAD 33	M2
PAD 12				PAD 24	M4
				PAD 25	M5
					M6

Table SOP.1.2. Locations of PAD monitoring sampling sites. Coordinates are reported in decimal notation and latitude/longitude (degrees, minutes, seconds).

Site ID	Local Name	Decimal Coordinates	Latitude dd°mm'ss []	Longitude dd°mm'ss []
PAD 01	Devil's Gate	58.8056, -111.2435	58°48'20" N	111°14'37" W
PAD 02	Lake 50	58.8352, -111.3191	58°50'06" N	111°19'09" W
PAD 03	Lake 587	58.8332, -111.2837	58°50'00" N	111°17'01" W
PAD 04	Lake 577	58.8426, -111.4012	58°50'33" N	111°24'04" W
PAD 05	Lake 595	58.8450, -111.4787	58°50'42" N	111°28'43" W
PAD 06	Pushup Lake	58.8116, -111.3866	58°48'42" N	111°23'12" W
PAD 08	Lake 582	58.8116, -111.3557	58°48'42" N	111°21'20" W
PAD 09	Lake 565	58.7745, -111.3293	58°46'28" N	111°19'45" W
PAD 12	Lake 586	58.9556, -111.3290	58°57'20" N	111°19'44" W
PAD 13	Lake 585	58.9482, -111.3752	58°56'53" N	111°22'31" W
PAD 14	Lake 530	58.9332, -111.4275	58°55'59" N	111°25'39" W
PAD 15	Lake 555	58.9501, -111.5069	58°57'00" N	111°30'25" W
PAD 16	Egg Lake	58.8823, -111.4005	58°52'56" N	111°24'02" W
PAD 17	Lake 594	58.8721, -111.4278	58°52'20" N	111°25'40" W
PAD 18	Greenstar Lake	58.8962, -111.3603	58°53'46" N	111°21'37" W
PAD 19	Lake 500	58.4423, -111.6289	58°26'32" N	111°37'44" W
PAD 20	Close to 502	58.4195, -111.5924	58°25'10" N	111°35'32" W
PAD 21	Close to 591	58.3693, -111.5689	58°22'09" N	111°34'08" W
PAD 22	Lake 525	58.3971, -111.4986	58°23'49" N	111°29'55" W
PAD 23	Lake 152	58.3896, -111.4447	58°23'23" N	111°26'41" W
PAD 24	Lake 159	58.3941, -111.3601	58°23'39" N	111°21'36" W
PAD 25	Blanche Lake	58.3849, -111.3297	58°23'06" N	111°19'47" W
PAD 26	Lake 160	58.4199, -111.2695	58°25'12" N	111°16'10" W
PAD 27	Lake 147	58.4358, -111.2621	58°26'09" N	111°15'44" W
PAD 30	Mamawi Creek Pond	58.5101, -111.5171	58°30'36" N	111°31'02" W
PAD 31	Johnny Cabin Pond	58.4961, -111.5184	58°29'46" N	111°31'06" W
PAD 32	Lake 590	58.4973, -111.4432	58°29'50" N	111°26'36" W
PAD 33	Dagmar Lake	58.4203, -111.4402	58°25'13" N	111°26'25" W
PAD 36	Lake 127	58.4660, -111.2510	58°27'58" N	111°15'04" W
PAD 37	Jemis Lake	58.6710, -111.4453	58°40'16" N	111°26'43" W
PAD 38	Richardson Lake	58.4175, -111.1228	58°25'03" N	111°07'22" W
PAD 39	Lake 123	58.4649, -111.1856	58°27'54" N	111°11'08" W

PAD 40	Grey Wavy Lake	58.5077, -111.1913	58°30'28" N	111°11'28" W
PAD 45B	Mamawi Lake	58.5842, -111.3777	58°35'03" N	111°26'39" W
PAD 46/M13	Otter Lake	58.5504, -111.5476	58°33'01" N	111°32'51" W
PAD 50	Lake 21	58.8589, -111.8893	58°51'32" N	111°53'21" W
PAD 52	Close to Lake 512	58.8771, -111.7505	58°52'38" N	111°45'02" W
PAD 53	Baril Lake	58.7723, -111.6488	58°46'20" N	111°38'56" W
PAD 54	Horseshoe Slough (223)	58.8697, -111.5777	58°52'07" N	111°34'40" W
PAD 57	Lake 390	58.8286, -111.5945	58°49'43" N	111°35'40" W
PAD 58	Lake 540	58.8297, -111.5523	58°49'47" N	111°33'08" W
PAD 62	Lake Claire	58.3719, -111.8506	58°22'19" N	111°51'02" W
M1	Big Egg Lake	58.4476, -111.0151	58°26'51" N	111°00'54" W
M2	Marta's Lake	58.4183, -110.9137	58°25'06" N	110°54'49" W
M3	Frezie Lake	58.4314, -110.9727	58°25'53" N	110°58'22" W
M4	Johnny's Lake	58.4919, -110.8438	58°29'31" N	110°50'38" W
M5	Lake South of Tokyo Snye	58.5025, -110.7931	58°30'09" N	110°47'35" W
M6	Rat Island	58.5334, -110.7939	58°32'00" N	110°47'38" W
M7	Ross's Corner	58.4362, -111.0467	58°26'10" N	111°02'48" W
M8	Frog Lake	58.3992, -111.8041	58°23'57" N	111°48'15" W
M9	Welstead Lake	58.3275, -111.8223	58°19'39" N	111°49'20" W
M10	Whitefish Lake	58.2893, -111.7918	58°17'22" N	111°47'30" W
M11	Donald's Lake	58.5501, -111.3325	58°33'00" N	111°19'57" W
M12	Lake Beside Spoonbill Creek	58.5563, -111.4135	58°33'23" N	111°24'48" W
M14	Willow Lake	58.5714, -111.5199	58°34'17" N	111°31'12" W
M15	Balloon Lake	58.8815, -111.2569	58°52'53" N	111°15'25" W
M16	Last Rat Lake	58.8943, -111.2091	58°53'39" N	111°12'33" W
M17	Mud Lake A	58.9026, -111.3222	58°54'09" N	111°19'20" W
M18	Mud Lake B	58.9055, -111.2894	58°54'20" N	111°17'22" W
M19	Rat Lake	58.8845, -111.3232	58°53'04" N	111°19'24" W
PAD R1	Revillon Coupé	58.90778, -111.4161	58°54'28" N	111°24'58" W
PAD R2	Peace River (at Rocky Point)	58.9086, -111.5867	58°54'31" N	111°35'12" W
PAD R3	Chenal des Quatre Fourches	58.8147, -111.5642	58°48'53" N	111°33'51" W
PAD R4	Athabasca River	58.3550, -111.5397	58°21'18" N	111°32'23" W
PAD R5	Embarras River	58.4903, -111.4372	58°29'25" N	111°26'14" W
PAD R6	Fletcher Channel	58.4861, -111.0644	58°29'10" N	111°03'52" W
PAD R7	Mamawi Creek	58.5433, -111.4939	58°32'36" N	111°29'38" W
PAD R9	Claire River	58.8583, -111.9100	58°51'30" N	111°54'36" W
PAD R11	Rivière des Rochers (near Chilloney Creek)	58.8214, -111.2808	58°49'17" N	111°16'51" W

SOP.1.1.2. Forms

The hydro-limnological monitoring field notes template (Figure SOP.1.1.) can be used for recording field observations and in-situ measurements using the YSI multi meter.

SOP.1.2 Timing and sequence of events

SOP.1.2.1 Seasonal schedule of sampling periods

This sampling takes place three times during the open-water season; typically, in mid- May, July, and early September. The purpose of sampling three times is to 1) capture the lake water isotope signatures directly after spring input water (May), which would also capture any flood events (and serves as a benchmark against which influence of summer precipitation and evaporation can be evaluated), 2) during the time of peak evaporation and following open-water flood events (late-July), and 3) before the lakes freeze up (September). The exact sampling dates within May, July, and September are not critical, although sampling earlier in July may miss capturing summer flood events in the Athabasca Delta. Typically, sampling

[YEAR] PAD Sampling Field Notes – Month (circle): May July Sept

Lake ID: _____ Date & Time: _____

Sampling Crew: _____

Lake ID: _____ Date & Time: _____
 Sampling Crew: _____
 Weather (light, rain, wind, etc.): _____
 Water-level logger deployed? (Circle) Yes / No Water depth at water logger (m): _____
 Water-level logger Serial #: _____ Logger GPS coordinates: _____
 YSI: Temp (°C): _____ Pressure (mmHg): _____ DO %: _____ Cond (µS/cm): _____
 pH: _____ Turbidity (FNU): _____ TSS (mg/L): _____ Depth (m): _____
 Lake Colour/Turbidity: _____
 Lake description (hydrology, algal blooms, water clarity/colour, etc.): _____
 Catchment description (vegetation, slope, unusual features, etc.): _____
Macrophytes – Type and cover (Circle): Benthic-continuous Benthic-discontinuous Emergent stands
 Floating –continuous Floating - discontinuous
 Macrophytes – Other notes: _____
Notes (use other side if needed): _____
Filtering Information (at lab/field base):
 Chlorophyll filtered volume: _____ ml
 HPLC filtered volume: _____ ml
 TSS filtered volume: _____ ml TSS Filter ID# or label: _____

SOP.1.2.2 Length of sampling

For each hydrological monitor

bit longer when water-level data loggers are deployed and retrieved. Typically, sampling can be performed over two days (weather permitting). An example sampling route can be found in Figure SOP.1.2.

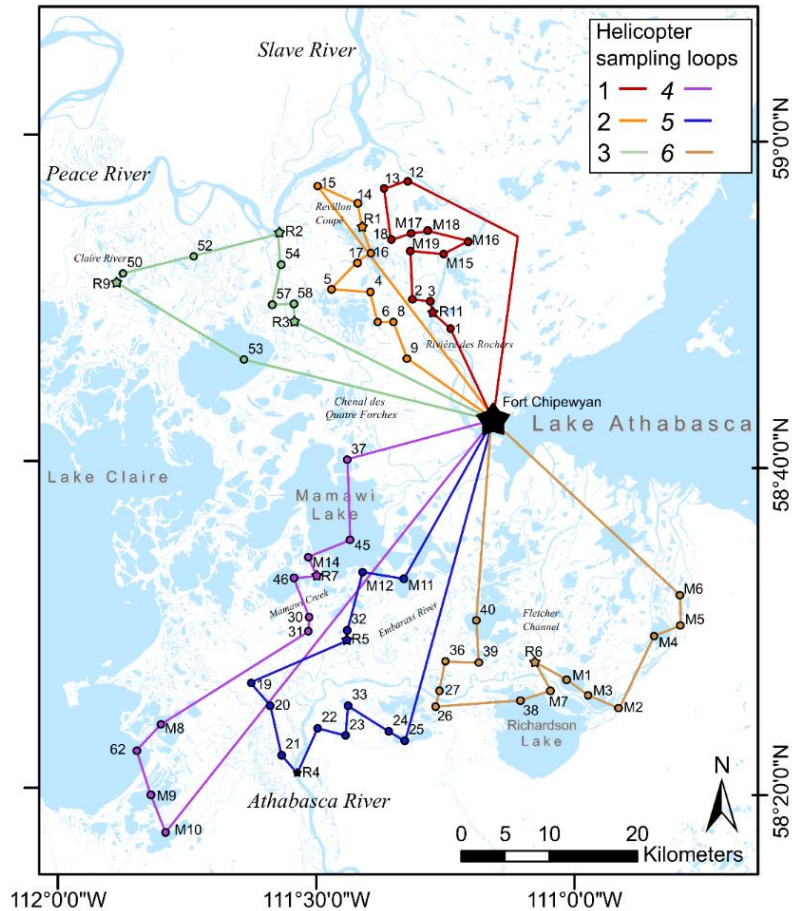


Figure SOP.1.2. Location of sampling sites within their respective ‘water loop’. Refer to Table SOP.1.3 for the order of recommended helicopter sampling loops. Figure was created by Laura Neary using ArcGIS Desktop version 10.7.1.

SOP.1.2.3 Field preparation for hydrological monitoring (May, July, and September)

- 1) Label 75 x 30 mL HDPE bottles using colourful electric tape. Label “PAD ____ and Month/year” (lake name will be filled just before sample collection, upon descent to the lake).
- 2) Place bottles into large Ziploc bag with a black sharpie marker. Label a second large Ziploc bag, “FULL Hydrological Monitoring Samples”, for filled water bottles in the field.
- 3) Prepare ‘emergency supply kit’ with spare batteries, pens/pencils, Sharpies, and tape.
- 4) Check [YSI Multi Meter](#) calibration and ensure battery is charged. Click [here](#) for a quick calibration guide.
- 5) Ensure GPS has correct coordinates and batteries are fully charged.

6) Prepare field notes clipboard with hydrological monitoring field notes in the order listed in the sampling route pertaining to the ‘water loops’.

May only: In advance of field sampling, individually plug the 62 water level loggers into a computer to set for deployment. Set deployment date and time the same for all loggers (e.g., May 13 at 09:00). Label 2 water level loggers using duct tape and permanent marker “AIR LOGGER” and leave outside in two different locations (safe, out of reach, outside of a house) for the duration of the summer in order to measure ambient air temperature and pressure at the same logging interval as loggers deployed in lakes (we have placed one Air Logger on a nail in the wall of the porch of the Brown TU next to the Park Office and Robert Grandjambe has placed a second one at his cabin at Devil’s Gate Pond (PAD1)).

Instructions for water-level logger set up:

Collect rocks from Fort Chipewyan rock pile (near the airport; approximate GPS coordinates: 58.0763191°N, -111.118167°W). Try to select rocks without pointy sharp edges. Stuff 2-3 large rocks into each sock and tie a knot at the open end of the sock. Cut rope lengths according to lake depths from previous season (considering fluctuations in water levels; estimates reported in Table SOP.1.3). Sharper rocks may rip small holes in sock. If this happens use duct tape to cover the small holes. Determine how many rope lengths needed for each category: 0.5m, 1.0 m, 1.5 m, 2.0 m, 2.5 m, 4.0 m and 8.0 m. Use scotch tape and sharpie to colour-code the different lengths of rope. Obtain wood planks that are roughly 1.5 feet long and drill holes in the middle (hole size specific to the diameter of the nylon rope). Spray paint the wood planks pink and wrap pink insulation foam (floating material) around either end securing with duct tape. Attach the different lengths of rope through the hole in the float and tie a knot on the top, then on the other end attach a rock-sock using two overhand knots. Fasten the logger to the rock sock by sliding a zip-tie through the knot in the rock-sock and through the black hole on the logger. See Figure SOP.1.3 for the typical water-level logger deployment set up.

SOP.1.2.4. Day-before tasks

Gather all required supplies for the first helicopter day and place them in one spot, ready to be checked and grabbed the morning prior to sampling:

Checklist 1-1. Packing lists for helicopter sampling

- ☐ GPS
- ☐ Camera
- ☐ YSI Multi Meter
- ☐ Clipboard + field sampling notes
- ☐ Sharpies
- ☐ Ziploc bag with empty sample bottles
- ☐ Ziploc bag for full sample bottles
- ☐ Emergency supply kit

May only:

- ☐ Water level loggers
- ☐ List of lake depths for water logger rope lengths (transfer specified anchor rope lengths from Table SOP.1.3 onto individual field site sampling notes)
- ☐ Rock-socks attached to wooden floats
- ☐ Extra zip-ties

September only:

- ☐ Knife or shears
- ☐ Zip-lock bags for storing water-level loggers in after collection
- ☐ Garbage bag for ripped socks and zip-ties

Table SOP.1.3. Suggested rope lengths (m) for deploying water-level loggers in PAD monitoring lakes. Note: loggers are only deployed in lake sites (black text), not river sites (grey text). These values were estimated using maximum and minimum lake water-levels recorded in 2018 and 2019.

Loop 1	Loop 2	Loop 3	Loop 4	Loop 5	Loop 6
PAD 1 (2.0)	PAD 9 (1.5)	PAD 53 (2.5)	PAD 37 (1.5)	M11 (2.0)	PAD 40 (2.0)
PAD R11	PAD 8 (2.5)	PAD R9	PAD 45B (2.0)	M12 (2.5)	PAD 39 (2.5)
PAD 3 (2.0)	PAD 6 (2.5)	PAD 50 (2.0)	M14 (2.5)	PAD 32 (2.5)	PAD 36 (2.0)
PAD 2 (2.0)	PAD 4 (2.5)	PAD 52 (2.0)	R7	R5	PAD 27 (2.5)
M19 (2.0)	PAD 5 (2.5)	PAD R2	PAD 46/M13 (3.0)	PAD 19 (2.0)	PAD 26 (3.0)
M15 (2.0)	PAD 17 (2.0)	PAD 54 (4.0)	PAD 30 (2.5)	PAD 20 (2.0)	PAD 38 (3.0)
M16 (2.5)	PAD 16 (2.0)	PAD 57 (2.5)	PAD 31 (2.5)	PAD 21 (2.0)	M7 (2.5)
M18 (2.5)	PAD R1	PAD 58 (2.5)	M8 (2.5)	R4	R6
M17 (2.5)	PAD 14 (2.5)	PAD R3	PAD 62 (3.0)	PAD 22 (2.5)	M1 (2.5)
PAD 18 (9.0)	PAD 15 (2.5)		M9 (2.5)	PAD 23 (3.0)	M3 (2.5)
PAD 13 (2.5)			M10 (2.5)	PAD 33 (3.0)	M2 (3.0)

SOP.1.2.6 Upon return to Fort Chipewyan – same day as sampling

Ziploc bag of hydrological samples (water isotope samples in 30 ml HDPE bottles) do not need to be refrigerated. Samples can be unpacked from Ziploc bag and packed for shipping to University of Waterloo (There is no required processing or laboratory work).

SOP.1.3. Hydrological monitoring field protocol

SOP.1.3.1. Hydrological monitoring sampling protocol [repeated at each sampling site]

- a. Identify the lake or river site using GPS coordinates.
- b. Take a photo of the site name on the field sampling notes.
- c. Upon descent to each site, take photos of the site from the helicopter.
- d. Prep 30 mL bottle in helicopter → write site name on bottle and place it in an easily accessible location to grab when at the lake (i.e., pocket).
- e. Once landed, take photos of site (three shots from left to right, covering the whole lake area).
- f. Once landed on the site, take 30 mL bottle and rinse with lake water three times. Fill to very brim and tightly cap [i.e., tight enough to prevent leakage and evaporation!]. Place in Ziploc bag labelled “filled bottles”.
- g. Turn on YSI, remove from calibration cup and submerge probe into water making sure that it is not touching sediment. Wait until numbers stabilize before recording. Record values of YSI measurements in field notes (Temperature, dissolved oxygen, conductivity, total dissolved solids, pH, turbidity, total suspended solids).
- h. Fill out remaining observational notes (Figure SOP.1.1 and section SOP.1.3.2)

Water-level logger deployment (lakes ONLY!)

May only: Confirm depth of lake and select logger set-up according to the pre-determined anchor rope length (Table SOP.1.3). Ensure the knots are fastened and zip-tie is pulled tight, then deploy into the lake. Record exact GPS coordinates of where logger was deployed on field sampling notes. See Figure SOP.1.3 for deployment design.

September only: Collect water logger by directing the pilot to the pink wooden float and lifting it out of the water onto the helicopter pontoon. Detach logger by snipping the zip tie using a knife or shears. Store water level logger in a pre-labelled zip-lock (front and back in case the permanent marker rubs off), cut into the rock-sock then dump the rocks into the lake. Put the sock and zip-ties in a garbage bag.

SOP.1.3.2 Filling in field notes: recording observations and YSI-multimeter measurements

Fill in the field sampling sheets for each lake/river site (Figure SOP.1.1).

Lake/River ID: site name (i.e. PAD 15)

Date: record the month, day, and year (DD-MM-YYY)

Time: record time of arrival at lake in 24-hour clock (hh:mm)

Sampling Crew: identify who is collecting samples.

Weather: record precipitation, wind strength and direction, cloud cover.

YSI Multi Meter:

Temperature: record in degrees Celsius (°C).

Dissolved Oxygen: record DO % and mg/L.

Conductivity: record in micro Siemens per centimeter ($\mu\text{S}/\text{cm}$).

Total Dissolved Solids (TDS): record in mg/L.

pH: record unit-less value.

Turbidity: recorded in FNU.

Total Suspended Solids (TSS): record in mg/L.

Water depth: record in meters (m); approximate depth at sampling point.

Lake Colour/Turbidity: record colour and visual turbidity of the water column.

Lake Description: record any useful information about the hydrology of the lake including channel flow, shoreline evidence of drawdown.

Catchment Description: describe features such as catchment vegetation, slope, unusual features.

Macrophytes: circle the most accurate description of type and cover from the options listed and provide any additional notes below.

SOP.1.4. Post-collection processing and storage

SOP.1.4.1. Post-collection tasks and procedures

1) Data Entry: Transfer data from field note sheets to the excel file template provided by research partner (U. Waterloo). Although all the data will be transcribed to a datasheet or a computer spreadsheet equivalent, original field notes should be preserved for at least one year, and preferably indefinitely as part of the record.

In September only, connect each logger to a computer, stop the logger from recording further, and transfer the data recorded on each logger to an electronic database

2) Shipping samples: Water isotope samples should be packaged and shipped to research partner: Professor Roland Hall, Department of Biology, University of Waterloo, 200 University Ave. W, Waterloo, ON, N2L 3G1. Send an email (rihall@uwaterloo.ca) to the research partners regarding the shipment and attach the field notes file.

3) Inventory: Do a complete inventory of all the supplies and store in an excel spreadsheet for easy access to order for next season

4) Storage: Package and store all the supplies back into storage room in an organized fashion

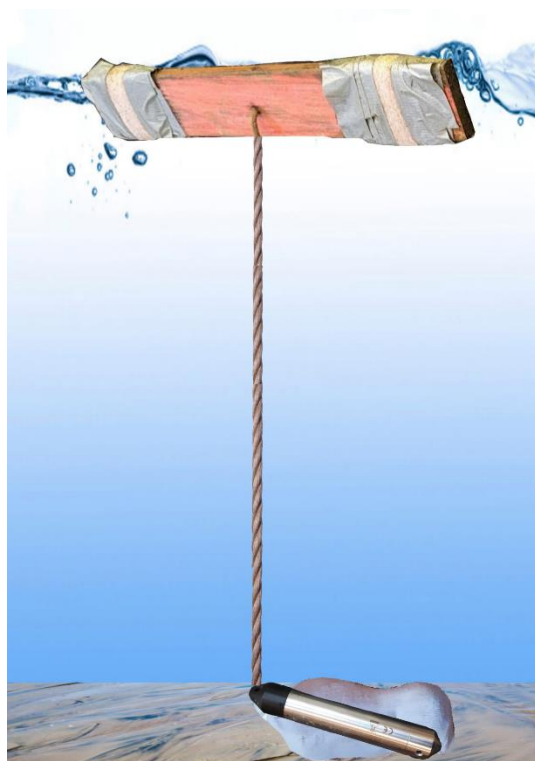


Figure SOP.1.3. Schematic of water level logger deployment showing the wooden float, sock filled with rocks and water-level logger.

A3.1.6.2 Standard Operations for Component #2: Field & laboratory procedures for monitoring water quality of lakes & rivers in the Peace-Athabasca Delta

This S.O.P. gives step-by-step instructions for the collection and processing of lake water samples for various water-quality variables (dissolved and total water chemistry variables) at 60 lakes and 9 river sites across the Peace-Athabasca Delta, Alberta.

This S.O.P. describes:

SOP.2.1 The field and laboratory equipment required.

SOP.2.2 The timing and sequence of sample collection.

SOP.2.3 Detailed methods on pre-field preparation, safety, field protocols and lab protocols.

SOP.2.4 Post-field work tasks.

A list of routine monitoring site locations, including site names and GPS coordinates are reported in Table SOP.1.2. An additional Excel file was sent with this Technical Report as a template for recording field observations and values obtained during sample collection and processing.

SOP.2.1. Required equipment and forms

SOP.2.1.1. Limnological monitoring field equipment [per sampling episode]

- ☐ 75 x 1L high density polyethylene (HDPE) Nalgene bottles
- ☐ 75 x 4L high density polyethylene (HDPE) Carboys
- ☐ YSI multi meter
- ☐ 6-10 Black sharpie markers
- ☐ Duct tape (Approximately 2 rolls)
- ☐ Camera
- ☐ Clipboard with field sampling notes
- ☐ Pens
- ☐ GPS loaded with lake and river locations (Table SOP.1.2)
- ☐ 2 regular-sized coolers (~45 L) with the lids removed (helps organize samples in the back of the helicopter)
- ☐ Booklet with map of ponds and photos for identification
- ☐ Extra batteries (AA) for GPS

SOP.2.1.2. Forms

The hydro-limnological monitoring field notes template (Figure SOP.1.1) can be used for recording field observations and in-situ measurements using the YSI multi meter.

SOP.2.2 Timing and sequence of events

SOP.2.2.1 Seasonal schedule of sampling periods

This sampling takes place three times during the open-water season; typically, in mid- May, late July, and early September. The purpose of sampling three times is to 1) capture the lake water isotope signatures directly after spring input water (May), which would also capture any flood events (and serves as a benchmark against which influence of summer precipitation and evaporation can be evaluated), 2) during the time of peak evaporation and following open-water flood events (late-July), and 3) before the lakes freeze up (September). The exact sampling dates within May, July, and September are not critical, although sampling earlier in July may miss capturing summer flood events in the Athabasca Delta. Typically, sampling has occurred within the first 2 weeks of May and July and the second week of September. Based on experience in 2018 and 2019, we advocate for sampling later in July to capture open-water flooding after Athabasca River flows peak.

SOP.2.2.2 Length of sampling

At each lake site, sampling will take 5-7 minutes. Sampling can be performed over 2-3 days (weather permitting), and is usually performed at the same time as the hydrological sampling of Component 1 (Hydrology). An example sampling route can be found in Figure SOP.1.2.

SOP.2.2.3 Field preparation for limnological monitoring (May, July, and September

Before field [likely by University partner]):

1) At least ~1 month in advance, combust (#sites +10%) GF-C Filters at 550 °C in a muffle furnace, then weigh them to 4 decimal points (i.e. 0.0001). Put each filter into its own mini Whirl-Pak bag then on the outside of the mini Whirl-Pak bag, write the following information on the Whirl-Pak using sharpie:

TSS
Weight of combusted filter (g)
Filter ID (e.g., year-number; 2020-01)

2) Enter the weight of the combusted GF-C filters into an excel spreadsheet including the Filter ID's (a unique number such as 202X-YY). This is necessary in case these numbers rub off the Whirl-Pak bag.

3) ~ 1 month in advance, order materials from BASL (U. Alberta) to be shipped to the PAD for water chemistry analyses (Checklist 2-2 for BASL list of materials)

In field:

1) Apply duct tape around 75 x 1L Nalgene bottles and around the handle of 70 x 4L Carboys.

2) Prepare 'emergency supply kit' with spare batteries, pens/pencils, Sharpies, and tape.

3) Check [YSI Multi Meter](#) calibration and ensure battery is charged. Click [here](#) for a quick calibration guide.

4) Ensure GPS has correct coordinates and batteries are fully charged.

5) Prepare field notes clipboard with hydro-limnological monitoring field notes in the order listed in the sampling route pertaining to the 'water loops'.

6) Remove the lid from 2 standard-sized coolers.

7) At field base, set up 'water filtering' station using materials listed in Checklist 2-2.

SOP.2.2.4. Day-before tasks

Gather all required supplies for the first helicopter day (3-4 water loops) and place them in coolers, ready to be checked and grabbed the morning prior to sampling:

Checklist 2-1. Packing lists for helicopter sampling

- ☐ GPS
- ☐ Camera
- ☐ YSI Multi Meter
- ☐ Clipboard + field sampling notes
- ☐ 6-10 Sharpies
- ☐ Coolers with (# of sites on water loop + 2 extras) 1 L Nalgenes and 4L Carboys
- ☐ Emergency supply kit
- ☐ Depth reader
- ☐ Tape measure with weight attached to the end (lock or washers)

Checklist 2-2. List of materials required for filtering water at field base

Materials required per site (+10% of sites for duplicates):

- ☐ 2 circular 47-mm diameter GF-F filters (sigmaaldrich.com; WHA1825025) for chlorophyll a and HPLC analysis

- ☐ 1 pre-combusted, pre-weighed 47-mm diameter GF-C filter (WHA1822047) for TSS stored in labelled mini Whirl-Pak
- ☐ 3 x aluminum foil squares (cut into 3 in x 3 in squares) to store filters in

Materials required for all samples:

- ☐ Filter tweezers
- ☐ Sharpies
- ☐ Filter stands + canisters + filter holders
- ☐ Vacuum pump
- ☐ Tubing and valves to hook up 4 filter stands to vacuum pump
- ☐ 4 x 500 mL plastic graduated cylinders
- ☐ Cheese cloth sieve
- ☐ 500 mL wash bottle

Materials to be ordered from U. Alberta – Biogeochemical Analytical Service Laboratory (BASL; per site):

- ☐ 2 plastic centrifuge tubes
- ☐ 1 x 500 mL plastic bottle
- ☐ 1 x 250 mL plastic bottle
- ☐ 1-2 x 60 mL syringe (can use 2-3 for filtering **all** samples)
- ☐ 50% nitric acid + plastic pipette dropper
- ☐ 1-2 x Sartorius syringe filter (0.45 µm)
- ☐ Cellulose acetate filter

SOP.2.2.5 Upon return to Fort Chipewyan – same day as sampling

All samples for water chemistry analysis **must be refrigerated** immediately after returning to the processing site (Parks Canada Transient Unit). All 70+ samples will fill approximately 2 fridges, so ensure extra fridges located in WBNP Garage are plugged in.

SOP.2.3. Limnological monitoring field protocol

2.3.1. Limnological monitoring sampling protocol

- a. Identify the next lake or river site using GPS coordinates
- b. Upon descent to the site, take photos of the site from the helicopter.
- c. Once landed, take photos of site (three shots from left to right, covering the whole site area).
- d. Prepare 1 Nalgene and 1 Carboy– write site name in two places with a sharpie in case one wipes off. Place it in an easily accessible location to grab when at the site (beside your feet).

- e. Once landed on the site, exit the helicopter and from the pontoon, rinse both the Nalgene and Carboy 3x with site water. Be careful the caps don't fly away from the wind-current of the helicopter! Begin filling the Carboy - submerge it 10-15 cm below surface of water, once filled half-way set it on the pontoon of the helicopter and use the washed 1 L Nalgene to fill the rest of the Carboy up. Fill it to the brim, ensuring no air bubbles are in the container. Trick for screwing on the cap: twist it in the opposite direction and feel a 'click' before tightening. BE CAREFUL: Once the cooler becomes filled with Carboys (samples stacked on top of each other), if the caps are not on tightly and threaded properly, you could lose all the water from that sampling site!
- f. Submerge the top of the yellow depth measuring device to measure the lake/river depth. If the area has dense submerged macrophytes, it may be best to use a tape measurer with a weight attached to the end.
- g. Turn on YSI, remove from calibration cup and submerge probe into water making sure that it is not touching sediment. Wait until numbers stabilize before recording (wait approximately the same amount of time for each sampling site. Record values of YSI measurements in field notes (Temperature, dissolved oxygen, conductivity, total dissolved solids, pH, turbidity, total suspended solids).
- h. Fill out remaining observational notes (Figure SOP.1.1 and section SOP.2.3.2)

Note: Field sampling is performed best with 3 field members. The field member sitting in the front of the helicopter holds the camera, YSI, GPS and field sampling notes, and completes steps a, b, c, g and h (while remaining in the helicopter). In the back seat of the helicopter, the other two field members will complete step d while approaching the sampling site, then steps e and f after the helicopter has landed. Observations made by the two field members exiting the helicopter (i.e. macrophyte cover) can be conveyed over the radio to the member recording observations in the front seat.

SOP.2.3.2 Filling in field notes: recording observations and YSI-multimeter measurements

Fill in the field sampling sheets for each lake/river site (Figure SOP.1.1).

Lake/River ID: site name (i.e., PAD 15)

Date: record the month, day, and year (DD-MM-YYY)

Time: record time of arrival at lake in 24-hour clock (hh:mm)

Sampling Crew: identify who is collecting samples.

Weather: note things like precipitation, wind strength and direction, cloud cover.

YSI Multi Meter:

Temperature: record in degrees Celsius (°C).

Dissolved Oxygen: record DO % and mg/L.

Conductivity: record in micro Siemens per centimeter (µS/cm).

Total Dissolved Solids (TDS): record in mg/L.

pH: record unit-less value.

Turbidity: recorded in FNU.

Total Suspended Solids (TSS): record in mg/L.

Water depth: record in meters (m); approximate depth at sampling point.

Water Colour/Turbidity: record colour and visual turbidity of the water column.

Site Description: record any useful information about the hydrology of the lake including, connections, channels, evidence of drawdown.

Catchment Description: describe features such as catchment vegetation, slope, unusual features.

Macrophytes: circle the most accurate description of type and cover from the options listed and provide any additional notes below.

SOP.2.4. Post-collection processing and storage

SOP.2.4.1. Post-collection tasks and procedures

1) Set up water filtering station similar to Figure SOP.2.1.

2) Filter water samples quickly and promptly starting the first day samples are collected.

Samples do not need to be processed on the day of collection (it is not possible), but try to work quickly and filter samples within 2 days of collection. While filtering, ensure the water samples do not reach air temperature by keeping a batch (5-6 samples) in a cooler while filtering one at a time. Make sure you utilize water collected in the 1 L Nalgene bottle and 4 L Carboy. Use flowchart located in Table SOP2.1 to divide water filtering tasks between 2 people.

Table SOP.2.1. Descriptions of optimized tasks for effective water filtering using 2 field members, one for managing the filter units and one for measuring water. Table was created by Eva Mehler.

Filter Person	Water Person
1. Prepare Filters: • 2 x GF/F for Chl a and HPLC • 1 x GF/C for TSS from Whirl-Pak bags • 1 x Sartorius cellulose acetate for Water Chem → Pre-rinse with dH₂O and ~ 10 ml of pre-screened site water, discard this first filtrate 2. Ask for water volumes according turbidity (250 – 1000 ml) for Chl a, HPLC and TSS → pump pressure gauge around 300 psi → adjust water volumes according to colour of filters 3. Transfer ~ 300 ml of filtrate from Chl a filter to water chem filter 4. Fold Chl a, HPLC and TSS filter papers → cover in aluminium foil and label with #, Filter Type, Date, Water Amount (and filter ID and weight for TSS) → put TSS filter into Whirl-Pak bags and label bag with all information 5. Put filters and NO₂ + NO₃-tube into freezer 6. Record volumes filtered as well as TSS filter ID and weight on fieldnote sheet	1. Label bottles/tubes: • 500 ml – #, TOTAL, Date • 250 ml – #, DISSOLVED, Date • 30 ml – #, ISOTOPES, Date • Centrifuge tubes: ○ #, MAJOR CATIONS, Date ○ #, NO₂ + NO₃, Date 2. Shake bottle and carboy, rinse measuring cylinders with pre-screened site water, fill cylinders with asked water volumes 3. Fill bottles: • 30 ml: raw lake water (bulging out) • 500 ml: rinse and fill with screened water • 250 ml: filtrate from acetate filter and put 250 ml and 500 ml bottle into cooler 4. Prepare syringe filter: • Rinse filter with a few ml of dH₂O first • Fill syringe with ~ 30 ml of screened site water • Squirt first ~ 5 ml into sink, then fill each centrifuge tubes with 12 ml of syringe filtrate • Add two drops of 50% nitric acid to major cations-tube and put into cooler 5. Empty filter beakers, clean Chl a and water chem beakers with tap and dH₂O

3) Data Entry: Transfer data from field note sheets to the excel file template provided by research partner (U. Waterloo). Although all the data will be transcribed to a datasheet or a computer spreadsheet equivalent, original field notes should be preserved for at least one year, and preferably indefinitely as part of the record. **In September only**, continued recording by each logger can be stopped, then data recorded on each logger must be transferred into an electronic database.

4) Shipping samples: Water chemistry samples (filtered water) should be packaged in coolers *with icepacks* with the lids securely shut with lots of duct tape (on either side of the shipping label, top-to-bottom, twice wrapped). Samples should be addressed to: Mingsheng Ma, University of Alberta Biogeochemical Analytical Service Laboratory (BASL), 11355 Saskatchewan Drive, Edmonton, Alberta, T6G 2E1, Tel: (780) 492-9014. Correspondence with Mingshen Ma via email (mingsheng.ma@ualberta.ca) is required to submit 2 items: 1) Chain of Custody form indicating analyses requested (see Figure SOP.2.2) and 2) an excel spreadsheet of the list of samples shipped (names, rough date of sample collection). Shipment involves a multi-step process, first McMurray Aviation (780-791-2182)

must be contacted to arrange cargo shipment of ~4 coolers filled with water samples to travel from Fort Chipewyan to Fort McMurray. Next, a shipping company (ground or air shipment) must be contacted to deliver the coolers to University of Alberta from McMurray Aviation Airport in Fort McMurray. We have had great success using Jazoo Express Courier (587-645-6364).

5) Inventory: Do a complete inventory of all the supplies and store in an excel spreadsheet for easy access to order for next sampling campaign or next season

6) Storage: Package and store all the supplies back into storage room in an organized fashion.

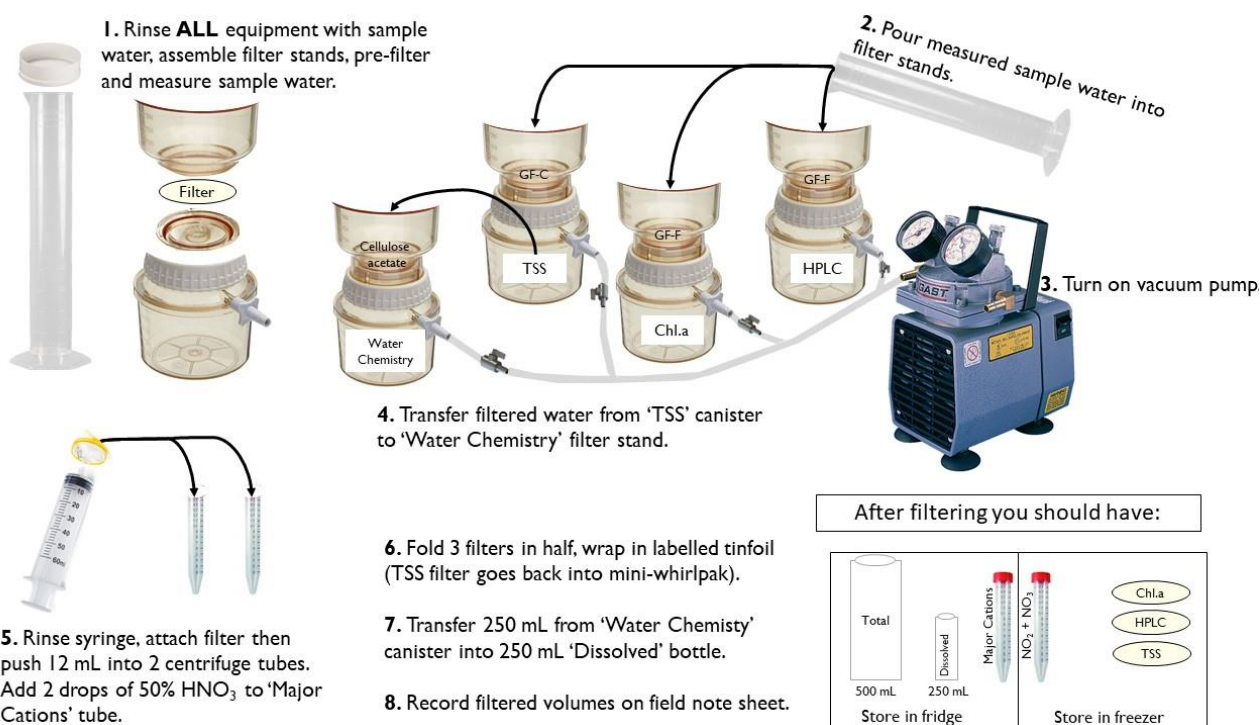


Figure SOP.2.1. Schematic depicting the set-up for filtering water. The numbers outline the necessary steps and order for filtering water to be analysed for select water chemical analyses at the University of Alberta Biogeochemical Analytical Service Laboratory (BASL). Schematic was compiled and created by Laura Neary.

Biogeochemical Analytical Service Laboratory
 2-255, CCIS Building, University of Alberta
 Edmonton, Alberta, T6G 2E1
 Phone: (780) 492-5497 Fax: (780) 248-5861


Chain of Custody Form

Project Name: _____ **Lab ID:** _____

Client Name: _____ Contact phone: _____ Contact e-mail: _____ Supervisor: _____ Billing Address or Speed Code: _____ _____ Comments: <u>Measure Si on dissolved fraction</u> Sample Matrix: <u>Surface water</u>	Submitted By: _____ Initial: _____ Date: _____ Comments: _____ _____ (For lab use only) Received By: _____ Date: _____ Comments: _____ _____
---	--

Analysis Requested:

Request	Parameter	ml Req.	Status
☐	PP (Particulate Phosphorus)	100	
☒	TP (Total Phosphorus)	100	
☒	TDP (Total Dissolved Phosphorus)	100	
☐	SRP (Soluble Reactive Phosphorus)	30	
☒	[NO ₂ ⁻ + NO ₃ ⁻] ([Nitrite+Nitrate])	15	
☐	NO ₂ ⁻ only (Nitrite)	15	
☐	NH ₄ ⁺ (Ammonium)	15	
☒	TN (Total Nitrogen) ☐ TN or TKN (RUSH: 4 weeks TAT)	20	
☒	TDN (Total Dissolved Nitrogen) ☐ TDN (RUSH: 4 weeks TAT)	20	
☒	TKN (Total Kjeldahl Nitrogen)	15	
	Done by calculation, both NO ₂ , NO ₃ and TN must be requested.		
☒	Na ⁺ (Sodium)	70	
☒	Ca ²⁺ (Calcium)		
☐	Fe (Iron)		
☒	Si (Silica)		
☐	K ⁺ (Potassium)		
☐	Mg ²⁺ (Magnesium)		
☐	Al (Aluminum)		
☐	Trace Metals	15	
☒	SO ₄ ²⁻ (Sulfate)	10	
☒	Cl ⁻ (Chloride)	20	
☒	DOC (Dissolved Organic Carbon)		
☒	DIC (Dissolved Inorganic Carbon)		
☐	CHN (Carbon Hydrogen Nitrogen)	Variable Volume	
☐	Chlorophyll a: Fluorimetric	Variable Volume	
☒	pH	60	
☐	Alkalinity ☒ Gran Alkalinity		
☐	Conductivity	40	
☐	Color (440 nm)	50	
☐	Absorbance (350 nm)		
☒	Turbidity	100	
☐	TDS (Total Dissolved Solids)	200	
☐	NFR/TSS (Non Filterable Residue/Total Suspended Solid)	200	
	Stable Isotope Ratio Analysis		
☐	¹⁸ O/ ¹⁶ O (Oxygen) ☐ D/H (Hydrogen)		
☐	¹³ C/ ¹² C (Carbon) ☐ ¹⁵ N/ ¹⁴ N (Nitrogen)		
☐	Archive raw sample	70	
☐	Other: _____		

Biogeochemical Analytical Service Laboratory, U of A

Figure SOP.2.2. Sample Chain of Custody (COC) Form to fill out and email (along with spreadsheet of samples) to the Biogeochemical Analytical Service Laboratory. X indicates selected analyses previously performed on lake and river samples from the Peace-Athabasca Delta, Alberta collected between the years of 2015 – 2019.

A3.1.6.3 Standard Operations for Component #3: Field & laboratory procedures for monitoring concentrations of metals of concern in lake surface sediments

This S.O.P. gives step-by-step instructions for collection of lake surface sediment in the Peace-Athabasca Delta, Alberta. Surface sediment samples should be kept refrigerated and shipped to University of Waterloo, where they will be prepared by a student (freeze-dried, weighed), under supervision of Dr. Roland Hall, and sent to a commercial laboratory (ALS, Waterloo) for metals concentration analysis.

This S.O.P. describes:

SOP.3.1 The field and laboratory equipment required.

SOP.3.2 The timing and sequence of data collection in the field.

SOP.3.3 Detailed methods on pre-field preparation, safety, field protocols and lab protocols.

SOP.3.4 Instructions for collecting lake surface sediment using mini-Glew gravity corer

SOP.3.5 Post fieldwork tasks

A list of routine monitoring site locations, including site names and GPS coordinates are reported in Table SOP.1.2. An additional Excel file was sent with this Technical Report as a template for recording field observations and values obtained during sample collection and processing.

SOP.3.1. Required equipment and forms

SOP.3.1.1. Surface sediment collection field equipment

- ☐ Mini-Glew corer
- ☐ several mini-Glew core tubes
- ☐ Sufficient lead sheet to provide the mass for corer to penetrate the sediment, duct-taped to 1-2 core tubes
- ☐ Elastic bands [for plunger of the mini-Glew corer]
- ☐ small bungs (make-shift using a 30 mL isotope bottle wrapped in enough duct tape to snugly fit inside the core tubes)
- ☐ Black Sharpie (permanent) markers
- ☐ Large Whirl-Pak bags (6 x 9 inches; 24 oz) [at least one per lake]
- ☐ Clipboard with field sampling notes
- ☐ Pens
- ☐ GPS loaded with lake and river locations
- ☐ Depth recorder

- ☐ Booklet with map of lakes and photos for identification
- ☐ Camera
- ☐ Extra batteries (AA)

SOP.3.1.2. Forms

The hydro-limnological monitoring field notes template (Figure SOP.1.1) can be used for recording field observations and in-situ measurements using the YSI multi meter.

SOP.3.2 Timing and Sequence of Events

SOP.3.2.1 Seasonal schedule of sampling periods

This sampling takes place in July, after spring flooding in the delta has been observed or confirmed using water isotope results. Typically, the collection of lake surface sediment coincides with hydrological and limnological monitoring conducted in July. The purpose of sampling right after a flood event is to collect lake sediment which originated as river sediment. The goal is to carefully obtain undisturbed cores of surficial sediment and carefully skim off the grey ‘flood layer’ at the upper surface. If collection of surface sediment cannot be conducted in July following a flood, contact research partners (U. Waterloo) to discuss options.

SOP.3.2.2 Length of sampling

For each lake, sampling will take ~15 minutes. Sampling can be performed over 2-3 days depending on how many lakes flooded (weather permitting). River sites cannot be cored due to effect of the current. An example sampling route can be found in Figure SOP.1.2.

SOP.3.2.3 Field preparation for lake surface sediment collection (July)

We recommend collecting surface sediment from the full suite of 60 lakes in years of widespread ice-jam flooding. Results from water isotope analysis are useful in determining which lakes flooded, so that only the uppermost, typically light grey coloured flood layer can be obtained from flooded lakes.

- 1) Determine which lakes flooded from water isotope values, May field observations and conductivity ($\mu\text{S}/\text{cm}$) measurements.
- 2) Gather all coring materials. Make sure there are extra mini-Glew tubes and ‘mini-bungs’.

- 4) Check [YSI Multi Meter](#) calibration and ensure battery is charged. Click [here](#) for a quick calibration guide.
- 5) Ensure GPS has correct coordinates and there are extra batteries.
- 6) Prepare field notes clipboard with monitoring field notes in the order listed in the sampling route pertaining to the 'water loops'. Indicate which lakes you will collect surface sediment at on the field notes.

SOP.3.2.4. Day-before tasks

Gather all required supplies for the first helicopter day and place them in one spot, ready to be checked and grabbed the morning prior to sampling:

Checklist 3-1. Packing lists for lake surface sediment helicopter sampling

- ☐ Mini-Glew corer, core tubes with lead sheet attached, rubber bands
- ☐ 6-10 Small bungs (made from isotope bottles wrapped in duct tape to fit snugly inside core tubes)
- ☐ Many large Whirl-Pak bags (at least one per lake)
- ☐ GPS + extra batteries
- ☐ Camera
- ☐ Clipboard + field sampling notes
- ☐ Sharpies
- ☐ Emergency supply kit

SOP.3.2.6 Upon return to Fort Chipewyan – same day as sampling

Labelled Whirl-Pak bags with lake **surface sediment should be stored in the** fridge directly after returning. (There is no corresponding processing or laboratory work).

SOP.3.3. Lake surface sediment collection: field protocol

SOP.3.3.1. Lake surface sediment collection protocol

- a. Identify lake using GPS coordinates.
- b. Take a photo of the site name on the field sampling notes.
- c. Upon descent to the lake site, take photos of the lake from the helicopter.
- d. Once landed, take photos of lake (three shots from left to right, covering the whole lake area).

- e. Collect surface sediment from lake (combine sediment from 3-5 deployments of the corer) using detailed instructions in section SOP.3.4.
- f. Turn on YSI, submerge probe into water making sure that it is not touching sediment. Wait until numbers stabilize before recording values on the field notes sheet. Record values of YSI measurements in field notes. (Temperature, dissolved oxygen, conductivity, total dissolved solids, pH, turbidity, total suspended solids).
- g. Fill out remaining observational notes (Figure SOP.1.1 and section 3.3.2).

SOP.3.3.2 Filling in field notes: recording observations and YSI-multimeter measurements

Fill in the field sampling sheets for each lake/river site (Figure SOP.1.1).

Lake/River ID: site name (i.e. PAD 15)

Date: record the month, day, and year (DD-MM-YYY).

Time: record time of arrival at lake in 24-hour clock (HH:MM).

Sampling Crew: identify who is collecting samples.

Weather: note things like precipitation, wind strength and direction, cloud cover.

YSI Multi Meter:

Temperature: record in degrees Celsius (°C).

Dissolved Oxygen: record DO % and mg/L.

Conductivity: record in micro Siemens per centimeter ($\mu\text{S}/\text{cm}$).

Total Dissolved Solids (TDS): record in mg/L.

pH: record unit-less value.

Turbidity: recorded in FNU.

Total Suspended Solids (TSS): record in mg/L.

Water depth: record in meters (m); approximate depth at sampling point and the lake.

Total Suspended Solids (TSS): record in mg/L.

Lake Colour/Turbidity: record colour and visual turbidity of the water column.

Lake Description: record any useful information about the hydrology of the lake including, connections, channels, evidence of drawdown.

Catchment Description: describe features such as catchment vegetation, slope, unusual features.

Macrophytes: circle the most accurate description of type and cover from the options listed and provide any additional notes below.

SOP.3.4 Instructions for collecting lake surface sediment using mini-Glew gravity corer

[See Figure SOP.3.1 for schematic]

1) Prepare corer

Pack extra supplies for another core tube in case one gets lost during sampling. Strap a lead weight to outside of the mini-Glew core tube. Ensure top of core tube (flat end, not tapered) and inside of corer housing have been lubricated lightly with Vaseline (once per coring day). Make sure rubber band is in good shape and is properly positioned (bring extra rubber bands in case one breaks). Attach core tube to corer and tighten band clamp on corer tube housing using nut driver. Core tube should not move within housing. Raise plunger of corer to “loaded” position.

2) Test and lower corer

Test to see if corer is properly sealed by lowering into water, triggering the corer, and lifting to see if the tube is filled and holds water. (If not, re-check seals and Vaseline, ensure band clamp is tightened). Release water and reset corer to “loaded” position. Slowly lower corer into water. Extend arm and lower slowly until corer penetrates into sediment (max. speed = ~ 0.5m/sec; increase speed of descent if sediment is very stiff and prevents sufficient penetration at slower deployment speed). If water depth is sufficiently shallow, push core manually into the sediment. Send messenger down line to trigger the corer to close. After plunger on the corer is triggered, raise slowly to the surface.

3) Retrieve corer

While core tube is submerged, insert a bung into bottom of tube to form a lower seal, then lift the corer out of the water and place on the helicopter pontoon. Check for clarity of water directly above sediment.

4) Separate core tube from corer

Firmly holding onto the corer, use the nut driver to loosen the band clamp on the corer tube housing. Rotate corer slightly to ensure band clamp is loosened sufficiently. Holding onto the core tube with one hand, use the other hand to slowly remove the corer from the top of the tube. Some back and forth rotation of the corer may be necessary. Again, be careful not

to disturb the sediment-water interface. Receive push rod from support person located at back seat of helicopter.

5) Insert the extruder rod and remove excess water

Grab the extruder push-rod and hold it vertically with the bottom end firmly flat on the wooden cover of the helicopter pontoon. Slide core tube down over extruder push-rod to allow water to flow smoothly out of the core tube top. Continue until core surface is within ~1 cm of the core tube top.

6) Collect the sample

Support person opens a Whirl-Pak bag and holds it firmly along the outer wall of the core tube, just below the top of core tube. Tip the core tube slightly towards you so the fluid sediment flows into the Whirl-Pak bag. Push down just enough to collect the surficial flood layer, which should be visible as a thin grey layer. Ensure label on bag is correct. Close top of bag, push out excess air, spin a few times around top twist tie. Tie off bag.

***Repeat above steps to obtain 3 (or more) surface sediment samples from the same lake site (store all in same Whirl-Pak bag).**

7) Rinse equipment

Place collected sample in a cooler. Discard rest of sediment into water, rinse out core tube and extruding rod. Prepare for next sampling site.

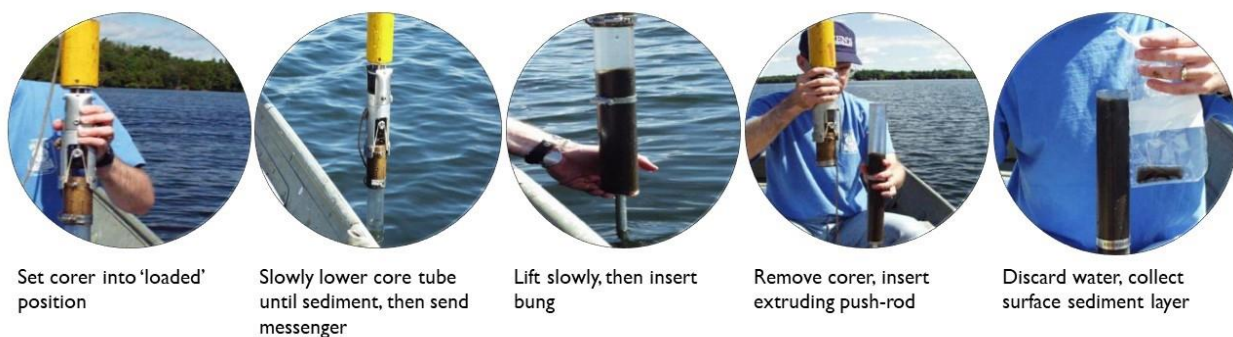


Figure SOP.3.1. Schematic depicting collection of lake surface sediment using a mini-Glew gravity corer.

SOP.3.5. Post-collection processing and storage

SOP.3.5.1. Post-collection tasks and procedures

1) Data entry:

-Transfer data from field note sheets to the excel file template provided by research partner (UW). Although all the data will be transcribed to a datasheet or a computer spreadsheet equivalent, original field notes should be preserved for at least one year, and preferably indefinitely as part of the record.

2) Ship samples: Surface sediment in Whirl-Paks should be packaged **in a cooler with sufficient ice packs** and shipped to research partner at Professor Roland Hall, Department of Biology, University of Waterloo, 200 University Ave. W, Waterloo, ON, N2L 3G1. (Lots of duct tape can be used to secure the cooler lid from opening during shipment).

-Send an email to the research partners (rihall@uwaterloo.ca) regarding the shipment and attach the field notes file.

3) Inventory: Do a complete inventory of all the supplies and store in an excel spreadsheet for easy access to order for next season

4) Storage: Package and store all the supplies back into storage room in an organized fashion.

Pond hydrology and water chemistry of the Whooping Crane Nesting Region in 2021 (Technical Report)

Prepared for:

Parks Canada Agency (Wood Buffalo National Park)

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A3.2.1 Executive Summary

The whooping crane (*Grus americana*) is a critically endangered species whose small population is threatened by potential climate-driven changes to wetland habitat. The only natural breeding area in North America lies within a water-rich sector of northern Wood Buffalo National Park where ponds and wetlands are abundant. Although it is known that drought and associated decline of water levels negatively affects their production and increases risk to predation, and water-level rise can decrease reproductive success by flooding of nests along shorelines, insufficient knowledge of the processes influencing the water quantity and quality of this wetland-rich landscape hampers ability to anticipate how climate change will affect whooping crane.

The research communicated in this report aims to begin to address this knowledge gap by using water isotope tracers, depth loggers and water chemistry measurements to characterize the current hydrological conditions and water quality of ponds in the Whooping Crane Nesting Region (WCNR). This report includes results of a pilot study when measurements were obtained during the open-water season of 2021 to compare the hydrological conditions and water quality of ponds actively used in the past for nesting by whooping crane *versus* ponds not-used for nesting. Complementary paleolimnological studies will be launched in 2022 to increase knowledge of change over time in hydrological conditions, water quality and habitat availability. The overarching goal is to understand the potential vulnerability and responses of ponds to climate change, which aligns with a recommendation made by 2016 UNESCO Reactive Monitoring Mission to “*Continue to closely monitor the entire used and potential nesting area of the Whooping Crane within the Greater WBNP Ecosystem so as to be able to respond to possibly changing management requirements*” [WHC/IUCN 2017, p.5].

A3.2.2 Key Results

- Isotope-based assessment of lake water balances revealed that water input exceeds water loss by evaporation ($E/I < 1.0$) at all ponds ($n = 50$) due to inflow from snowmelt, rainfall and possibly groundwater. A systematic shift in estimates of input

water isotope compositions identify snowmelt as the predominant input in June compared to rainfall and/or groundwater in September.

- Temporal assessment of pond-level fluctuations using hourly depth measurements revealed remarkably stable pond levels that respond variably to rainfall events.
- Water chemistry of ponds differed between June and September with increases in $[\text{Ca}^{2+}]$, $[\text{SO}_4^{2-}]$, $[\text{Mg}^{2+}]$, $[\text{TN}]$, $[\text{TDN}]$, $[\text{DOC}]$, $[\text{DSi}]$ and conductivity and decreases in $[\text{TP}]$, $[\text{TDP}]$ and turbidity.
- Proportionally high concentrations of Ca^{2+} and SO_4^{2-} in the pond water identify groundwater as an important input to ponds in the Whooping Crane Nesting Region.
- Results revealed no significant difference in hydrological conditions and water quality of ponds recently used (but not in 2021) versus not used by whooping cranes since at least 1966.

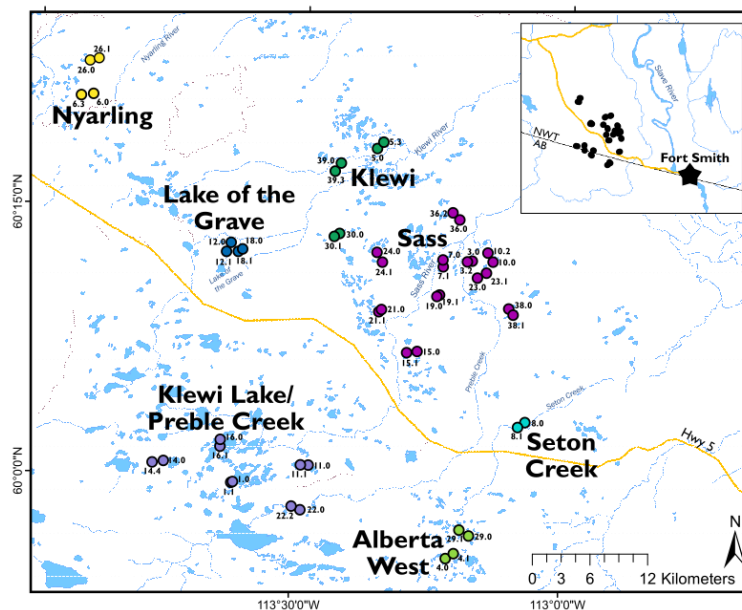
A3.2.3 Objectives

The objectives are to **1)** identify the influence of hydrological processes (i.e., water input from snowmelt, rainfall and groundwater, and water losses by evaporation) on pond water balances and pond-level variation, **2)** characterize the seasonal variability in pond water quality and **3)** compare hydrological conditions and water quality between two groups of ponds (recently used for nesting by whooping crane versus not-used since at least 1966). In 2021, the study design included 25 ponds that whooping cranes have actively used for nesting since 1966 (but not in 2021), paired with 25 nearby ponds that have not been used for nesting since monitoring began in 1966. Note that in 2021, we were unable to sample ponds that are currently being used by whooping crane because ECCC staff were unable to travel to the field location due to pandemic-related travel restrictions. This pilot sampling strategy was implemented to explore the possibility that some pond sites may have more favourable hydrological conditions and water quality for whooping crane.

A3.2.4 Methods

A3.2.4.1 Pond site selection, sample collection and depth logger deployment

Ponds were selected in collaboration with wildlife biologists (John Conkin and Mark Bidwell – Canadian Wildlife Service, Environment and Climate Change Canada) who have been tracking the whooping crane population in WBNP, including locations of nesting sites. Fifty ponds were selected across the Whooping Crane Nesting Region (WCNR) using a paired sampling design (Figure A3.2.1). Half of the ponds (25) are known to have been used for nesting by whooping crane in the recent past (but not in 2021) and were paired with a nearby (<2 km) pond that has not been used for nesting since monitoring began in 1966. The WCNR spans an elevational range of 160-320 masl (Table A3.2.1, A3.2.2; Figure A3.2.2a), with the highest elevations occurring in the southwest (towards the Caribou Mountains) and the lowest elevations in the northeast (towards the Slave River). Consequently, seven subregions within the WCNR can be sorted from highest to lowest elevation: Preble Creek, Alberta West, Lake of the Grave, Seton Creek, Nyarling, Sass and Klewi (Figure A3.2.2a,b).



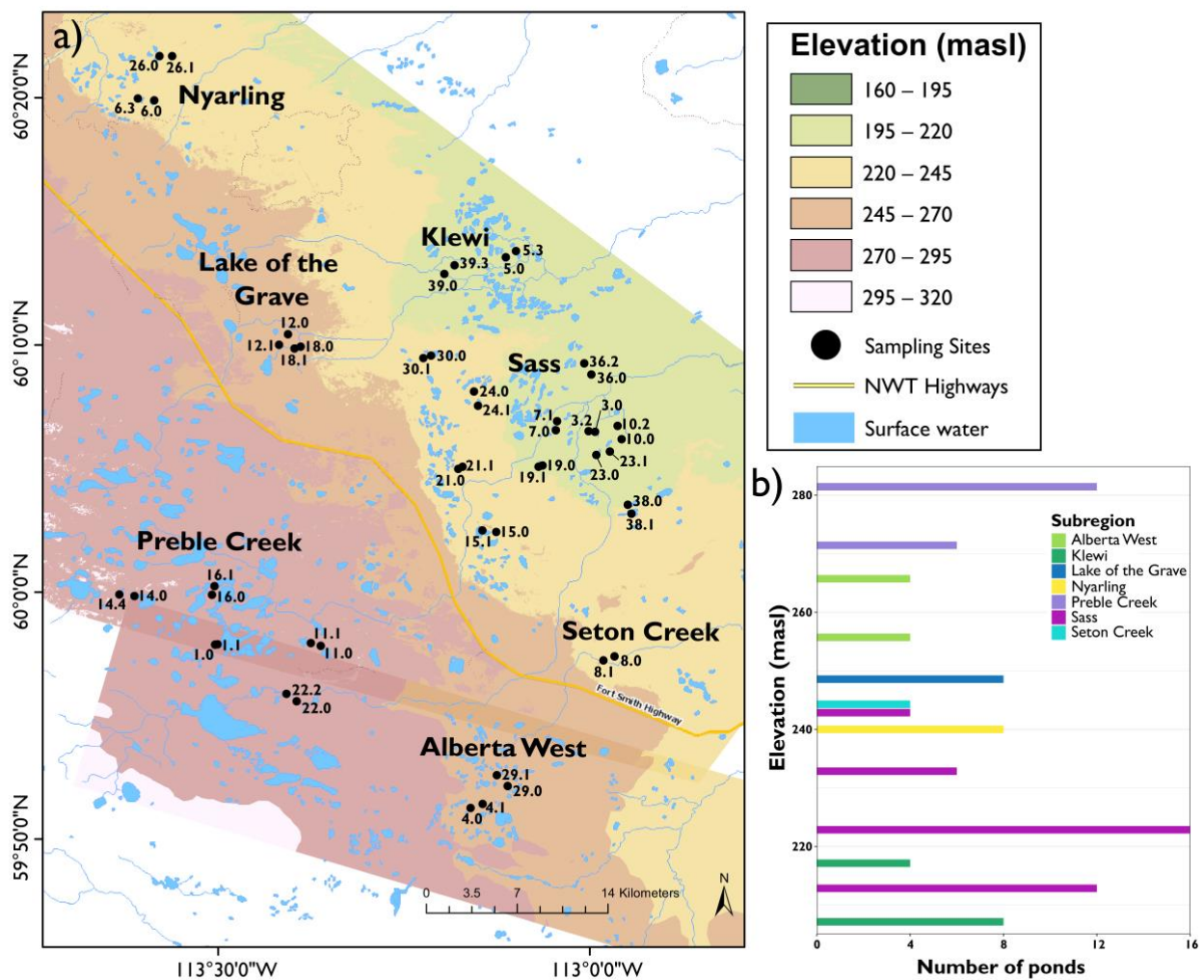


Figure A3.2.2. a) Elevation map of the Whooping Crane Nesting Region (WCNR) in Northwest Territories and Alberta, Canada. Pond sites are located with black points and are labelled according to their category, with a zero after the decimal place identifying the ‘used’ ponds compared to the ‘not used’ ponds which have a non-zero number after the decimal place. A number of external sources were used including: 1) a shapefile that includes water bodies and major highways in Northwest Territories (retrieved from <https://www.geomatics.gov.nt.ca/en>), 2) a high-resolution digital elevation model (HRDEM) of the Northwest Territories (retrieved from <https://open.canada.ca/data/en/dataset/957782bf-847c-4644-a757-e383c0057995>) and 3) a 1:20000 digital elevation model of Alberta (retrieved from <https://www.altalis.com/map?id=125> (1:20000 DEM)). **b)** a histogram showing the number of pond sites that are situated at various elevations, coloured according to their respective subregion.

Table A3.2.1. Decimal degrees, *in situ* measurements of pond depth, temperature, conductivity, pH and field notes for 50 ponds within the Whooping Crane Nesting Region from June 3-5, 2021. Note: dissolved oxygen sensor on the handheld YSI was not measuring properly in the field during this trip. Elevation for each site was extracted from digital elevation models available online (see direct link in Figure A3.2.2).

Pond ID	Decimal Degrees		Elevation (masl)	Depth (m)	Temp (°C)	Conductivity (µS/cm)	pH	Other additional notes
	Latitude	Longitude						
N1.0	59.997	-113.615	279.67	0.20	18.7	457.1	7.52	suspended particles in water?
N1.1	59.998	-113.612	279.67	0.15	18.4	1435.9	8.09	small rocks on bottom
N3.0	60.211	-113.178	214.62	0.60	21.4	953.9	7.99	High water (drowned shrubs)
N3.2	60.210	-113.187	214.45	0.60	21.3	952.9	7.8	
N4.0	59.934	-113.211	267.08	0.30	20.2	854.6	7.25	Partially submerged shrubs
N4.1	59.939	-113.196	265.95	0.50	21.1	919.7	7.35	small rocks on bottom
N5.0	60.312	-113.364	210.30	0.60	19.2	1064.0	8.00	Visible flow connection
N5.3	60.318	-113.352	209.98	0.60	18.8	1070.1	7.67	Less conifers surrounding (mostly shrubs)
N6.0	60.353	-113.902	239.51	0.40	19.6	1193.6	8.09	Submerged shrubs (wetland-like pond)
N6.3	60.351	-113.925	240.66	0.50	20.3	959.3	7.92	
N7.0	60.205	-113.232	214.94	0.50	21.9	1456.5	7.72	
N7.1	60.211	-113.234	214.32	0.40	21.8	1242.0	7.69	Submerged shrubs
N8.0	60.062	-113.071	236.48	0.50	18.7	875.6	7.38	
N8.1	60.057	-113.084	238.68	0.60	18.1	867.8	7.39	High water, cattails
N10.0	60.210	-113.139	215.49	0.40	21.9	750.6	7.67	
N10.2	60.219	-113.150	214.48	0.60	20.5	683.2	7.59	
N11.0	60.016	-113.472	272.69	0.60	22.1	732.2	8.24	Pond bottom is yellowish
N11.1	60.016	-113.486	273.30	0.50	22.4	912.8	8.42	
N12.0	60.220	-113.632	248.93	0.50	19.6	1127.0	7.45	Smaller pond, many conifers
N12.1	60.211	-113.640	249.00	0.60	19.6	983.4	7.94	
N14.0	60.015	-113.742	278.75	0.60	19.5	565.5	7.97	
N14.4	60.013	-113.763	278.94	0.40	19.3	664.9	7.90	More yellow/brown than surrounding ponds
N15.0	60.125	-113.276	237.67	0.50	21.9	1006.2	7.72	
N15.1	60.124	-113.296	238.42	0.60	21.1	1459.0	8.30	Very large pond, can't see bottom
N16.0	60.030	-113.638	275.13	0.25	21.1	760.1	8.10	
N16.1	60.037	-113.638	274.95	0.50	21.9	570.0	8.70	Caribou sighted nearby
N18.0	60.212	-113.618	247.21	0.40	21.4	1019.6	8.09	Visible flow connection with other wetlands
N18.1	60.214	-113.610	246.94	0.40	20.9	1350.2	8.30	
N19.0	60.178	-113.237	217.12	0.30	22.0	1868.5	8.54	Bubbles on top
N19.1	60.177	-113.242	217.95	0.20	21.9	2078.6	8.25	Peregrine falcon on pond (cliffs nearby)
N21.0	60.161	-113.351	231.87	0.40	21.2	1791.7	7.47	
N21.1	60.163	-113.346	231.70		22.1	1598.7	7.75	
N22.0	59.974	-113.484	276.69	0.70	20.4	639.2	7.24	Drowned tree stands
N22.2	59.977	-113.501	277.51	0.60	21.1	646.2	7.31	
N23.0	60.195	-113.168	215.92	0.70	21.4	875.1	7.96	
N23.1	60.200	-113.151	215.54	0.60	21.2	899.6	8.05	
N24.0	60.216	-113.357	224.30	0.30	21.9	1009.6	7.93	
N24.1	60.207	-113.347	225.09		21.4	1184.2	8.08	
N26.0	60.384	-113.911	236.69	0.60	20.1	1401.6	8.18	
N26.1	60.386	-113.894	236.18	0.60	19.4	1207.2	8.30	Large pond, small waves (choppy)
N29.0	59.955	-113.169	263.53	0.50	20.7	964.2	7.75	
N29.1	59.961	-113.187	262.62	0.40	20.5	839.2	8.36	Medium turbidity – hard to see bottom
N30.0	60.232	-113.429	224.38	0.20	16.8	1544.1	7.52	Very shallow, difficult to get YSI reading
N30.1	60.229	-113.439	224.60	0.30	18.4	1703.4	7.84	
N36.0	60.249	-113.204	210.79	0.30	18.0	1313.6	7.23	Mostly surrounded by cattails and grasses
N36.2	60.255	-113.218	212.51	0.60	19.3	1307.5	7.69	
N38.0	60.167	-113.107	219.74	0.50	19.9	1059.6	8.38	
N38.1	60.162	-113.099	220.08	0.50	20.4	1551.4	8.42	
N39.0	60.290	-113.441	212.05	0.60	19.6	1117.2	7.57	Drowned trees from previous years
N39.3	60.298	-113.430	211.42	0.60	20.0	1107.5	7.58	

Table A3.2.2. *In situ* measurements of pond temperature, conductivity, pH and field notes for 50 ponds within WCNR from September 18-19, 2021. Note that depth was not measured.

Pond ID	Temp (°C)	D.O (%)	Conductivity (µS/cm)	pH	Other additional notes
N1.0	8.5	72	813	7.43	
N1.1	7.6	102	1971	7.90	
N3.0	8.9	76	939	7.63	
N3.2	8.7	75	940	7.64	
N4.0	8.2	74	1416	7.48	
N4.1	7.7	67	566	7.29	
N5.0	8.7	101	1549	7.81	
N5.3	8.9	103	1449	7.77	
N6.0	9.5	100	1614	7.79	
N6.3	9.4	99	1343	7.81	
N7.0	9.1	100	2081	7.87	
N7.1	8.9	106	1547	8.00	Some aspen surrounding, drowned shrubs
N8.0	9.1	91	1337	7.62	
N8.1	9.0	67	1360	7.43	
N10.0	8.4	51	939	7.39	
N10.2	8.5	83	899	7.56	
N11.0	8.4	100	1048	8.02	
N11.1	8.5	100	1347	8.31	
N12.0	8.7	52	1667	7.33	
N12.1	8.7	92	1507	7.78	
N14.0	8.3	106	787	8.10	
N14.4	8.0	87	703	7.76	
N15.0	9.5	78	1518	7.39	
N15.1	9.5	106	1736	8.09	
N16.0	8.0	94	1391	7.93	
N16.1	7.8	102	973	8.03	
N18.0	9.0	105	1562	7.87	
N18.1	8.6	108	1692	7.99	
N19.0	9.0	104	2686	8.07	
N19.1	8.9	91	2152	7.85	Very shallow, can see crane tracks Lots of little islands
N21.0	9.0	74	1980	7.42	
N21.1	9.0	104	1895	7.83	
N22.0	8.0	77	775	7.27	
N22.2	8.2	70	773	7.39	
N23.0	9.5	87	977	7.78	
N23.1	9.6	97	921	7.79	
N24.0	8.6	86	1363	7.67	
N24.1	8.7	77	1644	7.47	
N26.0	9.3	102	1760	7.93	
N26.1	9.3	106	1865	7.97	
N29.0	8.1	83	1395	7.58	Kicked up sediment
N29.1	8.1	97	1365	8.02	
N30.0	8.1	82	2175	7.57	Whiteish pond colour
N30.1	8.3	85	2038	7.64	
N36.0	8.2	47	1968	7.19	
N36.2	8.9	93	1785	7.68	
N38.0	9.6	103	1870	8.11	
N38.1	9.8	108	2472	8.25	
N39.0	8.4	90	1729	7.60	Drowned trees nearby
N39.3	8.7	83	1721	7.58	

Sampling in 2021 was conducted using a helicopter with fixed floats on June 3-5 and again on September 18-19 to avoid migration periods of the whooping crane. Local Parks Canada Agency (PCA) employees, based in Fort Smith (Lori Parker, Keith Hartery, Teresa Little), led sample collection while university researchers were unable to travel north due to Covid-related travel restrictions. At each of the 50 ponds, two water samples were collected at 5-

10 cm below the surface. One 30 mL water sample was collected for measurement of water isotope composition and one 1-litre sample was collected for measurement of water chemistry variables. Depth loggers that recorded hourly water-level variation were deployed at each pond during the first field visit and retrieved during the second field visit. One logger was deployed at the WBNP compound in Fort Smith to measure variation in air temperature and pressure, which is needed to convert pond measurements to water depth.

A3.2.4.2 Isotope hydrology

A3.2.4.2.1 Laboratory methods

Water isotope samples were analyzed at the University of Waterloo – Environmental Isotope Laboratory (UW-EIL) using a Los Gatos Research (LGR) Liquid Water Isotope Analyser (LWIA) model T-LWIA-45-EP instrument with precisions (2σ) of $\delta^2\text{H} = \pm 0.8\text{‰}$ and $\delta^{18}\text{O} = \pm 0.2\text{‰}$. A suite of water standards (Vienna Standard Mean Ocean Water, Vienna Standard Light Antarctic Precipitation) from the International Atomic Energy Agency (IAEA) were used as reference materials and duplicate samples were run at a minimum of every tenth sample. See the [EIL website](#) for more information regarding their analytical QA/QC protocols. Isotope compositions are expressed as δ -values, representing deviations in per mil (‰) from Vienna Standard Mean Ocean Water (VSMOW) such that $\delta_{\text{sample}} = [(R_{\text{sample}}/R_{\text{VSMOW}}) - 1] \times 1000$, where R is the $^{18}\text{O}/^{16}\text{O}$ or $^2\text{H}/^1\text{H}$ ratio in the sample and VSMOW. Results of $\delta^{18}\text{O}$ and $\delta^2\text{H}$ analyses are normalized to -55.5‰ and -428‰ , respectively, for Standard Light Antarctic Precipitation (Coplen, 1996).

A3.2.4.2.2 WCSR isotope ‘framework’ for 2021

Development of a region-specific isotope framework allows for independent interpretation of water isotope data within a specific region of study. The WCSR isotope ‘framework’ is best represented by two linear relationships driven by the mass-dependent partitioning of stable isotopes contained within the water molecule as water passes through the hydrological cycle (Figure A3.2.3; Edwards et al., 2004):

1) The Local Meteoric Water Line (LMWL) represents a range of isotope values for precipitation in a given area (i.e., Fort Smith, NT; $\delta^2\text{H}$ 6.7 $\delta^{18}\text{O}$ – 19.2). The linear relationship for Fort Smith is derived from hydrogen and oxygen isotope

measurements in precipitation collected in Fort Smith from 1960-1969 as part of a worldwide initiative to monitor isotopes in precipitation (Global Network for Isotopes in Precipitation; IAEA, 2022; archive data available at <https://www.iaea.org/services/networks/gnip>).

2) The Local Evaporation Line (LEL) represents the expected trajectory of a water body fed by mean annual precipitation (δ_P) as it undergoes evaporation (Gibson and Edwards, 2002). It is anchored by two isotope compositions: δ_P and δ^* . δ_P represents the isotope composition of amount-weighted mean annual precipitation in Fort Smith, NT. This value was obtained from a global open-access database that uses data collected from 1492 stations across the world to model monthly changes in oxygen and hydrogen isotopes in precipitation (Bowen and Wilkinson, 2002; Bowen and Revenaugh, 2003; Bowen et al., 2005; IAEA/WMO, 2015; OIPC website: https://wateriso.utah.edu/waterisotopes/pages/data_access/oipc.html). δ^* represents the maximum potential evaporation of a lake as it approaches desiccation (also known as the limiting isotope composition). This value was derived using the equation provided by Gonfiantini (1986):

$$\delta^* = \frac{h \cdot \delta_{AS} + \varepsilon_k + \varepsilon^*/\alpha^*}{h - \varepsilon_k - \varepsilon^*/\alpha^*} \quad \text{EQ. A5.2.1}$$

where h is the relative humidity (average ice-free season flux weighted; available at www.climate.weather.gc.ca), δ_{AS} is the isotope composition of atmospheric moisture for the ice-free season (Gibson et al., 1999, 2002), ε_k is the kinetic separation factor between liquid and vapour phases (Gonfiantini, 1986), ε^* is the equilibrium separation factor between liquid and vapour phases (Gonfiantini, 1986) and α^* is the equilibrium fractionation factor between liquid and vapor phases (Horita and Wesolowski, 1994).

One other point that falls on the LEL is δ_{SSL} , the isotope composition of a terminal lake in “steady state” (i.e., evaporation = inflow). This value was determined using the classic linear resistance model of Craig and Gordon (1965) and the equation provided by Gonfiantini (1986):

$$\delta_{SSL} = \alpha^* \delta_P (1 - h + \varepsilon_k) + \alpha^* h \delta_{AS} + \alpha^* \varepsilon_k + \varepsilon^* \quad \text{EQ. A5.2.2}$$

where δ_P represents the average annual isotope composition of precipitation in Fort Smith, NT. The other parameters are listed below EQ. A5.2.1.

The WCSR isotope framework and a coupled-isotope tracer approach (Yi et al., 2008) were then used to determine the isotope composition of input water, δ_I , for each pond for June and September 2021. This value was estimated from the intersection of each pond's evaporation line ($\delta_E - \delta_L$) with the Local Meteoric Water Line assuming input water is primarily meteoric water that has not undergone evaporation. δ_E was determined using the Craig and Gordon (1965) model and equation provided by Gonfiantini (1986):

$$\delta_E = \frac{(\delta_L - \varepsilon^*)}{\alpha^* - h\delta_{AS} - \varepsilon_k} / (1 - h + \varepsilon_k) \quad \text{EQ. A5.2.3}$$

where δ_L is the measured pond water isotope composition, ε^* is the equilibrium separation factor between liquid and vapour phases (Gonfiantini, 1986), α^* is the equilibrium fractionation factor between liquid and vapor phases (Horita and Wesolowski, 1994), h is the flux-weighted relative humidity, δ_{AS} is the isotope composition of atmospheric moisture for the ice-free season calculated assuming isotope equilibrium with d_{Ps} (Gibson et al., 1999, 2002) and ε_k is the kinetic separation factor between liquid and vapour phases (Gonfiantini, 1986).

Evaporation-to-inflow ratios were then determined using the equation provided by Gibson and Edwards (2002):

$$\frac{E}{I} = \frac{(\delta_I - \delta_L)}{(\delta_E - \delta_L)} \quad \text{EQ. A5.2.4}$$

where δ_I is the pond-specific isotope composition of input water (determined as the intersection of $\delta_E - \delta_L$ with the LMWL), δ_L is the measured pond water isotope composition and δ_E is the isotope composition of evaporated vapour from a pond (EQ. A5.2.3).

A5.2.4.2.3 Water balance metrics

Figure A3.2.3 illustrates how water moves throughout the hydrological cycle and how measurements of water isotope composition (reported as $\delta^{18}\text{O}$, $\delta^2\text{H}$) in surface waters (ponds, lakes) can be used to understand the dominant sources of input waters (i.e., rain, snow, groundwater) and influence of evaporation on pond water balances. The WCSR isotope framework and a coupled-isotope tracer approach (Yi et al., 2008) were used to determine the isotope composition of input water (δ_I) and evaporation-to-inflow ratios (E/I) for each pond. Values of δ_I can be used to understand the dominant sources of water to ponds, including snowmelt, rain or groundwater, and their relative proportions. E/I ratios are an informative water-balance metric that represents the ratio of evaporation to inflow for a specific water body. For example, a lake with an E/I value greater than 1.0 experiences net evaporation whereas a lake with E/I less than 1.0 experiences net inflow.

a) Isotopes in the hydrological cycle

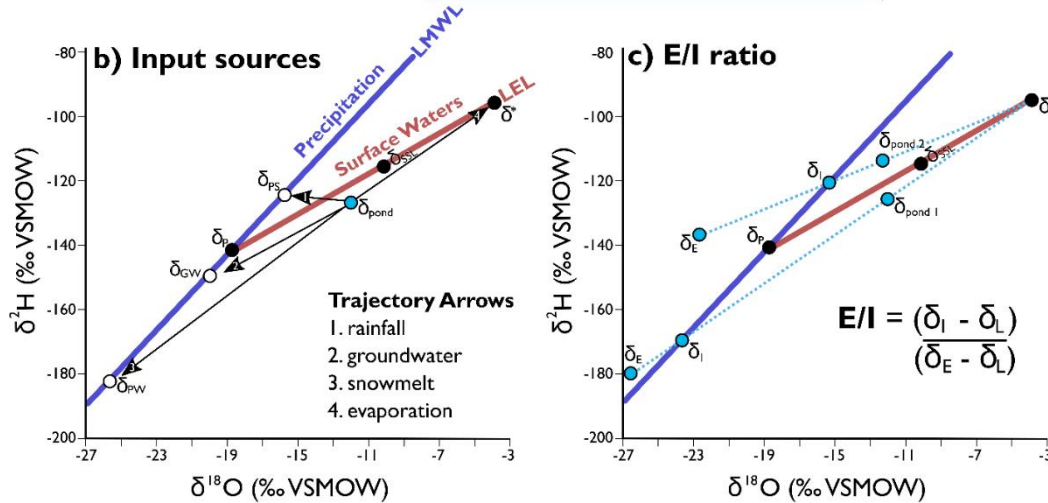
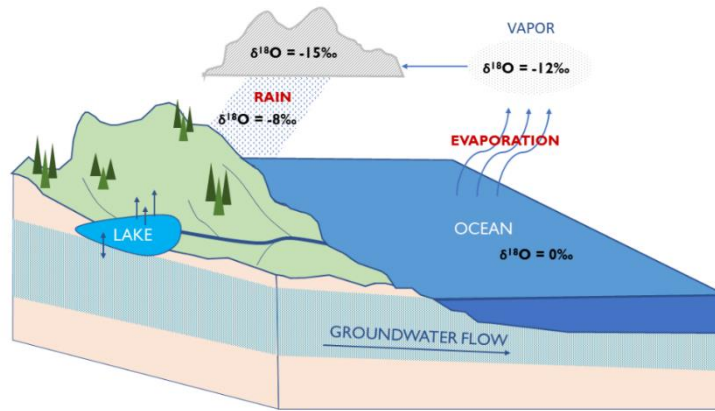


Figure A3.2.3. a) Schematic depicting the evolution of oxygen isotope compositions as water evaporates from the ocean, vapour moves inland, and rain forms (note that values listed are a potential scenario). Key processes that alter the ratio of heavy to light isotopes are written in red text. Black text indicates the oxygen isotope composition of the ocean, water vapor and rain in this scenario. **b)** Schematic $\delta^{18}\text{O}$ - $\delta^2\text{H}$ graph illustrating two linear relations: 1) the Local Meteoric Water Line (LMWL), which represents the region-specific distribution of the isotope composition of precipitation (rain, snow), and 2) the Local Evaporation Line (LEL), which represents the evaporative trajectory of surface water (lakes, ponds, rivers) fed by waters of mean annual isotope composition of precipitation (δ_p). Key points along the LEL include δ_{SSL} which represents the special case of a terminal (i.e., closed-drainage) basin in isotopic and hydrological steady-state fed by waters of mean annual isotope composition of precipitation, and δ^* which represents the theoretical limiting isotopic enrichment attainable by a desiccating water body under average ice-free season atmospheric conditions. Pond water isotope composition (δ_{pond}) will ‘move’ in $\delta^{18}\text{O}$ - $\delta^2\text{H}$ space along and about the LEL due to the influence of snowmelt, rainfall, groundwater and evaporation. Directional influence of these hydrological processes on a pond (δ_{pond}) are indicated as ‘trajectory arrows’ 1-4. The direction of the arrows indicates expected changes to pond water isotope values if each process (1: rainfall, 2: groundwater, 3: snowmelt, 4: evaporation) were acting independently. δ_{PW} and δ_{PS} represent average isotope composition of winter precipitation and summer precipitation, respectively. The isotope composition of groundwater (δ_{GW}) is presumed to be a mixture of rain and snow. **c)** Schematic depicting how evaporation-to-inflow ratios are calculated using δ_i (pond-specific composition of input water) following the methods of Yi et al. (2008), δ_E (pond-specific isotope composition of evaporated vapour) and δ^* (the theoretical limiting isotopic enrichment value).

Mann-Whitney U tests (nonparametric equivalent of a two-sample t-test) were used to determine if there were significant differences in δ_i and E/I for used vs. not-used ponds in June and September 2021.

A3.2.4.3 Depth logger data handling

Hourly logger data were converted from absolute pressure (psi) measurements to pond depth (m) using the Barometric Compensation Assistant in the HOBOWare Pro software program and air temperature and pressure data recorded on a logger deployed in the air (tied to a fence at the Parks Canada Compound in Fort Smith) at the same time intervals. Data were compiled using Microsoft Excel and a moving average (24-hour) was applied in RStudio to reduce minor high-frequency variability in the datasets. The amount of drawdown that occurred at each pond was determined as = minimum – initial x100, to express drawdown in cm. Mann-Whitney U tests were used to determine whether mean pond depth and drawdown differ significantly between the two pond categories (used vs. not used).

A3.2.4.4 Water quality

A3.2.4.4.1 Laboratory methods

Water samples were refrigerated prior to shipment to University of Alberta's Biogeochemical Analytical Services Laboratory (BASL) for the following analyses: pH, Gran alkalinity, turbidity, and concentrations of total nitrogen (TN), total phosphorus (TP), total dissolved nitrogen (TDN), total dissolved phosphorus (TDP), conductivity, dissolved silica (DSi), dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), bicarbonate (HCO_3^-), major anions (Cl^- , SO_4^{2-}) and major cations (Ca^{2+} , K^+ , Mg^{2+} , Na^{2+}). Concentrations of TN, TDN, TP, TDP and DSi were determined by Flow Injection Analysis (TN/TDN Method: 353.2; TP/TDP Method: 4500-P-G; Dissolved Silica Method: 370.1) using a Lachat QuickChem QC8500 FIA Automated Ion Analyzer. Concentrations of DIC and DOC were determined using a Shimadzu TOC-5000A Total Organic Carbon Analyzer (Method 415.1). Gran alkalinity, conductivity and pH in water samples were determined using the PC-titrate instrumentation (Alkalinity Titration Method 2320 B, pH Electrometric Method 4500-H+B, Gran alkalinity Method Series 09-A6.6). Turbidity was determined using a Hach 2100N Turbidimeter

(Method 180.1). Ionic concentrations were determined using a Thermo ICAP-6300 Inductively Coupled Argon Plasma – Optical Emission Spectrometer (Method 200.7).

A5.2.4.4.2 Statistical analyses

The variation (i.e., range, mean, median) in water chemistry parameters was explored using simple boxplots to visualize whether differences exist between the used versus not-used pond categories. Principal Component Analysis (PCA) was used to visualize how the water chemistry variables covary and to distinguish pond sites with similar water chemical properties from ponds with different water chemical properties. For this, the water chemistry variables were normalized, centred and standardized before being projected into 2-dimensional space. Data manipulation (normality tests were performed using the *shapiro.wilk()* function and standardizing was applied using the *apply()* function) and summary statistics were performed using the ‘dplyr’ package. The boxplots were created using the ‘ggplot2’ package and the PCA was calculated and created using ‘prcomp’ and ‘factoextra’ packages in RStudio.

Multivariate analysis of similarities (ANOSIM) tests were performed on the water chemistry data to test whether water chemistry conditions differed significantly **1)** between the two whooping crane pond-use categories (used vs. not used) and **2)** between seasons (June vs. September). The R package ‘vegan’ was used to conduct the ANOSIM tests and was performed on Euclidean distance matrices with 99999 random permutations. A follow-up univariate non-parametric Mann-Whitney U test was performed independently on each water chemistry variable for the ANOSIM tests that resulted in a significant global R value ($p \leq 0.05$).

Regional comparisons in water chemistry variables were also explored for the 7 respective subregions, however, due to the low number of sites sampled within Seton Creek and Alberta West ($n=2$ and 4 , respectively), we do not present major interpretations of this comparison in the Results and Interpretation section (see section A3.2.8; Table A3.2.11 and Figures A3.2.11 to A3.2.19).

A3.2.5 Results and Interpretation

A3.2.5.1 Fort Smith meteorological conditions in 2021

Daily average air temperatures measured in Fort Smith, NT, in 2021 ranged from -38°C to +25°C (Figure A3.2.4a). February, April and December were colder on average, and January, June and October were warmer on average compared to the climate normal (1981-2010). Marked warming occurred from late February to mid-May with temperatures ranging between -30°C to +5°C. Monthly precipitation in 2021 did not deviate substantially from the climate normal, except for greater-than-average precipitation in April, September and November and lower-than-average precipitation in August and October (Figure A3.2.4b). Two discernable multi-day precipitation events occurred, one in mid-June, shortly after the depth loggers were deployed, and the other in mid-September, shortly after the loggers were collected. They are identified by a sharp rise in the daily cumulative precipitation (Figure A3.2.4b). Monthly average snow depth from October 2020 to March 2021 was greater than the climate normal for the months of October-January and April (Figure A3.2.5).

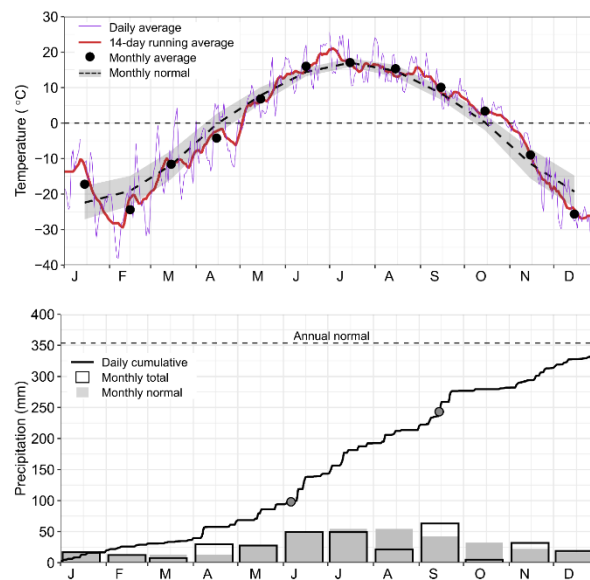


Figure A3.2.4. Meteorological data for January 1, 2021, to December 31, 2021, recorded near the Fort Smith airport in Northwest Territories, Canada (Climate ID: 2202202; WMO ID: 71362). The upper panel shows air temperature (°C) expressed as a daily mean (purple line), 14-day running mean (red line) and a monthly mean (black-filled circles) with respect to the monthly normal (1981-2010) identified by a black dashed line and the associated standard deviation (grey area). The bottom panel includes daily cumulative precipitation (mm) for 2021 represented by the solid black line as well as monthly total precipitation (black-outlined bars), with respect to the monthly normal (1981-2010) identified by the grey bars. Grey-filled points that fall along the cumulative daily precipitation line indicate the timing of sample collection (June 3, 4 and September 18, 19 of 2021).

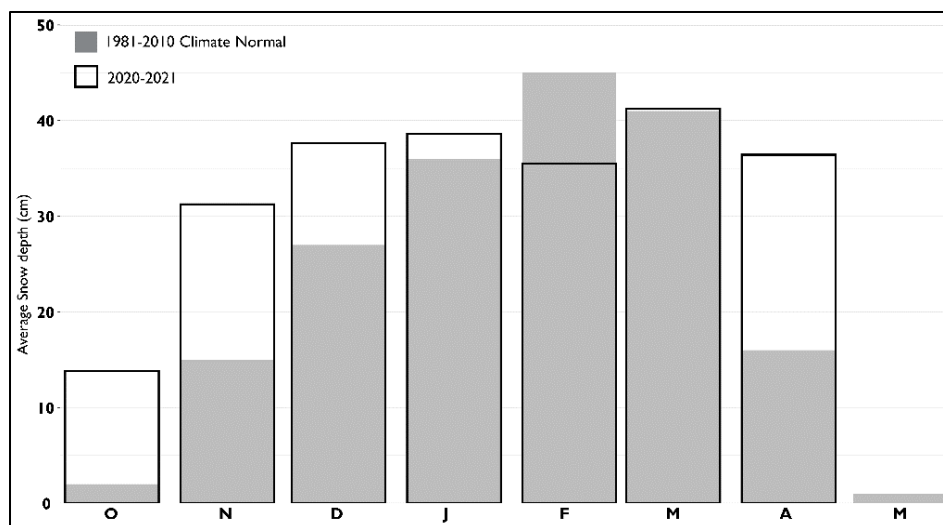


Figure A3.2.5. Snow depth (cm) data from October 2020 to May 2021 recorded near the Fort Smith airport in Northwest Territories, Canada (Climate ID: 2202202; WMO ID: 71362). Monthly averages are displayed as black-outlined bars with respect to the climate normal (1981-2010; grey-filled bars).

A3.2.5.2 Water isotope composition, pond water balance (E/I) and input sources (δ_i)

Water isotope compositions of the 50 ponds range from -19.3 to -13.3‰ in $\delta^{18}\text{O}$ and -154.6 to -127.4‰ in $\delta^2\text{H}$ in June and -19.2 to -11.0‰ in $\delta^{18}\text{O}$ and -151.2 to -112.1‰ in $\delta^2\text{H}$ in September (Figure A3.2.6a; Table A3.2.3). In $\delta^{18}\text{O}$ - $\delta^2\text{H}$ space, the lake water isotope compositions fall along the Local Evaporation Line (LEL) and are positioned between δ_p (mean annual isotope composition of precipitation) and δ_{SSL} (terminal basin steady-state isotope composition; Figure A3.2.6a). From June to September, pond water isotope compositions move from below to above the LEL and towards δ_{PS} , suggesting a potential shift in dominant input sources from snow to rain. Groundwater, which is assumed to be a mixture of precipitation that falls across the recharge zone (i.e., the Caribou Mountains), may also be an important input source to the ponds. A smaller shift along the LEL towards δ_{SSL} also occurs from June to September indicative of increased evaporation during the summer, however, this is less prominent than the shift towards δ_{PS} .

Water isotope ‘metrics’, including evaporation-to-inflow ratios (E/I) and δ_i values, were used to characterize pond water balances and identify dominant input sources. In 2021, pond E/I ratios are all less than 0.5 and span a near-identical range (0.08 - 0.44) and mean (0.223) in June compared to September (range: 0.04 - 0.49; mean: 0.220; Figure A3.2.6b, Table A3.2.3). Peak evaporative influence during the open-water season likely was not captured because

samples for water isotope composition were not collected during mid-summer. Nevertheless, extremely low E/I ratios (or positive water balances where inflow >> evaporation) that persisted during the 2021 season suggests that the ponds receive continuous input at rates that substantially exceed losses via evaporation.

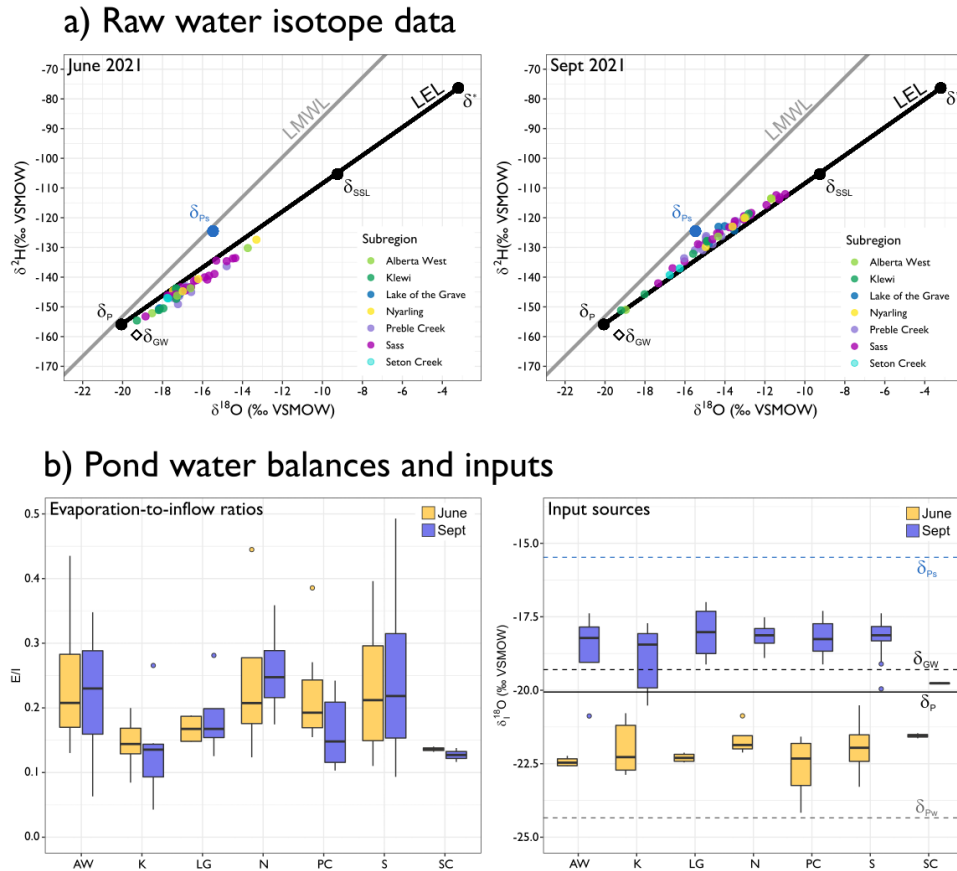


Figure A3.2.6. a) Water isotope compositions ($\delta^{18}\text{O}$ and $\delta^2\text{H}$) for pond sampling sites in the Whooping Crane Nesting Region (WCNR) measured on June 3-5 and on September 18-19, 2021. Coloured data points indicate respective subregions according to the legend. LMWL refers to the Local Meteoric Water Line ($\delta^2\text{H} = 6.7 \delta^{18}\text{O} - 19.2$; Wolfe et al., 2007b) and the LEL is the predicted Local Evaporation Line ($\delta^2\text{H} = 4.75 \delta^{18}\text{O} - 61.2$) constrained by the mean annual isotope composition of precipitation (δ_p), the terminal basin steady-state isotope composition (δ_{SSL}) and the limiting non-steady-state isotope composition of a water body approaching complete desiccation (δ^*) (Table A3.2.4). Other points are also included: δ_{PS} as the mean isotope composition of precipitation collected in Fort Smith from May-September of 1960-1969 (IAEA, 2022) and δ_{GW} as the mean isotope composition of groundwater obtained from Déri-Takács et al. (2022). **b)** Boxplots comparing the evaporation-to-inflow ratios and estimated oxygen isotope compositions of input water ($\delta^{18}\text{O}_i$) for ponds sampled in the WCNR in June and September 2021. These two water isotope metrics were estimated using equations found in Appendix A1. Select isotope parameters included in the bottom right plot include δ_{PS} (isotope composition of summer precipitation), δ_{GW} (isotope composition of groundwater), δ_p (isotope composition of mean annual precipitation) and δ_{PW} (isotope composition of winter precipitation collected in Fort Smith from October-April of 1960-1969; IAEA, 2022) which are indicated with dashed blue, dashed black, solid black and dashed grey lines, respectively.

Calculated $\delta^{18}\text{O}_i$ values reveal a marked increase from June to September (significant at $\alpha = 0.05$; Mann-Whitney U test; $p\text{-value} = 7.97 \times 10^{-18}$; Figure A3.2.6b) indicating a shift in dominant source waters from a more isotopically-depleted input source (i.e., snowmelt) to a more isotopically-enriched input source (i.e., rainfall and/or groundwater). Rainfall is likely the predominant input source in September because most of the pond water δ_i values are higher than δ_{GW} . Additional information, such as $[\text{Ca}]$ and $[\text{SO}_4^{2-}]$, may better characterize the role of groundwater in maintaining the positive pond water balances.

Table A3.2.3. Measured water isotope compositions and calculated δ_i and evaporation-to-inflow ratios for 50 pond sites located in WCSR sampled in June (3-5) and September (18-19) of 2021.

Pond ID	June 2021				September 2021			
	$\delta^{18}\text{O}$	$\delta^2\text{H}$	$\delta_i (^{18}\text{O})$	E/I	$\delta^{18}\text{O}$	$\delta^2\text{H}$	$\delta_i (^{18}\text{O})$	E/I
N1.0	-17.44	-146.02	-21.69	0.17	-15.25	-129.29	-18.09	0.13
N1.1	-14.79	-136.32	-23.39	0.44	-14.17	-126.17	-18.70	0.24
N3.0	-16.31	-141.03	-21.76	0.24	-14.95	-127.99	-18.05	0.15
N3.2	-17.09	-143.41	-21.13	0.17	-14.27	-125.07	-17.96	0.19
N4.0	-17.33	-147.39	-22.78	0.22	-15.11	-130.93	-19.11	0.19
N4.1	-17.17	-146.37	-22.60	0.22	-14.62	-128.89	-19.12	0.23
N5.0	-17.09	-145.21	-22.12	0.21	-14.80	-128.65	-18.63	0.19
N5.3	-17.18	-145.74	-22.20	0.21	-13.53	-124.24	-19.12	0.32
N6.0	-17.23	-149.09	-24.17	0.29	-15.50	-130.94	-18.36	0.13
N6.3	-16.57	-145.01	-23.53	0.30	-13.46	-121.12	-17.62	0.23
N7.0	-15.76	-140.83	-23.29	0.35	-12.68	-118.38	-17.84	0.32
N7.1	-15.89	-140.05	-22.35	0.30	-13.04	-119.47	-17.65	0.27
N8.0	-17.26	-145.28	-21.73	0.18	-14.94	-126.18	-17.30	0.11
N8.1	-17.50	-146.94	-22.05	0.18	-14.95	-126.76	-17.53	0.13
N10.0	-18.14	-150.46	-22.41	0.16	-14.32	-123.06	-17.00	0.14
N10.2	-18.19	-150.78	-22.45	0.16	-14.01	-122.85	-17.42	0.18
N11.0	-15.40	-138.91	-23.13	0.37	-11.92	-115.73	-18.15	0.43
N11.1	-14.36	-133.58	-22.87	0.45	-11.29	-113.82	-18.65	0.55
N12.0	-17.60	-146.06	-21.34	0.15	-16.02	-134.70	-19.10	0.14
N12.1	-17.75	-146.13	-21.05	0.13	-16.62	-137.00	-19.03	0.10
N14.0	-17.09	-145.08	-22.04	0.20	-13.40	-121.92	-18.16	0.27
N14.4	-16.83	-143.07	-21.58	0.20	-16.04	-133.64	-18.59	0.11
N15.0	-16.85	-143.52	-21.78	0.21	-13.51	-121.19	-17.56	0.23
N15.1	-15.31	-134.43	-20.51	0.25	-11.43	-112.41	-17.38	0.43
N16.0	-17.56	-146.29	-21.54	0.16	-13.55	-122.45	-18.11	0.26
N16.1	-15.67	-139.51	-22.66	0.33	-12.72	-119.12	-18.15	0.34
N18.0	-17.48	-144.58	-20.87	0.14	-14.96	-129.85	-18.90	0.19
N18.1	-13.30	-127.39	-21.95	0.51	-11.67	-113.64	-17.52	0.41
N19.0	-13.74	-130.20	-22.56	0.50	-11.69	-113.51	-17.38	0.40
N19.1	-18.53	-152.14	-22.37	0.14	-18.96	-150.99	-20.88	0.07
N21.0	-18.85	-153.23	-22.17	0.12	-17.34	-142.07	-19.95	0.10
N21.1	-16.86	-144.50	-22.32	0.23	-13.07	-120.58	-18.17	0.30
N22.0	-17.33	-144.12	-20.96	0.15	-14.90	-127.79	-18.04	0.15
N22.2	-17.32	-143.72	-20.78	0.14	-14.89	-128.03	-18.16	0.16
N23.0	-17.63	-146.37	-21.44	0.15	-15.35	-128.90	-17.76	0.11
N23.1	-17.35	-145.76	-21.77	0.18	-14.39	-125.22	-17.81	0.18
N24.0	-16.57	-143.27	-22.38	0.25	-14.63	-127.27	-18.31	0.18
N24.1	-16.43	-141.30	-21.60	0.23	-13.78	-123.21	-18.01	0.23
N26.0	-17.31	-147.28	-22.74	0.22	-12.81	-118.66	-17.72	0.30
N26.1	-17.96	-150.49	-22.88	0.19	-15.59	-132.12	-18.73	0.14
N29.0	-16.60	-143.73	-22.59	0.26	-14.34	-126.35	-18.44	0.21
N29.1	-17.27	-146.22	-22.23	0.20	-12.96	-119.82	-18.00	0.30
N30.0	-19.29	-154.60	-21.90	0.09	-18.02	-145.75	-20.32	0.09
N30.1	-18.18	-151.11	-22.65	0.17	-19.20	-151.19	-20.52	0.05
N36.0	-16.99	-144.75	-22.12	0.21	-13.62	-122.99	-18.23	0.26
N36.2	-16.21	-140.63	-21.77	0.25	-13.02	-120.10	-18.03	0.30
N38.0	-14.79	-134.62	-22.13	0.37	-11.25	-113.07	-18.26	0.53

N38.1	-14.48	-133.71	-22.53	0.42	-10.98	-112.07	-18.36	0.58
N39.0	-17.76	-147.01	-21.46	0.14	-16.26	-137.05	-19.73	0.15
N39.3	-17.72	-147.14	-21.63	0.15	-16.76	-139.28	-19.80	0.13

Table A3.2.4. Parameters used to construct and that comprise the WCNR isotope framework.

Parameter	Value(s)	Source
h (%)	66.6	www.climate.weather.gc.ca
T (K)	288.1	www.climate.weather.gc.ca
α^* (^{18}O , ^2H)	1.0102, 1.0905	Horita and Wesolowski (1994)
ε^* (^{18}O , ^2H) (‰)	10.2, 90.5	Gonfiantini (1986)
ε_k (^{18}O , ^2H) (‰)	4.7, 4.2	Gonfiantini (1986)
δ_{As} (^{18}O , ^2H) (‰)	-25.5, -197.1	Gibson and Edwards (2002)
δ^* (^{18}O , ^2H) (‰)	-3.2, -76.4	Gonfiantini (1986)
δ_{SSL} (^{18}O , ^2H) (‰)	-9.2, -107.3	Gonfiantini (1986)
δ_{P} (^{18}O , ^2H) (‰)	-20.1, -155.9	GNIP (IAEA, 2022)
δ_{Ps} (^{18}O , ^2H) (‰)	-15.5, -124.5	GNIP (IAEA, 2022)
δ_{GW} (^{18}O , ^2H) (‰)	-19.3, -159.4	Déri-Takács et al. (2022)
LWML	$6.7 \times \delta^{18}\text{O} - 19.2$	Wolfe et al. (2007b)
LEL	$3.78 \times \delta^{18}\text{O} - 81.4$	

^aisotope composition of groundwater was extracted from Déri-Takács et al. (2022) who sampled 6 springs within the WCNR in 2013. ^bisotope composition of precipitation collected in Fort Smith, NT from 1960-1969 extracted from the Global Network for Isotopes in Precipitation (GNIP; IAEA (2022); accessible at <https://nucleus.iaea.org/wiser/index.aspx>).

A3.2.5.3 Pond level variation

Depth loggers recorded hourly measurements at the 50 ponds from June 8 to September 17, 2021, capturing pond-level fluctuations caused by various hydrological processes (Table A3.2.5, Figure A3.2.7). Pond depth ranged between 0.15 and 1.06 m, with an average of 0.65 m. Note that loggers were unable to be retrieved from ponds N10.2 and N22.2.

Pond levels were remarkably stable across all subregions during the 2021 open-water season. One major rain event from June 16-19 resulted in pond-level rise in most of the subregions by as much as 13 cm (N6.0). Two ponds (N28.0 and N6.0) in Nyarling showed marked pond-level rises of 13.8 and 15.3 cm, respectively, in early July coincident with a rainfall event. Varying pond catchment size, hydrologic connectivity and the localized influence of rainfall across a vast region may explain the varying influence of rainfall on pond depth across the WCNR. Evaporation and outflow after rainfall events cause pond level declines that can be differentiated by the overall flashiness or rate of change. Declines caused by outflow are most rapid in the Nyarling ponds following the two large rainfall events in mid-June and mid-July. Evaporation on the other hand, occurs over a longer period during the warmest months and periods of minimal rainfall. For example, a number of ponds in Preble Creek showed prolonged declines from mid-July to mid-August. However, the average drawdown recorded at ponds across all 7 subregions was only 3.8 cm (s.d. = 3.7).

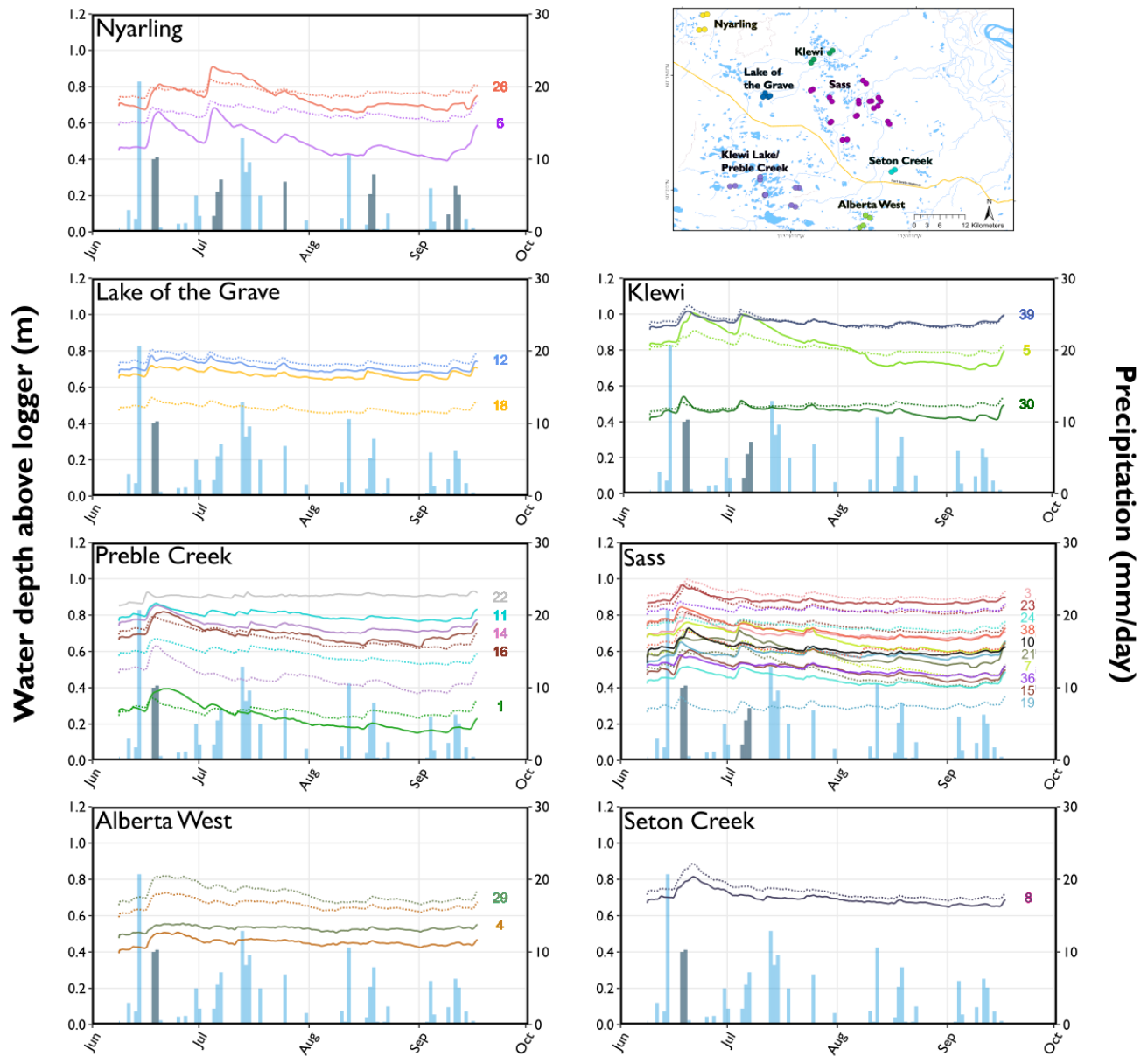


Figure A3.2.7. Temporal variation in pond-levels in the WCNR from June 8 to September 17, 2021. Also included is daily precipitation measured at Fort Smith Airport (Climate ID: 2202202; WMO ID: 71362). Panels are divided by subregion according to the map in the upper right panel. The solid lines represent the category of ponds that have been recently used (but not in 2021) by whooping crane and the dotted lines represent the ponds that have not been used since at least 1966. Some rainfall events are coloured dark blue to identify influential events that resulted in pond-level increases.

Table A3.2.5. Pond level metrics for 50 ponds located in the Whooping Crane Nesting Region (Alberta-Northwest Territories) in 2021.

Pond ID	Mean (m)	Initial (m)	Minimum (m)	Maximum (m)	Min-Max (m)	Min-Initial (cm)
N1.0	0.24	0.26	0.15	0.40	-0.25	-11.6
N1.1	0.27	0.25	0.23	0.35	-0.12	-2
N3.0	0.70	0.68	0.66	0.83	-0.17	-1.5
N3.2	0.91	0.89	0.88	1.01	-0.13	-1.3
N4.0	0.45	0.39	0.40	0.51	-0.12	0
N4.1	0.66	0.59	0.59	0.73	-0.14	0
N5.0	0.82	0.82	0.69	1.0	-0.33	-13.5
N5.3	0.82	0.80	0.76	0.91	-0.15	-3.8
N6.0	0.50	0.45	0.39	0.69	-0.31	-6.2
N6.3	0.65	0.59	0.59	0.73	-0.15	0
N7.0	0.67	0.68	0.59	0.77	-0.18	-8.6
N7.1	0.56	0.57	0.45	0.73	-0.27	-11.7
N8.0	0.70	0.67	0.64	0.82	-0.18	-2.9
N8.1	0.73	0.72	0.67	0.89	-0.22	-4.1
N10.0	0.61	0.54	0.54	0.73	-0.19	0
N10.2			did not retrieve logger			
N11.0	0.80	0.77	0.76	0.87	-0.11	-0.9
N11.1	0.58	0.58	0.52	0.67	-0.15	-5.4
N12.0	0.71	0.68	0.67	0.78	-0.11	-0.7
N12.1	0.74	0.72	0.71	0.81	-0.10	-0.7
N14.0	0.74	0.74	0.69	0.86	-0.17	-4.8
N14.4	0.46	0.48	0.36	0.63	-0.27	-11.8
N15.0	0.50	0.47	0.42	0.61	-0.19	-4.7
N15.1	0.74	0.74	0.69	0.82	-0.13	-4.8
N16.0	0.70	0.66	0.62	0.82	-0.20	-4.2
N16.1	0.68	0.69	0.61	0.79	-0.19	-8.5
N18.0	0.67	0.65	0.64	0.73	-0.09	-1.4
N18.1	0.48	0.47	0.45	0.55	-0.10	-2.4
N19.0	0.56	0.52	0.48	0.66	-0.18	-4.3
N19.1	0.30	0.27	0.23	0.36	-0.14	-3.8
N21.0	0.59	0.57	0.47	0.72	-0.25	-9.6
N21.1	0.50	0.49	0.40	0.65	-0.25	-9.7
N22.0	0.91	0.85	0.85	0.95	-0.09	-0.1
N22.2			did not retrieve logger			
N23.0	0.88	0.87	0.85	0.97	-0.12	-1.6
N23.1	0.84	0.83	0.80	0.98	-0.18	-3.4
N24.0	0.45	0.42	0.40	0.52	-0.13	-2.8
N24.1	0.74	0.72	0.71	0.79	-0.09	-1.8
N26.0	0.74	0.69	0.65	0.92	-0.26	-3.9
N26.1	0.77	0.73	0.73	0.86	-0.13	0
N29.0	0.53	0.48	0.48	0.57	-0.09	0
N29.1	0.72	0.66	0.65	0.83	-0.17	-0.6
N30.0	0.46	0.41	0.40	0.55	-0.14	-0.4
N30.1	0.49	0.44	0.44	0.53	-0.10	0
N36.0	0.51	0.51	0.46	0.58	-0.13	-5.4
N36.2	0.83	0.80	0.79	0.88	-0.10	-1.5
N38.0	0.72	0.75	0.65	0.85	-0.20	-9.5
N38.1	0.62	0.62	0.56	0.72	-0.16	-5.9
N39.0	0.95	0.91	0.91	1.02	-0.11	0
N39.3	0.96	0.94	0.92	1.06	-0.14	-2.3

A3.2.5.4 Pond water quality

The ponds can be described as clear (average turbidity: 0.58 NTU), neutral to slightly alkaline (average pH: 7.9), and oligotrophic to eutrophic based on [TP] (range: 1-65 µg/L; average: 37.6 µg/L) with relatively high concentration of DOC (average: 24.8 mg/L), DIC (average: 28.8 mg/L) and major ions (Table A3.2.6, A3.2.7; Figure A3.2.8). The predominant cation and anion are Ca^{2+} and SO_4^{2-} (Figure A3.3.8), respectively, as is expected for groundwater-fed surface waters underlain by gypsum karst ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$).

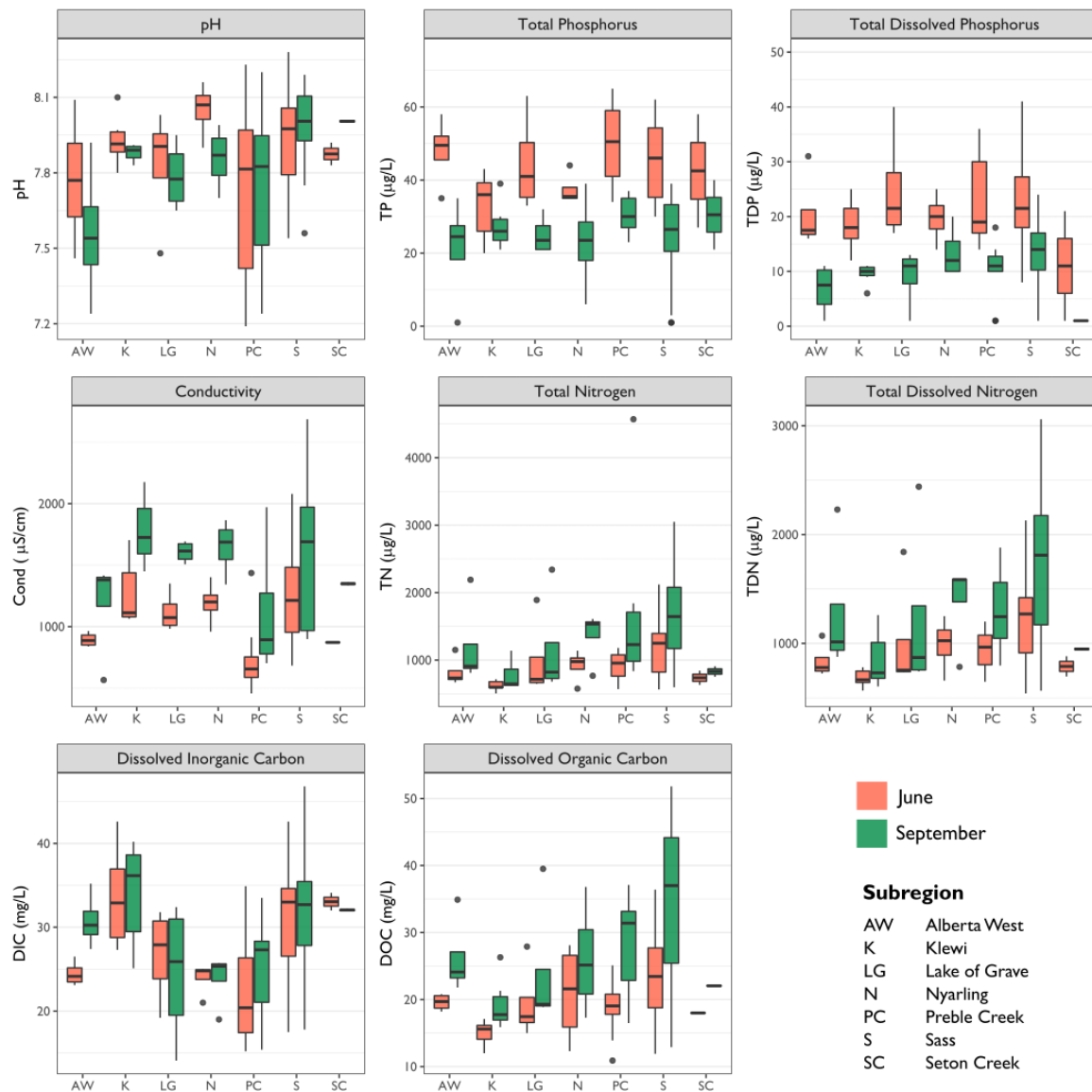


Figure A3.2.8. Seasonal variation (June vs. September) of select water chemistry variables displayed as boxplots for each subregion. See Figures A3.2.11- A3.2.19 for the full suite of water chemistry variables.

Table A3.2.6. Water chemistry variables for 50 ponds located in WCNR on June 3-5, 2021.

Site	Category	Subregion	[DIC] (mg/L)	[DOC] (mg/L)	[HCO ₃] (mg/L)	[SO ₄ ²⁻] (mg/L)	[Ca ²⁺] (mg/L)	[K ⁺] (mg/L)	[Mg ²⁺] (mg/L)	[Na ⁺] (mg/L)	[Cl ⁻] (mg/L)	Cond (µS/cm)	pH	Temp (°C)	Turb (NTU)	DSi (mg/L)	[TP] (µg/L)	[TDP] (µg/L)	[TN] (µg/L)	[TDN] (µg/L)
1	used	PC	27.5	24.4	148.5	151.4	93.5	1.4	13.5	2.6	0.4	457.1	7.3	18.7	0.5	1.1	51	30	1110	1060
1.1	not used	PC	17.1	10.9	93.2	982.0	395.5	1.7	27.4	4.6	1.2	1435.9	7.8	18.4	0.5	3.8	40	17	757	785
3	used	Sass	35.0	19.0	195.0	427.5	204.1	3.2	26.5	7.4	4.5	953.9	8.1	21.4	0.5	1.6	44	21	787	893
3.2	not used	Sass	33.0	18.8	188.6	427.6	203.6	3.1	26.4	7.3	4.5	952.9	8.0	21.3	0.3	1.5	55	24	736	845
4	used	AW	23.1	20.5	130.2	399.8	197.1	2.1	18.6	4.6	3.0	854.6	7.5	20.2	0.4	0.7	35	18	740	753
4.1	not used	AW	24.7	18.9	137.0	437.6	200.7	2.3	21.5	7.3	6.3	919.7	7.7	21.1	0.3	1.2	50	16	671	723
5	used	Klewi	27.8	17.1	160.5	574.5	258.6	2.3	25.0	8.3	7.0	1064.0	8.1	19.2	0.4	2.3	20	16	716	769
5.3	not used	Klewi	27.3	16.0	152.5	587.1	258.6	2.3	27.2	9.3	8.0	1070.1	7.9	18.8	0.4	2.7	43	20	705	780
6	used	N	24.7	26.1	135.3	701.0	307.1	1.4	37.7	4.2	2.0	1193.6	8.1	19.6	1.0	5.1	35	25	994	1080
6.3	not used	N	21.0	28.1	116.7	488.9	187.0	2.0	38.8	15.2	15.8	959.3	7.9	20.3	0.7	1.8	44	21	1140	1250
7	used	Sass	34.7	31.0	188.6	618.9	210.8	4.5	63.4	70.7	79.2	1456.5	8.0	21.9	1.1	3.4	37	22	1440	1480
7.1	not used	Sass	31.6	18.3	176.7	500.3	187.4	4.7	47.8	49.4	61.8	1242.0	8.0	21.8	0.5	1.0	46	41	1290	1400
8	used	SC	32.0	18.0	176.4	411.9	191.4	2.9	26.1	9.5	7.2	875.6	7.8	18.7	0.4	1.2	58	21	843	881
8.1	not used	SC	34.1	18.0	183.5	409.2	199.0	2.6	26.5	9.3	7.0	867.8	7.9	18.1	0.3	0.4	27	1	632	696
10	used	Sass	42.6	29.2	233.5	235.5	116.2	3.6	34.3	15.2	12.9	750.6	7.7	21.9	0.4	0.8	33	32	1310	1350
10.2	not used	Sass	40.3	27.5	209.3	221.9	110.5	3.5	32.1	14.9	12.7	683.2	7.7	20.5	0.4	0.6	50	37	1250	1250
11	used	PC	21.4	20.8	120.6	327.9	122.5	2.6	31.8	8.0	1.4	732.2	8.0	22.1	0.6	2.8	44	16	1040	1080
11.1	not used	PC	17.1	20.3	98.5	449.9	161.7	2.4	37.5	11.2	3.1	912.8	8.1	22.4	0.5	3.9	60	19	1090	1080
12	used	LG	30.4	17.8	171.1	587.1	279.6	2.4	23.7	4.3	2.0	1127.0	7.5	19.6	0.4	2.3	63	19	671	738
12.2	not used	LG	31.8	17.1	171.9	490.7	234.1	1.7	21.6	4.0	2.0	983.4	7.9	19.6	0.3	1.8	36	24	650	766
14	used	PC	26.9	20.8	144.7	222.1	118.7	1.6	15.4	2.8	0.8	565.5	7.8	19.5	0.8	1.6	56	54	923	991
14.4	not used	PC	34.9	25.1	195.9	250.4	132.2	2.9	26.0	3.9	0.9	664.9	7.9	19.3	0.7	3.9	65	19	1180	1200
15	used	Sass	24.3	19.3	133.2	484.2	201.8	2.4	32.0	12.4	10.8	1006.2	7.7	21.9	0.5	1.7	30	19	833	919
15.1	not used	Sass	27.3	36.4	160.8	790.5	222.2	6.5	92.0	38.4	29.7	1459.0	8.0	21.1	2.1	7.0	62	28	2120	2070
16	used	PC	24.7	13.9	137.7	339.8	166.6	1.8	17.8	3.1	0.8	760.1	8.0	21.1	0.8	2.0	50	36	570	647
16.1	not used	PC	15.2	17.8	85.2	253.6	111.4	1.4	14.5	2.9	1.0	570.0	8.2	21.9	0.9	1.2	60	14	987	939
18	used	LG	25.4	15.0	139.1	519.8	246.4	1.7	20.5	4.0	2.3	1019.6	7.9	21.4	1.1	3.0	46	40	761	741
18.1	not used	LG	19.2	27.9	112.0	806.7	313.2	2.2	39.5	10.7	6.5	1350.2	8.0	20.9	0.6	3.7	33	17	1890	1840
19	used	Sass	17.5	26.6	102.3	1094.3	286.7	4.2	121.9	52.9	45.5	1868.5	8.3	22.0	1.1	11.5	62	33	2060	2130
19.1	not used	Sass	27.7	11.9	154.6	1364.0	136.3	1.8	43.7	12.6	11.0	2078.6	8.1	21.9	0.4	3.4	53	16	572	540
21	used	Sass	34.5	14.2	188.2	1106.7	483.2	2.2	31.7	6.9	6.2	1791.7	7.7	21.2	0.3	2.0	31	20	564	652
21.1	not used	Sass	34.6	18.7	194.0	933.7	394.4	3.3	34.5	8.2	8.8	1598.7	7.8	22.1	1.2	6.6	62	26	978	1060
22	used	PC	18.4	17.8	102.8	295.7	136.6	2.3	16.1	3.2	0.8	639.2	7.2	20.4	0.5	0.5	34	27	775	818
22.2	not used	PC	19.4	17.8	111.3	285.6	133.4	2.1	15.9	3.1	0.7	646.2	7.3	21.1	0.7	0.7	40	31	731	799
23	used	Sass	34.1	17.9	188.3	370.5	188.0	3.1	24.3	6.6	3.5	875.1	8.0	21.4	0.4	1.1	36	17	707	773
23.1	not used	Sass	37.8	22.6	211.9	375.0	182.7	3.7	31.2	9.5	5.9	899.6	8.1	21.2	1.0	2.1	32	16	1010	1080
24	used	Sass	30.9	28.3	178.0	374.4	150.8	2.9	44.9	38.5	50.6	1009.6	8.0	21.9	0.6	1.7	36	26	1370	1360
24.1	not used	Sass	22.6	27.2	123.9	616.8	248.5	3.6	35.7	23.1	25.0	1184.2	7.9	21.4	0.8	0.8	58	8	1200	1290
26	used	N	24.9	17.1	144.0	773.8	267.6	2.5	62.1	36.1	39.6	1401.6	8.1	20.1	0.9	3.0	35	14	960	969
26.1	not used	N	25.0	12.3	138.5	675.0	264.8	1.7	43.5	18.7	18.8	1207.2	8.2	19.4	0.5	1.9	36	19	576	658
29	used	AW	26.5	18.2	151.4	456.4	210.5	2.7	27.1	11.1	10.2	964.2	7.9	20.7	0.5	1.1	58	17	728	803
29.1	not used	AW	23.6	20.8	131.7	359.7	130.6	2.9	38.7	24.4	25.6	839.2	8.1	20.5	1.5	0.8	49	31	1150	1070
30	used	Klewi	42.6	13.7	232.5	868.5	403.1	2.3	41.5	29.6	43.5	1544.1	7.8	16.8	0.5	2.0	40	25	595	566
30.1	not used	Klewi	37.9	12.0	213.4	1093.5	467.8	2.3	38.7	14.9	19.4	1703.4	8.0	18.4	1.4	3.8	37	22	506	635
36	used	Sass	33.0	24.3	188.1	600.7	252.1	2.4	44.1	59.0	80.8	1313.6	7.5	18.0	0.5	2.5	46	18	1250	1190
36.2	not used	Sass	33.2	29.5	185.1	478.0	200.2	4.7	45.7	71.0	113.0	1307.5	7.9	19.3	0.8	1.7	33	21	1530	1530
38	used	Sass	18.9	22.4	106.8	485.8	116.6	3.4	73.3	48.3	46.1	1059.6	8.1	19.9	0.9	1.9	48	27	1380	1330
38.1	not used	Sass	22.6	27.2	122.3	784.2	191.1	5.3	100.3	76.4	80.4	1551.4	8.2	20.4	1.1	6.2	54	18	1830	1810
39	used	Klewi	34.1	15.2	183.1	577.3	282.2	2.1	24.0	6.3	4.1	1117.2	7.9	19.6	0.2	0.9	35	16	590	656
39.3	not used	Klewi	31.7	16.2	174.3	557.9	270.0	2.1	23.6	6.4	4.3	1107.5	7.9	20.0	0.2	0.9	23	12	601	673

Table A3.2.7. Water chemistry variables for 50 ponds located in WCNR on September 18-19, 2021.

Site	Category	Subregion	[DIC] (mg/L)	[DOC] (mg/L)	[HCO ₃ ⁻] (mg/L)	[SO ₄ ²⁻] (mg/L)	[Ca ²⁺] (mg/L)	[K ⁺] (mg/L)	[Mg ²⁺] (mg/L)	[Na ⁺] (mg/L)	[Cl ⁻] (mg/L)	Cond (µS/cm)	pH	Temp (°C)	Turb (NTU)	DSi (mg/L)	[TP] (µg/L)	[TDP] (µg/L)	[TN] (µg/L)	[TDN] (µg/L)
1	used	PC	32.7	31.6	181.4	305.9	145.7	1.9	22.6	4.5	1.1	813	7.5	8.5	4.4	7.2	291	10	4570	1260
1.1	not used	PC	15.4	16.5	91.4	1207.7	456.4	2.8	37.3	6.8	2.0	1971	7.6	7.6	0.4	5.0	36	10	1070	1230
3	used	Sass	35.4	24.9	207.2	351.9	164.1	2.0	25.2	7.1	3.8	939	8.1	8.9	0.3	6.5	1	19	933	991
3.2	not used	Sass	34.7	25.6	207.4	370.0	165.1	2.1	25.6	7.6	3.9	940	8.0	8.7	0.2	6.4	36	5	996	961
4	used	AW	27.4	21.8	155.9	836.6	329.9	3.0	32.7	7.0	4.2	566	7.2	7.7	0.2	5.6	24	10	811	876
4.1	not used	AW	29.7	23.7	170.6	774.2	292.0	3.0	33.9	10.0	8.1	1416	7.5	8.2	0.2	7.2	25	5	919	1070
5	used	Klewi	27.4	21.3	142.6	818.0	315.8	2.1	36.1	13.5	10.5	1549	7.8	8.7	0.3	8.0	39	10	932	1100
5.3	not used	Klewi	25.1	26.3	146.3	757.0	283.9	2.6	37.4	14.3	10.9	1449	7.9	8.9	0.3	7.8	25	11	1140	1260
6	used	N	19.0	22.0	104.0	901.6	321.0	1.8	50.7	6.7	2.6	1614	7.7	9.5	0.5	6.3	22	10	1550	1580
6.3	not used	N	25.6	36.8	140.8	649.5	228.4	2.5	49.6	20.4	18.8	1343	7.8	9.4	0.5	9.4	39	20	1610	1590
7	used	Sass	35.6	42.3	209.6	919.3	250.4	6.3	97.5	98.2	115.7	2081	8.1	9.1	0.4	10.0	33	12	2160	2220
7.1	not used	Sass	31.3	45.8	178.0	649.5	201.6	6.6	63.4	66.9	73.9	1547	8.1	8.9	0.6	5.1	39	20	3050	2390
8	used	SC	31.9	22.1	191.1	602.0	247.3	3.6	38.6	12.8	9.0	1337	8.0	9.1	0.2	5.9	40	1	909	934
8.1	not used	SC	32.2	22.0	189.4	653.7	253.3	3.8	38.4	12.6	9.2	1360	8.0	9.0	0.3	6.4	21	1	752	961
10	used	Sass	46.8	47.8	272.7	279.7	127.9	3.9	44.2	21.7	17.7	939	7.9	8.4	0.3	6.4	28	15	1930	1960
10.2	not used	Sass	43.3	46.9	257.8	264.6	120.8	4.7	40.2	20.7	18.0	899	7.9	8.5	0.4	4.0	23	1	1900	2040
11	used	PC	27.7	37.1	159.7	483.3	162.7	3.1	44.4	11.3	1.6	1048	8.0	8.4	0.6	9.3	30	18	1740	1660
11.1	not used	PC	17.5	35.3	105.4	735.4	222.0	3.0	54.1	16.3	4.5	1347	8.2	8.5	0.4	12.0	30	13	1840	1880
12	used	LG	30.5	19.1	169.3	913.1	364.2	2.2	29.2	4.9	2.7	1667	7.7	8.7	0.2	5.0	21	12	681	760
12.2	not used	LG	32.4	19.5	176.8	797.4	325.3	2.1	29.1	5.6	3.4	1507	8.0	8.7	0.3	5.7	21	13	743	743
14	used	PC	26.6	31.2	144.7	307.2	140.7	1.4	20.4	3.9	0.7	787	8.1	8.3	0.4	7.6	26	12	1270	1250
14.4	not used	PC	33.5	31.8	192.0	230.3	110.9	2.1	24.7	4.4	1.2	703	7.9	8.0	0.4	4.6	37	14	1190	1240
15	used	Sass	28.9	32.6	174.7	747.4	267.0	3.6	52.3	22.7	19.1	1518	8.0	9.5	0.3	8.0	23	17	1320	1540
15.1	not used	Sass	23.4	47.2	138.5	902.7	247.2	7.4	93.1	41.6	31.2	1736	8.1	9.5	0.7	5.1	3	1	2470	2570
16	used	PC	28.5	20.0	160.9	751.7	288.1	2.3	30.8	5.5	1.3	1391	7.9	8.0	2.0	6.4	27	1	953	875
16.1	not used	PC	19.2	33.6	112.5	457.2	177.9	1.8	25.8	5.5	1.5	973	7.8	7.8	0.4	5.0	35	12	1620	1720
18	used	LG	21.3	18.9	122.0	873.7	338.4	1.6	28.7	5.3	3.5	1562	7.9	9.0	0.5	6.1	32	10	899	980
18.1	not used	LG	14.1	39.5	82.4	980.7	338.0	2.6	46.1	13.1	7.7	1692	7.7	8.6	0.3	4.2	26	1	2340	2440
19	used	Sass	17.8	36.4	110.3	1668.0	451.9	4.9	137.3	61.2	49.5	2686	8.0	9.0	0.9	14.5	38	14	2370	2540
19.1	not used	Sass	34.8	12.9	208.0	1260.3	493.2	2.1	40.4	10.7	9.3	2152	8.1	8.9	0.4	4.8	22	15	695	565
21	used	Sass	29.9	15.5	166.1	1163.7	452.7	2.8	33.2	7.6	5.6	1980	7.9	9.0	0.2	5.4	21	12	596	680
21.1	not used	Sass	22.3	24.9	127.5	1120.4	411.1	4.7	37.1	10.1	9.8	1895	8.0	9.0	0.4	9.0	34	13	1230	1230
22	used	PC	26.9	22.5	147.2	304.7	139.7	2.8	19.1	3.9	1.2	775	7.2	8.0	0.2	4.5	32	1	836	796
22.2	not used	PC	27.8	23.9	157.8	307.7	138.7	2.5	18.1	3.5	1.2	773	7.4	8.2	0.3	5.2	23	10	908	985
23	used	Sass	39.7	22.9	234.8	85.3	174.3	2.2	24.2	6.8	3.4	977	8.2	9.5	0.2	6.3	1	14	780	805
23.1	not used	Sass	34.0	31.2	207.1	349.3	154.8	2.2	29.8	9.5	4.6	921	8.1	9.6	0.4	6.1	35	12	1250	1310
24	used	Sass	36.3	43.6	212.6	484.1	170.5	2.5	57.4	54.5	73.7	1363	8.0	8.6	0.3	7.6	26	17	1650	1830
24.1	not used	Sass	24.6	42.0	140.1	838.0	305.6	2.5	45.4	28.6	32.6	1644	7.8	8.7	0.4	6.0	15	3	1640	1800
26	used	N	25.1	28.3	140.9	886.7	281.1	2.8	69.6	45.0	46.1	1760	7.9	9.3	0.6	10.3	6	14	1520	1590
26.1	not used	N	25.7	17.3	148.6	1050.3	362.6	1.7	63.6	19.6	18.8	1865	8.0	9.3	0.4	6.7	25	10	767	784
29	used	AW	30.8	24.5	170.2	716.4	271.9	3.0	34.3	11.3	10.1	1395	7.6	8.1	0.2	7.2	1	11	899	957
29.1	not used	AW	35.2	34.9	199.1	608.0	194.1	3.9	57.2	38.8	42.2	1365	7.9	8.1	1.2	9.3	35	1	2190	2230
30	used	Klewi	40.2	16.7	236.0	1030.9	402.7	3.2	49.8	39.2	52.3	2175	7.9	8.1	0.2	5.4	27	9	617	604
30.1	not used	Klewi	39.3	15.9	228.5	1125.2	462.8	2.9	40.4	12.8	15.3	2038	7.9	8.3	0.4	4.2	30	10	663	732
36	used	Sass	32.6	29.7	183.3	869.9	303.4	3.1	60.9	81.7	110.6	1968	7.6	8.2	0.4	5.9	19	15	1350	1240
36.2	not used	Sass	32.8	37.6	193.0	699.5	244.2	4.8	58.6	88.2	127.2	1785	7.9	8.9	0.3	7.4	27	18	1790	1820
38	used	Sass	29.2	39.9	167.9	787.6	177.6	5.9	123.3	90.5	82.9	1870	8.2	9.6	0.5	10.8	31	2	2050	2160
38.1	not used	Sass	22.6	51.8	140.8	1055.5	253.2	8.9	168.3	111.1	125.4	2472	8.2	9.8	0.6	6.9	28	24	2860	3060
39	used	Klewi	36.6	17.7	208.8	922.1	378.3	2.8	34.0	9.2	7.5	1729	7.9	8.4	0.2	6.4	21	11	625	663
39.3	not used	Klewi	35.7	17.8	204.5	918.1	374.4	2.9	34.9	9.6	7.9	1721	7.9	8.7	0.2	6.9	23	6	630	726

Principal Component Analysis (PCA) was used to visualize patterns of variation in nutrient concentrations (TP, TDP, TN, TDN), ionic concentrations (HCO_3^- , Cl^- , SO_4^{2-} , Ca^{2+} , K^+ , Mg^{2+} , Na^{2+}) and other water chemistry variables (pH, turbidity, DIC, DOC, conductivity) among ponds. PCA axes 1 and 2 capture 35.1% and 18.0% of the total variance, respectively (Figure A3.2.9a). In June, ponds are positioned farther to the left along Axis 1, indicating lower concentrations of major ions (K^+ , Na^+ , Mg^{2+} , Cl^-), DSi, TN and TDN, lower conductivity and pH, and higher concentrations of TP and TDP. The sample scores are positioned farther to the right along Axis 1 in September, indicating higher pH, conductivity and concentrations of ions, TN and TDN, and lower concentrations of TP and TDP (Figure A3.2.9b). The shift of sample scores to the right along Axis 1 reflects a rise of pH between June and September likely due to biological productivity during the summer. The modest shift of sample scores downward along Axis 2 captures a rise in concentrations of Ca^{2+} and SO_4^{2-} possibly due to input of groundwater. The results of an ANOSIM test indicates that pond water chemistry differs significantly (p value < 0.05) between the June and September sampling periods (Table A3.2.8). Individual Mann Whitney U tests (Table A3.2.9) identify that turbidity, conductivity, and concentrations of Ca^{2+} , SO_4^{2-} , Mg^{2+} , TP, TDP, TN, TDN, DOC and DSi differ significantly.

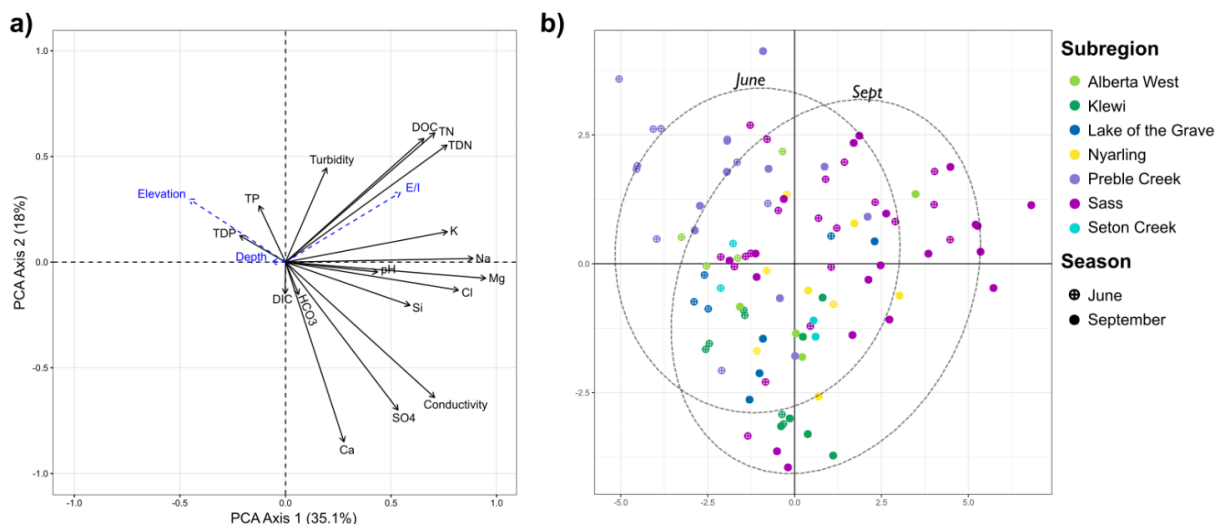


Figure A3.2.9. Principal Component Analysis (PCA) ordination showing seasonal variation in water chemistry variables for ponds in the WCNR. Panel a) shows the association between active (black) and passive (blue) environmental variables and panel b) shows the individual sample scores colour-coded by region according to the legend. 90% confidence ellipses are illustrated for June and September sample scores.

Table A3.2.8. Results of ANOSIM tests on water chemistry variables in June and September 2021.

Season	Comparison	R statistic	P value
June	used, not used	-0.01687	0.749
September	used, not used	-0.01592	0.704
June and September	June, September	0.13120	1x 10⁻⁵

99,999 random permutations were run for each comparison. Bolded p-values are significant at $\alpha = 0.05$.

Table A3.2.9. Mean, standard deviations (s.d.) and p values from a series of Mann-Whitney U tests used to assess whether the water isotope metrics, pond-level metrics and water chemistry variables differ significantly between June and September of 2021. The results are based on 50 ponds sampled in June and September 2021. P values in bold are significant at $\alpha = 0.05$. Shaded values for means in September indicate an overall increase (green) or decrease (red) associated with a significant p-value.

Variable	June		September		P value
	mean	s.d.	mean	s.d.	
δ_i (^{18}O)	-22.21	0.78	-18.42	0.86	7.97 x 10⁻¹⁸
E/I	0.22	0.10	0.22	0.12	5.60 x 10 ⁻⁰¹
Na ⁺ (mg/L)	18.24	19.98	24.48	27.99	1.52 x 10 ⁻⁰¹
Cl ⁻ (mg/L)	18.77	25.88	24.49	34.84	4.34 x 10 ⁻⁰¹
Ca ²⁺ (mg/L)	220.57	90.83	269.70	103.44	1.40 x 10⁻⁰²
SO ₄ ²⁻ (mg/L)	552.49	264.88	735.45	320.11	2.68 x 10⁻⁰³
Mg ²⁺ (mg/L)	36.55	21.67	47.78	29.84	7.87 x 10⁻⁰³
HCO ₃ ⁻ (mg/L)	156.45	37.41	170.36	41.97	9.80 x 10 ⁻⁰²
K ⁺ (mg/L)	2.74	1.06	3.21	1.56	1.33 x 10 ⁻⁰¹
TP ($\mu\text{g/L}$)	44.22	11.55	31.06	38.81	1.57 x 10⁻¹⁰
TDP ($\mu\text{g/L}$)	22.54	9.03	10.52	5.91	5.05 x 10⁻¹²
TN ($\mu\text{g/L}$)	989.98	399.21	1404.28	774.12	2.53 x 10⁻⁰³
TDN ($\mu\text{g/L}$)	1027.96	378.35	1383.76	617.03	3.35 x 10⁻⁰³
pH	7.87	0.37	7.73	0.27	6.61 x 10 ⁻⁰²
Turbidity (NTU)	0.67	0.37	0.48	0.63	7.21 x 10⁻⁰⁶
DIC (mg/L)	28.13	6.95	29.54	7.08	2.68 x 10 ⁻⁰¹
DOC (mg/L)	20.52	5.71	29.02	10.23	1.74 x 10⁻⁰⁵
Silica (mg/L)	2.42	2.03	6.86	2.11	2.14 x 10⁻¹⁴
Conductivity ($\mu\text{S/cm}$)	1091.91	359.81	1480.06	483.41	7.27 x 10⁻⁰⁵

A3.2.5.5 Comparison of hydrological conditions and water quality of used versus not-used ponds

Figure A3.2.10 provides a comparison of the hydrological conditions and water quality of ponds used versus not used for nesting by whooping crane. Mann-Whitney U tests identify no significant difference in water isotope metrics (δ_i and E/I) or pond level metrics (mean depth and drawdown) for the used versus not-used ponds (Table A3.2.10). Similarly, ANOSIM test results identify no significant difference in water chemistry of used versus not-used ponds in June and September 2021 (Table A3.2.8). Overall, there are no measurable differences in hydrological conditions and water quality between the used and not-used ponds.

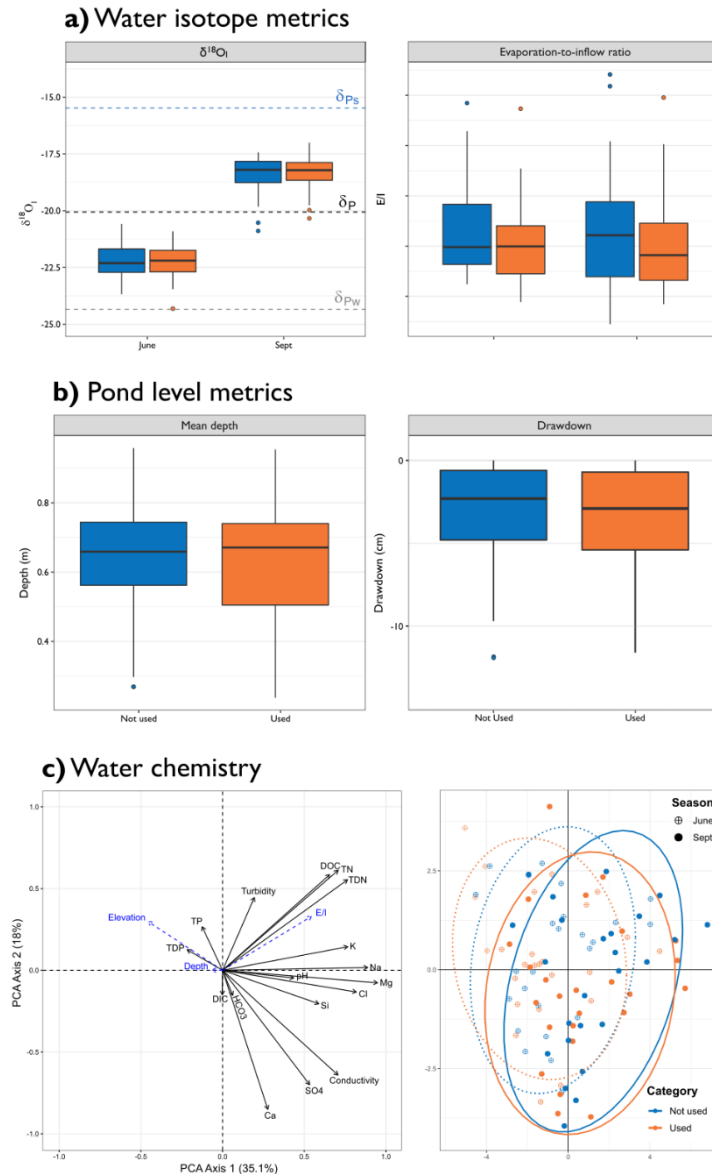


Figure A3.2.10. Compilation of water isotope and pond level metrics and water chemistry to compare ponds that have been recently used (but not in 2021) by whooping crane to ponds that have not been used since at least 1966. **a)** Boxplots comparing the estimated oxygen isotope compositions of input water ($\delta^{18}\text{O}_i$) and evaporation-to-inflow ratios for ponds sampled in the WCNR in June and September 2021. Select isotope parameters included in the top left plot include δ_P (isotope composition of mean annual precipitation), δ_{PS} (isotope composition of summer precipitation) and δ_{PW} (isotope composition of winter precipitation) which are indicated with dashed black, blue and grey lines, respectively. **b)** Boxplots comparing mean pond depth and amount of drawdown. Boxes are grouped by whooping crane use category (blue = not used, orange = used). **c)** Principal Component Analysis (PCA) ordination showing seasonal variation in water chemistry conditions for ponds in the WCNR. The left panel shows the association between active (black) and passive (blue) environmental variables and the right panel shows the individual sample scores colour-coded by pond category where blue represents not-used ponds and green represents used ponds. Ellipses identify 90% confidence levels for pond categories (blue and green) and seasons (June = dotted line, September = solid line). Results from Mann-Whitney U tests are reported in Table A3.2.10.

Table A3.2.10. Mean, standard deviations (s.d.) and p values from Mann-Whitney U tests used to assess whether pond depth, drawdown, δ_i or E/I differ significantly between whooping crane pond use categories in 2021.

Variable	Used		Not used		P value
	mean	s.d.	mean	s.d.	
Pond depth (m)	0.64	0.16	0.65	0.17	0.622
Drawdown (cm)	-4.59	6.10	-3.42	3.56	0.448
δ_i (June)	-22.2	0.78	-22.2	0.79	0.744
δ_i (Sept)	-18.4	0.81	-18.5	0.92	0.432
E/I (June)	0.21	0.09	0.24	0.10	0.893
E/I (Sept)	0.21	0.10	0.24	0.13	0.404

A3.2.6 Conclusions and Recommendations

The hydrology and water quality of ponds in the WCNR showed some variation between June and September 2021. Isotope-based calculations of input sources identify a systematic shift from snow input prior to June to rainfall and possibly groundwater input by September with minimal influence of evaporation on pond water balances ($E/I \ll 1.0$) across all 7 subregions. The remarkably stable pond levels align with isotopic evidence of minimal evaporation during the open-water season of 2021.

While the paired study design was a fruitful exploration of the hydrology and water chemistry between ponds recently used versus not used since at least 1966 by nesting whooping crane, the hydrological conditions and water quality did not differ substantially between these two pond categories. These results have informed a decision to alter the sampling strategy for 2022. We have opted to sample within the subregions most prevalently used by whooping crane (Sass and Klewi) with more spatially comprehensive and higher density of site locations. This strategy aligns with the goal of characterizing the range of hydrological conditions and water quality of ponds used by whooping crane as to understand their potential vulnerability to climate change. We plan to expand this same sampling strategy across the entire whooping crane nesting region in years to come.

A3.2.7 Supplementary table and figures

Table A3.2.11. Summary of water chemistry variables, elevations and pond depths for 7 subregions within the WCNR in June and September 2021.

Variable	Subregion													
	Preble Creek (n=10)		Sass (n=20)		Nyarling (n=4)		Alberta West (n=4)		Klewi (n=6)		Seton Creek (n=2)		Lake of the Grave (n=4)	
Season	June	Sept	June	Sept	June	Sept	June	Sept	June	Sept	June	Sept	June	Sept
Bicarbonate (mg/L)	123.8	145.3	171.5	186.9	133.6	133.6	137.6	174.0	186.1	194.5	180.0	190.3	137.6	148.5
DIC (mg/L)	22.3	25.6	30.8	31.8	23.9	23.9	24.5	30.8	153.6	195.6	33.1	32.1	24.6	26.7
DOC (mg/L)	19.0	28.4	23.5	35.1	20.9	26.1	19.6	26.2	15.0	19.3	18.0	22.1	24.3	19.5
pH	7.7	7.8	7.9	8.0	8.1	7.9	7.8	7.6	7.9	7.9	7.9	8.0	7.8	7.8
Si (mg/L)	2.2	6.7	3.0	7.1	3.0	8.2	1.0	7.3	2.1	6.5	0.8	6.2	5.3	2.7
Turbidity (NTU)	0.7	0.9	0.7	0.4	0.8	0.5	0.7	0.4	0.5	0.3	0.4	0.3	0.3	0.6
Cond (µS/cm)	738.4	1058.1	1252.1	1615.6	1190.4	1645.5	894.4	1185.5	1267.7	1776.8	871.7	1348.5	1607.0	1120.1
SO ₄ (mg/L)	355.8	509.1	614.5	743.3	659.7	872.0	413.4	733.8	709.8	928.5	410.6	627.8	891.2	601.1
Ca (mg/L)	157.2	198.3	214.4	256.8	256.6	298.3	184.7	272.0	161.2	204.5	195.2	250.3	341.5	268.3
K (mg/L)	2.0	2.4	3.6	4.2	1.9	2.2	2.5	3.2	2.2	2.7	2.7	3.7	2.1	2.0
Mg (mg/L)	21.6	29.7	49.3	62.9	45.5	58.4	26.5	39.5	30.0	38.8	26.3	38.5	33.3	26.3
Na (mg/L)	4.5	6.6	31.4	42.3	18.5	22.9	11.8	16.8	12.5	16.4	9.4	12.7	7.2	5.8
Cl (mg/L)	1.1	1.6	34.6	45.9	19.0	21.5	11.3	16.1	14.4	17.4	7.1	9.1	4.3	3.2
TP (µg/L)	50.0	56.7	45.4	24.2	37.5	23.0	48.0	21.3	33.0	27.5	122.1	137.3	25.0	44.5
TDP (µg/L)	26.3	10.1	23.5	12.5	19.8	13.5	20.5	6.8	18.5	9.5	11.0	1.0	9.0	25.0
TN (µg/L)	916.3	1599.7	1210.9	1651.0	917.5	1361.8	822.3	1204.8	618.8	767.8	737.5	830.5	1165.8	993.0
TDN (µg/L)	939.9	1289.6	1247.6	1685.6	989.3	1386.0	837.3	1283.3	679.8	847.5	788.5	947.5	1230.8	1021.3
Temperature (°C)	20.5	8.1	21.1	9.0	19.9	9.4	20.6	8.0	18.8	8.5	18.4	9.1	8.8	20.4
Elevation (masl)	276.7		220.3		238.3		264.8		215.5		237.6		248.0	
Mean depth (m)	0.596		0.644		0.665		0.591		0.739		0.716		0.652	

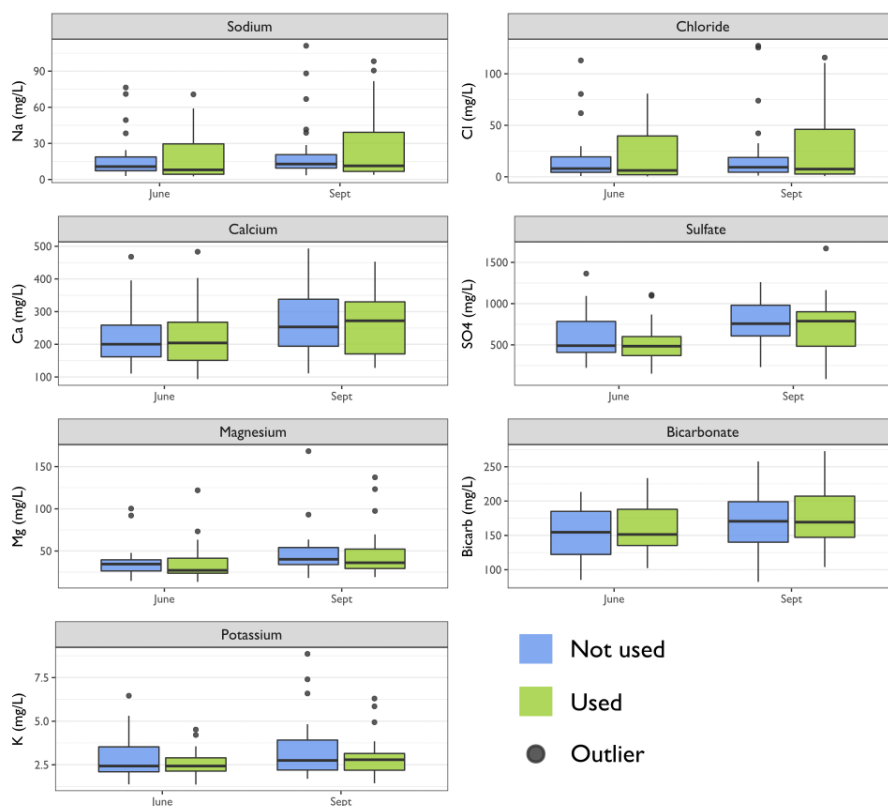


Figure A3.2.11. Boxplots comparing major ion concentrations in ponds of the two whooping crane use categories (blue = not used, green = used) in June and September 2021.

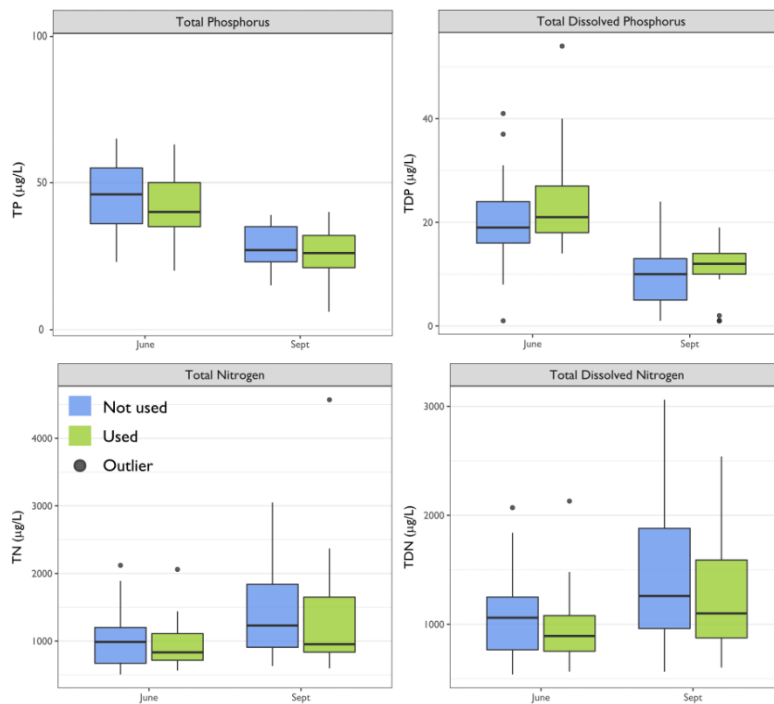


Figure A3.2.12. Boxplots comparing nutrient concentrations of whooping crane use category ponds (blue = not used, green = used) in June and September 2021.

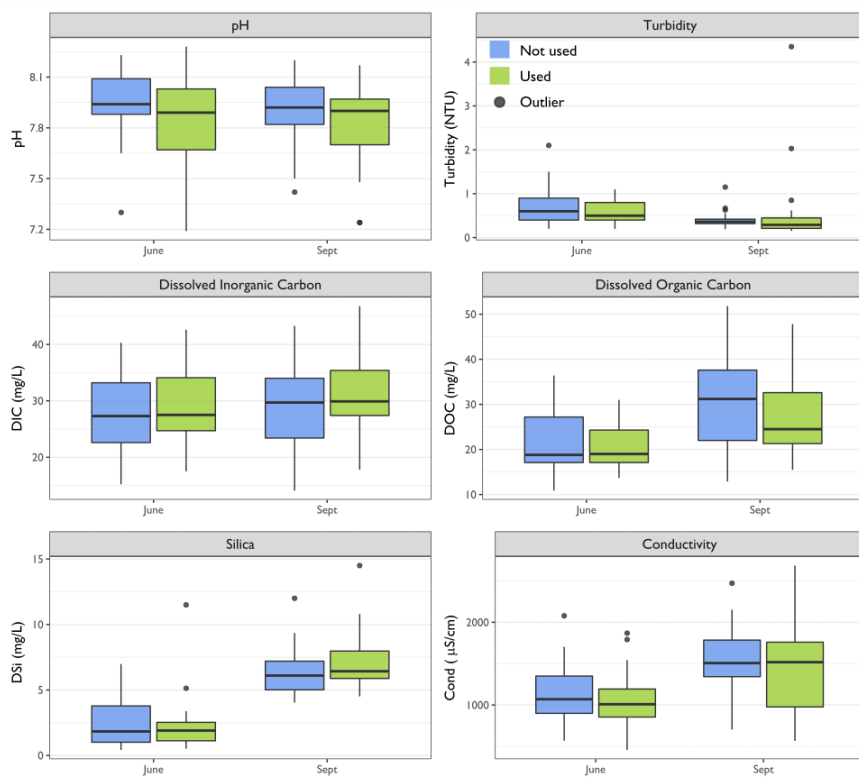


Figure A3.2.13. Boxplots comparing pH, turbidity, DIC, DOC, Silica and Conductivity of whooping crane use category ponds (blue = not used, green = used) in June and September 2021.

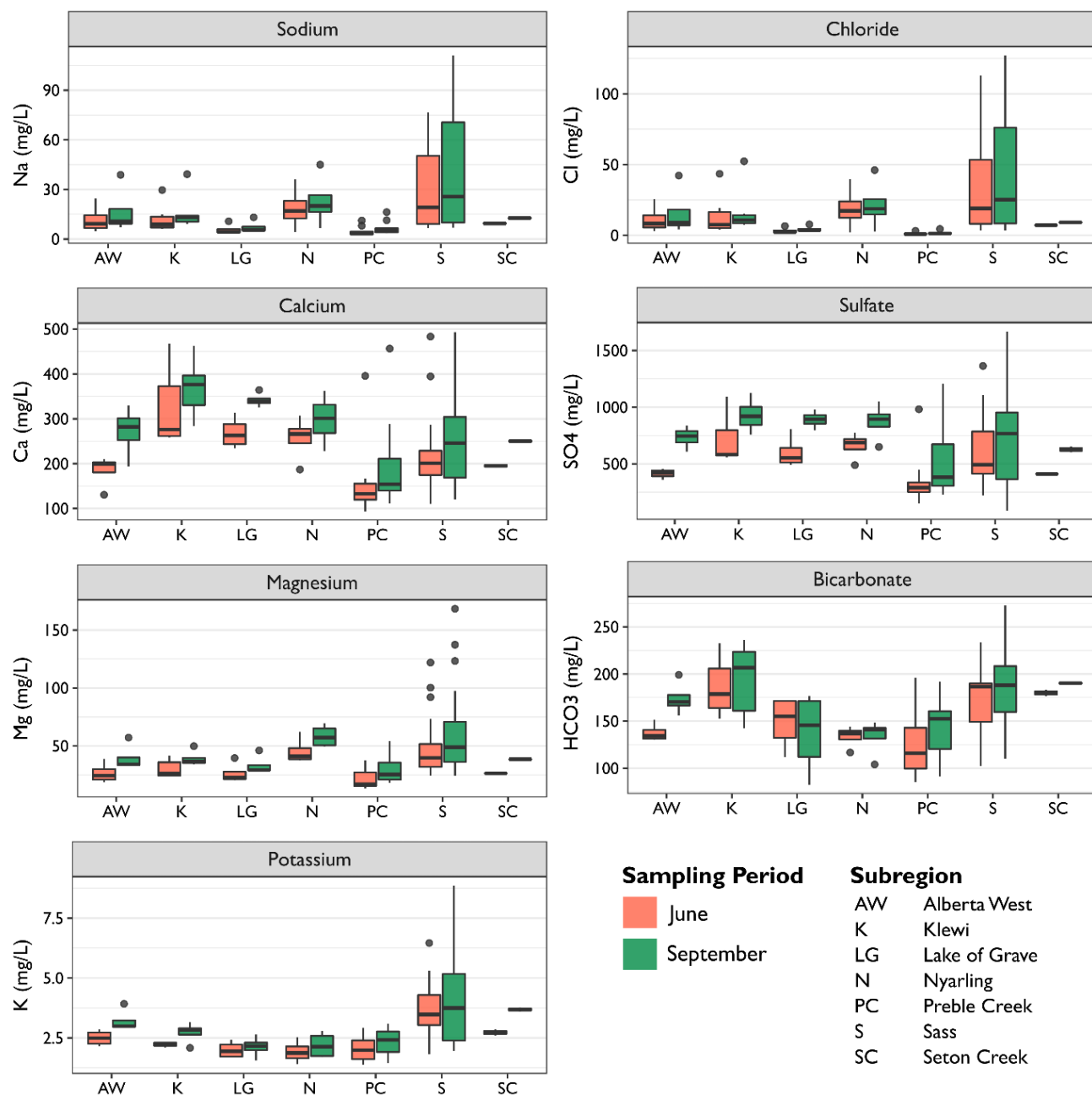


Figure A3.2.14. Boxplots comparing seasonal differences (June, September) in major ion concentrations across 7 WCNr subregions in 2021.

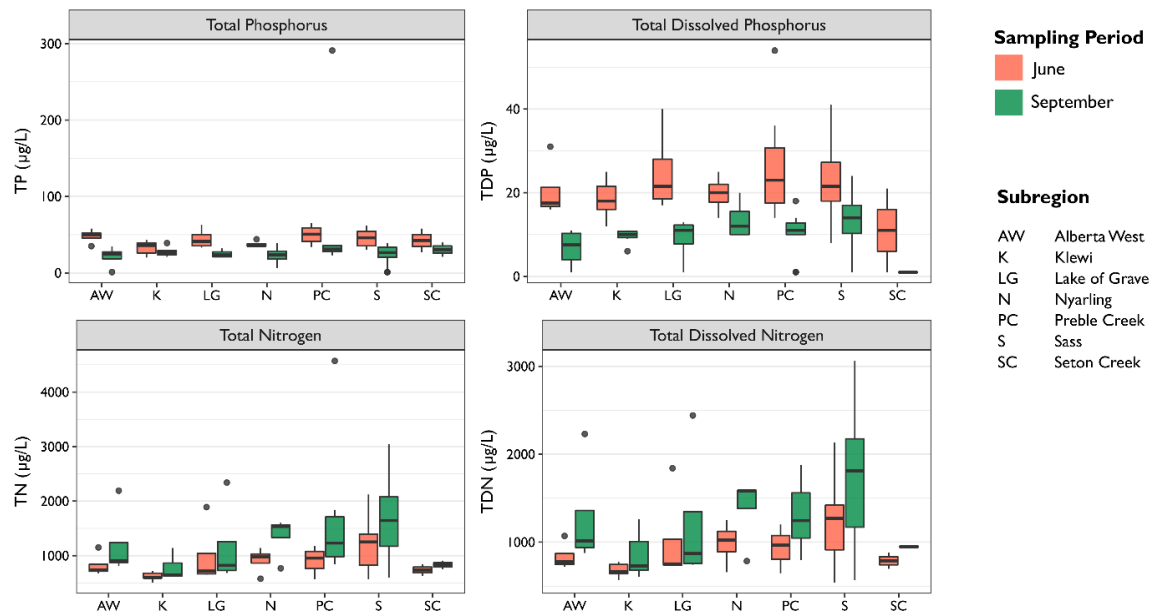


Figure A3.2.15. Boxplots comparing seasonal differences (June, September) in nutrient concentrations across 7 WCNR subregions.

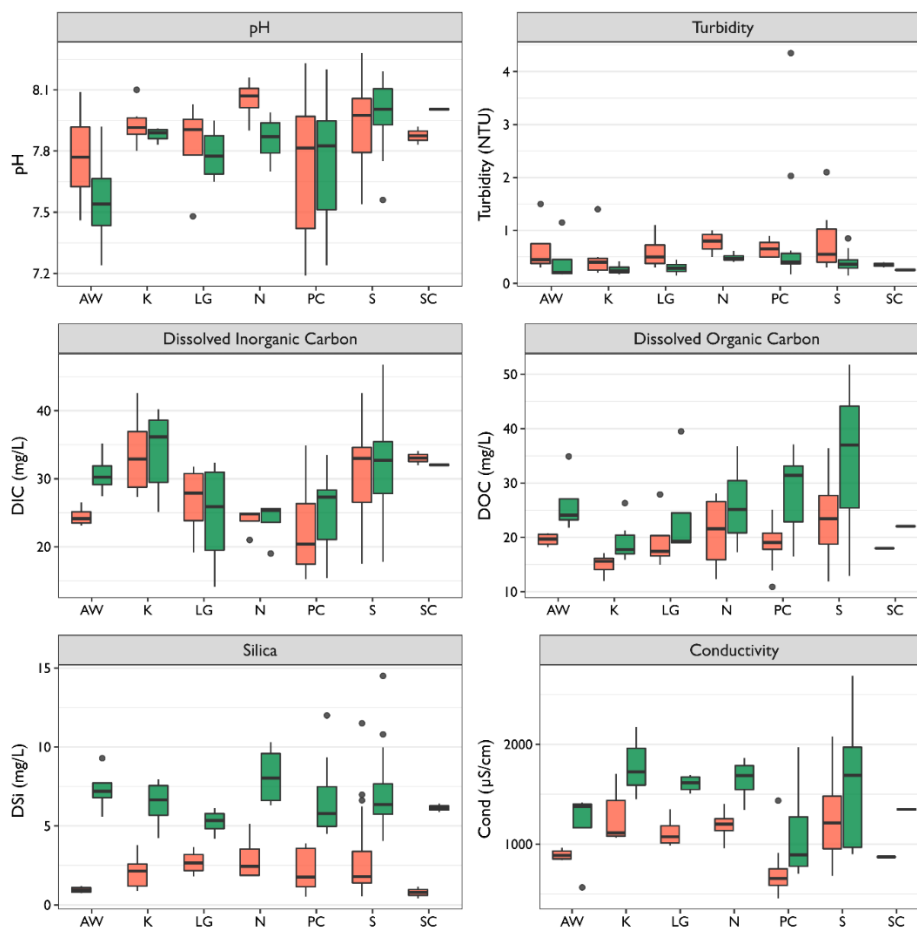


Figure A3.2.16. Boxplots comparing seasonal (June, September) differences in pH, turbidity, DIC, DOC, dissolved silica and conductivity across 7 WCNR subregions in 2021.

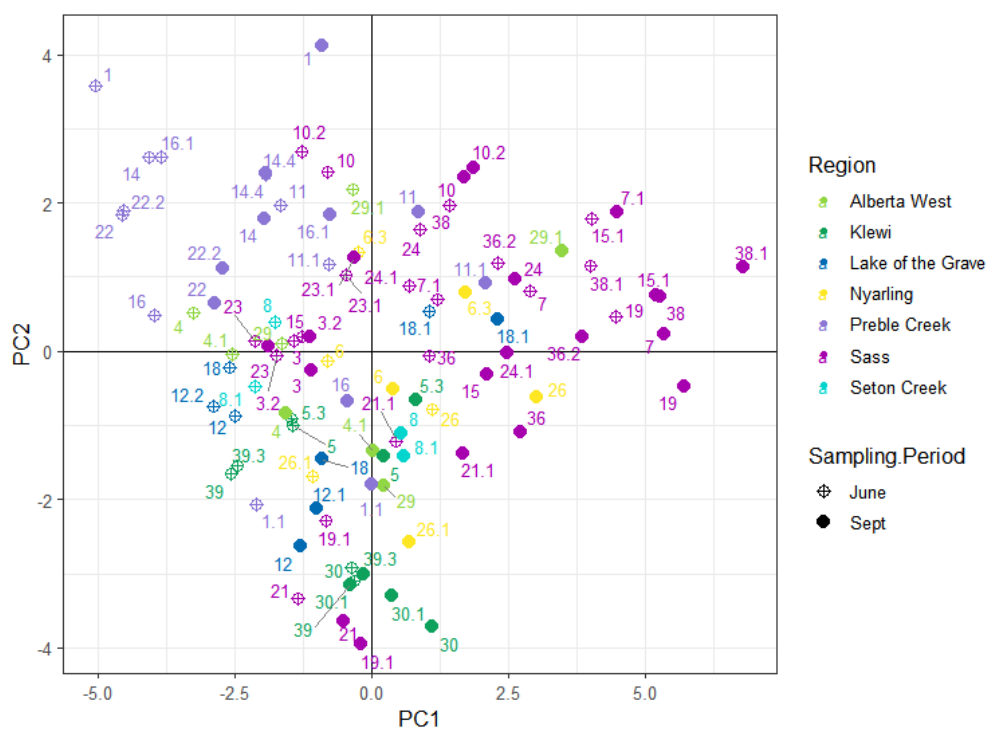


Figure A3.2.17. Same PCA as Figure A3.2.10c with sample scores labelled as with pond site IDs.

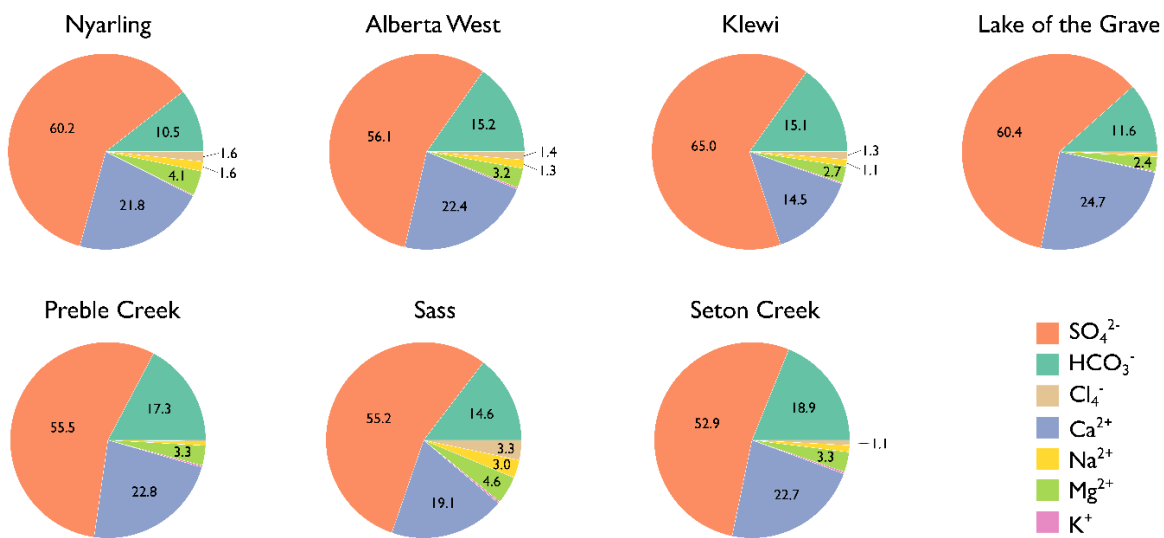
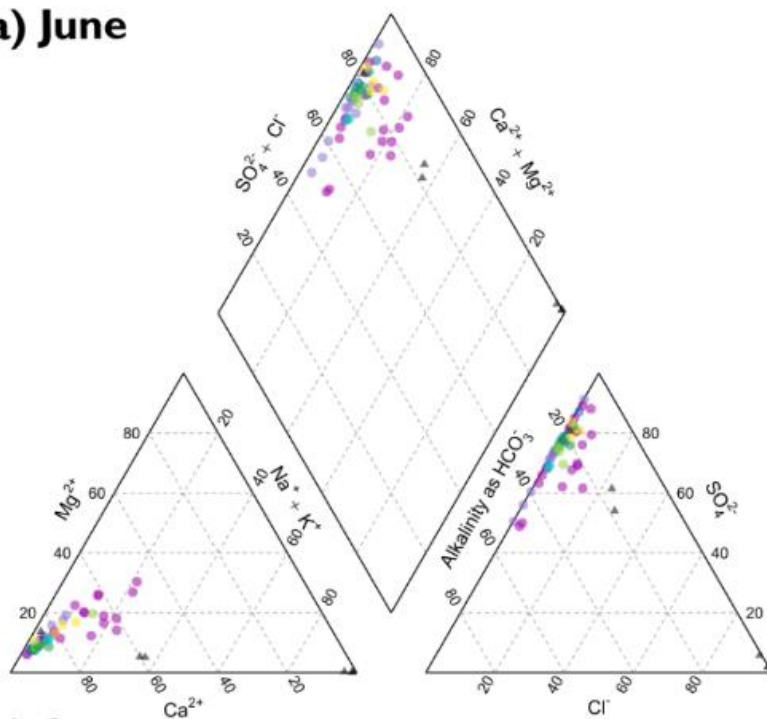


Figure A3.2.18. Pie charts of the proportions of major ions within ponds of each subregion within the WCNR in 2021.

a) June



Subregion

- Alberta West
- Klewi
- Lake of the Grave
- Nyarling
- Preble Creek
- Sass
- Seton Creek
- ▲ Groundwater

b) Sept

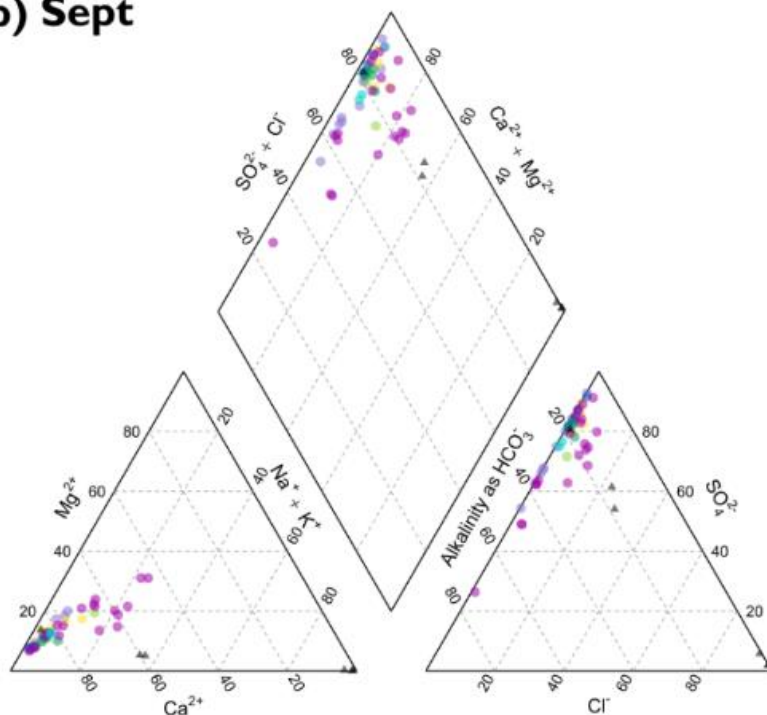


Figure A3.2.19. Piper diagram showing the most abundant cations and anions in the pond water samples collected in 7 subregions within the WCNR in 2021. Raw concentration (mg/L) of major ions were converted to milliequivalents per liter (mEq/L) based on their respective atomic weights, molarity and valence charge before plotting within a piper plot.