The limnological response of Arctic deltaic lakes to alterations in flood regime

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ABSTRACT

Arctic freshwaters are being rapidly altered by global climate change with consequences to hydrology, biogeochemistry, and ecology, but in many cases the trajectory of these changes is poorly understood. We collected a unique 5-year time series of major ion, nutrient, and trace metal data from lakes in the Mackenzie Delta (NT, Canada) to examine limnological changes during a period of variable flood conditions, including years of recent historic high and low peak river levels. Previous work in the Mackenzie Delta has established that lake water chemistry is strongly related to connection time with the river during the period of spring ice-jam flooding or via channel connections through the growing season. We show that differences in peak spring water levels explain differences in lake chemistry in lakes isolated from the channel during the summer. Isolated, macrophyte-rich lakes in the Mackenzie Delta have been shown to be CO2 absorbers during summer. We demonstrate a response to alterations in flood regime by variables related to macrophyte productivity in isolated lakes with the greatest connectivity to the river that suggests productivity declines with increasing connection time. The connectivity of low-elevation lakes, which represent a majority of lake number and area in the Mackenzie Delta, has been projected to increase with climate change. Our work suggests that an increase in connection time may decrease the macrophyte productivity of these lakes, with potential consequences to the CO2 balance of individual lakes and the Mackenzie Delta as a whole.

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Introduction

The pan-Arctic river systems of Eurasia and North America are critical components of the global hydrological and carbon cycles. These watersheds deliver ∼4200 km³ of freshwater (Haine et al. 2015), 34 Tg dissolved organic carbon (DOC), and 57 Tg dissolved inorganic carbon (DIC) to the Arctic Ocean annually, with nearly half of the flux originating from the 6 largest Arctic rivers (Holmes et al. 2012, Tank et al. 2012). Arctic lakes and rivers are important sites of biogeochemical cycling (Cole et al. 2007, Bastviken et al. 2011, Tank et al. 2018), contributing to global fluxes of carbon dioxide (CO2) and methane (CH4; Emmerton et al. 2016, Jammet et al. 2017). Under current climate conditions, freshwater discharge and dissolved carbon fluxes to the Arctic Ocean have increased over the last several decades (Peterson et al. 2002, Tank et al. 2016, Rood et al. 2017), with consequences to marine ecology and freshwater storage (Carmack and Macdonald 2002, Haine et al. 2015) and implications for global circulation patterns (Brown et al. 2019).

At the same time, the release of aged carbon and increases in terrestrial–aquatic connectivity related to permafrost degradation will lead to increased carbon emissions from northern regions (Abbott et al. 2016). Permafrost thaw has been observed across the Arctic (Plaza et al. 2019, Wild et al. 2019) and at increasing rates in some regions (Segal et al. 2016, Kokelj et al. 2017). Thaw releases permafrost-derived organic matter that may be particularly labile to mineralization (Vonk et al. 2013, Drake et al. 2015), with impacts to the hydrology (Walvoord and Kurylyk 2016), biogeochemistry (Frey and McClelland 2009), and ecology (Vonk et al. 2015a) of freshwater systems, and potentially contributing to a positive carbon feedback leading to further warming (Schuur et al. 2015). Considerable uncertainty exists regarding the net carbon balance of Arctic lakes (Abbott et al. 2016), particularly in regard to the importance of terrestrial–aquatic connectivity in determining transport rates and lake responses to releases of permafrost carbon (Tank et al. 2009, Feng et al. 2013, Vonk et al. 2015b, Bogard et al. 2019). Because Arctic and boreal regions contain the greatest abundance of lakes and smaller waterbodies (Lehner and Döll 2004) and ∼50% of global soil carbon stores
(Tarnocai et al. 2009), it is critical to develop our understanding of the changes occurring in the great Arctic drainages.

A common feature of many pan-Arctic rivers is ice-jam flooding (Prowse and Culp 2003), which occurs when increased discharge during snowmelt induces mechanical breakup of solid river ice in the northern parts of the system (Beltaos et al. 2012). Ice-jam floods generate higher river levels than would occur with increased discharge alone (Goulding et al. 2009, Lesack et al. 2013). This process is mediated by climate: higher air temperatures may weaken the ice before discharge rises, resulting in thermal breakup (Beltaos 2013) with reduced peak water levels and earlier ice breakup (Beltaos and Prowse 2009, Lesack et al. 2014, Wang et al. 2017). Combined with multidecadal trends toward earlier onset of spring freshet in tributary basins (Woo and Thorne 2003, Abdul Aziz and Burn 2006), lower spring maximum discharge, and increased winter baseflow (Lesack et al. 2013, Yang et al. 2015), the hydrological regime of ice-jam deltas are potentially susceptible to alterations due to climatic change (Rouse et al. 1997).

Climate-mediated alterations to the hydrological regime of Arctic deltas may likely impact the ecological and biogeochemical functioning of deltaic lakes, which depend on the annual influx of water, sediment, nutrients, and organisms (Junk et al. 1989, Tockner et al. 2000). In the Mackenzie River Delta (NT, Canada), globally the second-largest Arctic delta, the extent and duration of connections between lakes and the river are determined by variation in the elevation of the sill above the channels (Marsh and Hey 1989). The resulting flooding gradient has profound implications for habitat quality and lake functioning, including impacts on water renewal (Lesack and Marsh 2010), water balance (Marsh and Lesack 1996), sedimentation (Marsh et al. 1999), major ion composition (Lesack et al. 1998), transparency (Squires and Lesack 2003a), and primary productivity (Squires and Lesack 2002, Squires et al. 2002, 2009). Variation in primary productivity, particularly of macrophytes, has been shown to influence the quality (Tank et al. 2011) and biogeochemical processing (Febria et al. 2006, Cunada et al. 2018) of DOC, as well as lake metabolic balance (Squires et al. 2009). In many cases, thick macrophyte beds cause the lakes to absorb rather than emit CO$_2$ (Tank et al. 2009), as is more typical of Arctic lakes. By contrast, thermokarst, the process of permafrost degradation in ice-rich soils generally responsible for lake formation in the Mackenzie Delta (Hill et al. 2001), causes deepening of lake basins (Emmerton et al. 2007) and release of recalcitrant coloured DOC (Tank et al. 2011). Lakes highly affected by thermokarst are typically net CO$_2$ emitters (Tank et al. 2009). Variation in flooding and thermokarst result in a mosaic of lakes across the Mackenzie Delta, in which nearby lakes can differ greatly in characteristics and function.

While these previous studies have contributed greatly to understanding of lake processes and function in the Mackenzie Delta, considerable uncertainty remains as to the response of lake environments to alterations in the overall flooding regime, particularly in regard to lakes that are less connected to the surrounding landscape (Lesack and Marsh 2010). We take a step toward addressing this knowledge gap using a unique 5-year dataset of lake water quality (major ions, nutrients, and trace metals) to examine responses in water quality among lakes spanning a hydrological connectivity gradient to changes in connection time during flooding. Fortuitously, our study spans years that include recent historical highs and lows of peak discharge river height in the East Channel of the Mackenzie River, allowing us to collect observations of lake limnological responses to a range of flood conditions and lake connectivity to the river. We used water quality parameters collected during summers from 2013 to 2017 in combination with river level data to address the following questions: (1) Does water quality differ among lakes with contrasting flood regimes and connectivity to the surrounding landscape? (2) Does water quality change over the time period of the study in relation to variation in connection time? and (3) Do lakes with contrasting flood regimes respond similarly to alterations in flood regime over the course of the study? Finally, we interpret gradients in water quality in light of prior research and understanding of lake function to suggest implications for the carbon balance of Arctic deltaic lakes under conditions of climate-induced alterations to flood regimes.

**Methods**

**Study area**

The Mackenzie system is the largest Arctic drainage in North America, with a watershed area of $1.8 \times 10^6$ km$^2$ that encompasses a large portion of northwestern Canada (Rosenberg and Barton 1986). The north-flowing Mackenzie River mainstem terminates at Point Separation, where the river divides into anastomosing channels that run through the Mackenzie Delta into the Beaufort Sea. Marine discharge from the delta is rich in nutrients, suspended sediment, and dissolved organic and inorganic compounds (Holmes et al. 2002, Emmerton et al. 2008, Graydon et al. 2009, Tank et al. 2016).
Spring flooding with concurrent ice breakup is the major annual hydrological event in the Mackenzie Delta, typically commencing in mid-May and ending in early June (Marsh and Hey 1989). Nearly half of the discharge during this period is temporarily stored in the floodplain, where it mixes with lake water before ~80% of this stored water is exported eventually to the Beaufort Sea as the flood recedes (Emmerton et al. 2007). The remaining water is stored on the delta floodplain in the form of ~45 000 mainly small shallow lakes (Emmerton et al. 2007). These basins are formed by thermokarst processes in which standing water on top of permafrost increases the active layer depth, which leads to thaw subsidence of ice-rich permafrost and the formation of depressions that deepen into lake basins (Hill et al. 2001). The progressive deepening of lake basins from thermokarst activity combined with estimates of maximum ice cover thickness (0.6–1.2 m, depending on snow cover) suggest that the vast majority of the lakes are of sufficient depth to retain unfrozen water through the winter (Emmerton et al. 2007).

The physicochemical diversity of lakes in the Mackenzie Delta is primarily a result of differences in the length of connection of the lakes to the distributary channels during the flowing water season that lasts from May to September, and in particular the period of high water levels during ice breakup (Lesack and Marsh 2010). Mackay (1963) classified Mackenzie Delta lakes as No Closure (NC), Low Closure (LC), and High Closure (HC), defined by the frequency and duration of connections between the lake and the river. NC lakes possess connecting channels ranging from several metres to kilometres in length that allow flow between the lake and the channel. Flow in the connecting channels is typically into the lake during spring flooding and out of the lake during base flow periods in the summer but may briefly reverse during precipitation-driven increases in discharge (Hill et al. 2001). Over 80% of lakes in the Mackenzie Delta lack connecting channels and are perched to varying degrees above the river (Marsh and Hey 1989), only exchanging water during the spring flood period when river levels are higher than the minimum elevation of the sill of land between the lake and the channel. LC lakes have sill elevations lower than average peak river levels, so they are flooded annually during the spring high flow, but high enough to isolate them from the channels during summer base. HC lakes have sill elevations approaching or exceeding average peak flood levels and are inundated only for a period of days during the spring flood, or not at all in years with low peak river levels. HC lakes with the highest sill elevations may have a flood return period of up to 10 years and are subject to evaporative water loss in the absence of water renewal from flooding for consecutive years (Marsh and Lesack 1996). In the upper delta, NC lakes account for ~12%, LC 55%, and HC 33% of all lakes (Marsh and Hey 1989) and 23%, 51%, and 25%, respectively, of total lake area (Marsh et al. 1999).

Site selection and sample collection

Study lakes were located in the southeastern portion of the subaerial delta along the East Channel and adjacent Big Lake Channel near the town of Inuvik, NT (Fig. 1). Isolated lakes were located close to the main channels and were accessed by boat directly from the channel, whereas lakes connected to the river by distributary channels (ranging from ~ 0.1 to 2.1 km) were accessed via canoe. The lakes were selected to represent the 3 closure groups, with 6 HC and 2 LC sampled each year (2013–2017). In 2013, 3 NC lakes were sampled, but access via boat to these lakes was prevented in 2014 by low water levels in the distributary channels, so 2 more accessible NC lakes were selected for sampling in subsequent years (2015–2017). Lake connection time (CT) was determined using water level data from a hydrometric monitoring station (10LC002) operated by Environment and Climate Change Canada (ECCC; extracted from the ECCC Real-time Hydrometric Data web site [https://wateroffice.ec.gc.ca/mainmenu/real_time_data_index_e.html] on 30 May 2021) and historical lake sill elevation measurements (Marsh and Hey 1988). Water level in the East Channel of the Mackenzie Delta was provided by ECCC for 1985–1990 and 2002–2019, which we used to assess ranges in date and water stage at peak flow. CT was calculated as the number of days the river level exceeded the lake sill elevation (Supplemental Fig. S1).

Surface water was collected by hand in the near-shore zone of each lake in duplicate 2 L Nalgene bottles. Temperature, specific conductivity, pH, and dissolved oxygen (DO) were measured in situ using a YSI-600QS multiparameter probe (YSI, Yellow Springs, OH, USA). The samples were transported to the Western Arctic Research Centre, Aurora Research Institute, in Inuvik where they were processed within 12 h of sampling. All samples were first run through a 200 µm sieve to remove filamentous algae or floating debris. A subsample was passed through a pre-ashed 25 mm GF/C filter (pore size 1.2 µm) for analysis of particulate organic carbon (POC), particulate organic nitrogen (PON), and a 47 mm GF/C filter for chlorophyll a (Chl-a). The filtrate was retained for measurement of total dissolved phosphorus (TDP), soluble
reactive phosphorus (SRP), and dissolved Kjeldahl nitrogen (DKN), and unfiltered subsamples were collected for total phosphorus (TP) and total Kjeldahl nitrogen (TKN). Subsamples were also collected for analysis of DOC and DIC, major ions, ammonium (NH₄), nitrate/nitrite (NO₃/NO₂), and trace metals. Filters with samples were kept frozen until transported under refrigerated conditions to the National Laboratory for Environmental Testing, Burlington, ON, for analysis using standardized protocols (ECCC 2017a, 2017b). Water quality data for the East Channel were obtained from a public monitoring database (NWT-wide Community-based Water Quality Monitoring, Environment and Natural Resources, Government of the Northwest Territories, doi:10.25976/4der-gd31).

**Data analysis**

All water chemistry parameters were tested for normality (Shapiro–Wilk, α = 0.05; Royston 1995) and variables transformed as necessary to achieve normal distributions (Supplemental Table S1). We used linear regressions of CT against individual water chemistry variables to assess the relationship across years between the amount of time the lakes were flooded and the concentrations of ions, nutrients, and particulates, with p values adjusted for multiple testing using the method of Benjamini and Hochberg (1995). To establish water quality differences among lake closure types (HC, LC, or HC) we used a permutational multivariate analysis of variance of the Euclidean distances among samples (Anderson 2001) followed by pairwise permutational

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**Figure 1.** Study area, with base map data from Natural Resources Canada (https://atlas.gc.ca/toporama/en/index.html). Contains information licensed under the Open Government Licence – Canada.
multivariate analysis of variance (PERMANOVA) among closure types. We additionally assessed differences among lake closure types in water chemistry parameters using analysis of variance (ANOVA) and Tukey pair-wise comparisons (Tukey 1949) among closure types as well as the main channel for variables that differed significantly (adjusted \( p < 0.05 \)). These contrasts between closure types are useful for comparison to prior work and are included in the supplementary material (Supplemental Figs. S2 and S3).

To assess temporal changes in water quality parameters over the study period and the relationship of any changes to differences in CT resulting from variation in spring flood levels, we used principal component analysis (PCA) of the nutrient/ion and trace metal variables separately. The relationship of the resulting PCA axes to variation in CT was assessed by a posteriori projection of CT onto the ordination and using permutations of the environmental variables to test for significance using the \textit{envfit} function of the \textit{vegan} package in R (Oksanen et al. 2019).

**Results**

**Peak flood levels and lake connectivity during the study period**

Over the period of record, peak water levels ranged from 4.43 m a.s.l in 2016 to 6.13 m a.s.l in 2007 (Fig. 2). The date of the annual peak occurred within 2.5 weeks in the spring, the earliest peak occurring on 20 May 2016 and the latest on 7 June 1986. The 2013–2017 sampling period included years of high peak water level and late date of peak (2013), 2 years of low peak levels (2014 and 2016), and 2 years within one standard deviation of the average for level and date of peak (2015 and 2017).

CT differed markedly among the lake types based on differences in sill elevation (Table 1). HC lakes were flooded for up to 11 days but usually for a week or less, and the HC lakes with highest sill elevations were not connected for any period during the years with

![Figure 2](https://example.com/figure2.png)

**Figure 2.** Peak annual water level and the date it occurred measured in the East Channel of the Mackenzie Delta at Inuvik. The period of the current study is from 2013 to 2017. Lines and shaded bars indicate historical means and standard deviations for peak level (solid line) and date of peak (dashed line). Data from 2002 to 2012 are included for historical comparison, and the means and standard deviations were calculated using additional historical data from the same station (1985–1990, 2002–2017).
the lowest peak channel levels. LC lakes were inundated for 2 weeks to more than a month in the spring, and NC lakes were connected for approximately half the year on average.

**Does water quality differ among lakes with contrasting flood regimes and connectivity?**

Concentrations of 12 water chemistry parameters were significantly related to CT (adjusted $p < 0.05$) by linear regression (Fig. 3). The parameters included specific conductivity and all major ions except sodium (Na$^+$) and chloride (Cl$^-$), either positively (conductivity, sulfate [SO$_4^{2-}$], calcium [Ca$^{2+}$], potassium [K$^+$], silicon [Si]) or negatively (magnesium [Mg$^{2+}$], fluoride [F$^-$]). DO, pH, and total and dissolved organic nitrogen were significantly higher in lakes with shorter CTs. The total composition of ions and nutrients of lake water was significantly different in a PERMANOVA across the 3 closure types ($F = 16.327, R^2 = 0.443, df = 2, p < 0.001$), and significant differences ($p < 0.001$) were found in pairwise tests among all 3 lake closure types. The greatest differences were between the connected (NC) lakes and each of the isolated lake groups.

![Figure 3](image-url) **Figure 3.** Water chemistry parameters that vary significantly with connection time. Linear regression models ($p < 0.05$) are shown with 95% confidence intervals and adjusted $R^2$ in the corners of the plots. Units are in mg/L except for specific conductivity (Cond, µS/cm), temperature (Temp, °C), total Kjeldahl nitrogen (TKN, µg/L), dissolved organic nitrogen (DON, µg/L), and pH.
(NC vs. HC: $R^2 = 0.480$; NC vs. LC: $R^2 = 0.525$; LC vs. HC: $R^2 = 0.183$), indicating that water quality in the isolated (HC and LC) lakes was distinct from the NC lakes. Of the 24 water quality parameters, 9 were significantly different among closure type (Supplemental Table S1 and Figs S1, S2). Trace metal composition was significantly different in a multivariate ANOVA across the closure types ($F = 32.8$, $R^2 = 0.615$, $p < 0.001$), with pairwise contrasts significant between HC and the other 2 groups, and not significant between the LC and NC groups ($p = 0.055$). Only 6 metals were significantly different among closure groups (Supplemental Table S2 and Fig. S4). Fe and Sr were the dominant trace metals in the lakes, comprising 45.1% and 26.6% of total trace metals on average, followed by barium (Ba: 13.9%), aluminum (Al: 7.8%), manganese (Mn: 2.0%), and boron (B: 1.8%), with all other elements comprising <1% of the total ionic concentration (Supplemental Table S3). While missing variables in the channel data prevented direct comparison of percentages between the lakes and the channel, trace metals in the channel were dominated by iron (Fe) and Al. Concentrations of nearly all trace metals for which channel data were available were significantly higher in the channel than the lake (ANOVA, $p < 0.001$; Supplemental Table S3), the exceptions being Ba and strontium (Sr). Elements that were particularly enriched in the channel relative to the lakes were titanium (Ti; channel with 55× the mean lake concentration), Al (44×), caesium (Cs: 38×), beryllium (Be: 31×), vanadium (V: 17×), silver (Ag: 17×), and lead (Pb: 16×). Averaged across variables, concentrations of trace elements were ∼13.4× higher in the channel than the lakes, regardless of closure type.

**Discussion**

In this study we analysed a unique long-term dataset of surface water chemistry from a set of lakes along a gradient of hydrological connectivity in the Mackenzie River Delta during a period that encompassed extreme conditions in flood regime (as maximum river level height during spring peak discharge). We built on prior work to demonstrate how lake chemistry depends on connection to the river and to the landscape and how autochthonous processes mediate this relationship. We further demonstrated that the response of lakes to alterations in flood regime depends on their position on the connectivity gradient and suggest that these differences in response have implications for the delta under conditions of projected climate change.

We identified a gradient of CT and flood inputs from connected lakes to highly isolated thermokarst lakes. Lakes with distributary connections or long spring CTs receive suspended solids (POC and PON) from the river (Marsh et al. 1999), limiting light available for primary productivity (Squires and Lesack 2003a, 2003b). The river is also the likely source of SO$_4^{2−}$ in the more connected lakes (Lesack et al. 1998), especially given the high levels recorded in the main channel during the study period (Supplemental Fig. S5). We suggest that dissolved trace metals in the lakes originate from the river and are introduced by flooding or distributary connections for 2 reasons. First, riverine trace metals

Does water quality vary with connection time during the study period?

A PCA of nutrient/ion variables across all sites and years (Fig. 4a) produced 2 axes that cumulatively explained 41.9% of the variation among samples (PC1: 22.1%; PC2: 19.8%). These axes were significantly related to variation in CT (Table 1), both in an a posteriori projection onto the ordination ($R^2 = 0.688$, $p < 0.001$) and in individual linear regressions between each of the first 2 axes and CT (PC1: $R^2 = 0.287$, $p < 0.001$; PC2: $R^2 = 0.370$, $p < 0.001$). Among lakes, PC2 clearly reflected a sill elevation gradient from a cluster containing the NC lakes to a cluster containing high elevation lake L520, whereas PC1 reflected a gradient among the isolated lakes from lake MD3 (the lowest elevation LC lake) to HC lakes. Among years (Fig. 4b), the NC lakes showed little variation, whereas 2013 was distinct from the 2014–2017 samples for each of the LC and HC lakes. 2013 was notable in having the highest peak flood level and latest date of breakup during the study period (Fig. 2) and the longest CTs times in the isolated lakes (Table 1; details of the temporal changes in water chemistry parameters in the lakes summarized in Supplemental Fig. S5).

Variables with high loadings in the PCA (Fig. 4) were generally those with significant regressions against CT (Fig. 3), with the exception of DIC. Upon further examination, DIC was significantly positively related to CT in a linear regression among the isolated lakes, excluding the connected lakes (Fig. 5).

The first 2 axes of a PCA of the trace metals (Fig. 6) explained a cumulative 51.1% of the variation across all lakes and years, primarily along the first axis (PC1 39.4%; PC2 11.7%). Unlike the analysis of nutrients and ions, trace metals had no significant relationship with either CT time as an a posteriori projection ($R^2 = 0.11$, $p = 0.076$) or to the lake closure types. Instead, a strong distinction across years was noted, with 2013 having higher concentrations of nearly all trace metals, reflected in the positioning of samples and variables on PC1 (Fig. 6). The only exceptions to this trend were Ba, cadmium (Cd), and lithium (Li).
were enriched relative to lake water, often greatly so (Supplemental Table S3). Work from Siberian thermokarst lakes, reviewed in Colombo et al. (2018), suggested that while thermokarst processes release trace elements during periods of active thaw and subsidence, surface concentrations are primarily a factor of lake ontological development in which concentrations decrease as the lake deepens, and in mature lakes metals are sequestered in the sediments. Second, nearly every metal in all lakes, regardless of closure type, decreased between the high flood year of 2013 and subsequent years (Fig. 6), suggesting trace elements in the water column in late summer reflect flood conditions from the spring of the same year.

Increased loads of Mg2+ and F− were associated with lakes that flood infrequently (PC2, Fig. 4a). Evaporative concentration is a possible explanation because the annual water balance of closed lakes in the Mackenzie Delta is usually negative in the absence of spring flooding (Marsh and Bigras 1988). However, other conservative solutes such as Na+ and Cl− would likely be similarly enriched in the most isolated lakes, and we

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**Figure 4.** Principal component analysis (PCA) of nutrients and major ions of Mackenzie Delta lakes 2013–2017. (a) PCA scaling 2 showing relationships of sites to gradients of water chemistry parameters, with connection time (dashed arrow) projected onto the ordination. (b) PCA scaling 2 showing contrast between 2013 and later years for each isolated (low or high closure) lake; the arrows are drawn from the 2013 sample to the mean coordinate of the 2014–2017 samples for each lake, and years are indicated by shade of the symbols (darkest for 2013 and lightest for 2017).
found no relation of Na\(^+\) or Cl\(^-\) to Mg\(^{2+}\) or the closure gradient. Lesack et al. (1998) provided 2 other explanations for ionic enrichment in high closure lakes. Post-flood rillflow (water entering the lake indirectly as flood waters recede) is enriched in Mg\(^{2+}\), but this seems likely to affect all lakes equally, and the effect should therefore be a function of lake volume, which does not seem to be the case. Ionic concentrations may also increase because of solute exclusion during ice formation and incomplete flushing due to persistent ice cover during spring flooding. However, these processes also do not explain the discrepancies among the conservative ions. A final possible explanation is release of ions from thermokarst. Slumping of lake margins occurs as ice-rich permafrost soils degrade, exposing surface soils to the overlying water column and allowing release of sequestered ions into overlying water, including Ca\(^{2+}\), Mg\(^{2+}\), SO\(_4^{2-}\), and Si (Reyes and Lougheed 2015). The lake with the most pronounced thermokarst activity, L520, had elevated Mg\(^{2+}\) and Si relative to all other study lakes and elevated Ca\(^{2+}\) relative to other high closure lakes. Combined with the enrichment of Mg\(^{2+}\) in post-flood rillflow (Lesack et al. 1998), the catchment is likely the source of the elevated Mg\(^{2+}\) in high closure lakes, whether it originates in the permafrost or the active layer.

Elevated dissolved organic matter (DOM: DOC and DON) in some highly isolated lakes (particularly L520) is likely also due to thermokarst activity. However, our DOM measurements must be treated with caution because we measured bulk DOM. While DOM generally increases with sill elevation in Mackenzie Delta lakes, this DOM has diverse origins and characteristics (Tank et al. 2011). While most DOM in connected lakes originates from the river, lower turbidity in closed lakes permits high productivity of macrophytes that release colourless, highly labile DOM (Squires et al. 2009, Tank et al. 2011), which is rapidly utilized by bacterial communities (Spears and Lesack 2006, Cunada

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**Figure 5.** Dissolved inorganic carbon (DIC) and connection time among lakes that become isolated during the summer. Linear regression (\(p = 0.005\), adjusted \(R^2 = 0.193\)) shown with 95% confidence intervals. The connected lakes had DIC concentrations \(\sim 20\) mg/L.

**Figure 6.** Principal component analysis (PCA; scaling 2) of trace metals in Mackenzie Delta lakes from 2013 to 2017. Variable labels excluded for clarity. Numerals are the last digit of the year the sample was taken (3 = 2013, 4 = 2014, 5 = 2015, 6 = 2016, 7 = 2017).
et al. 2018). In thermokarst lakes, coloured DOM originating in the permafrost and/or from shoreline slumping accumulates because it is relatively recalcitrant to bacterial metabolism (Squires and Lesack 2003b, Tank et al. 2011, Cunada et al. 2018). So, while prior knowledge of L520 suggests thermokarst as a source of DOM, much of the variability in DOM in other isolated lakes is potentially allochthonous DOM originating from macrophytes. We also note evaporative concentration as a possible cause of elevated DOC in higher elevation lakes.

The PC2 axis therefore represents a gradient of sources of chemical constituents from riverine inputs (POC, PON, SO4−2) to those derived from the terrestrial landscape including permafrost (Mg2+, DOM). Within lakes, variability among years was related to closure type and spring flood conditions, with 2013 having greater peak flood level (Fig. 2) and longer CTs (Table 1) than subsequent years. No systematic variation was observed in the main channel over the study period, so little interannual variation was found in NC lakes, which are strongly influenced by riverine inputs. Every highly isolated lake shifted on PC2 between 2013 and subsequent years, becoming more distinct from the connected lakes when peak flood levels were at or below historical average. The active thermokarst lake (L520) is notably distinct from the other isolated lakes in all years, so whether the shift along PC2 represented the development or increase (in L520) of thermokarst activity as opposed to shifting to alternate stable states based on flood conditions is unclear. In either case, our results demonstrate a limnological response to large changes in flood conditions based on physical and chemical processes.

A second limnological gradient (PC1, Fig. 4a) operating within the isolated lakes involved a set of variables suggesting this axis represents a gradient of summer macrophyte productivity. Prior work (Squires et al. 2009) has demonstrated turbidity and depth limitation of submerged macrophytes (primarily Potamogeton spp.) in Mackenzie Delta lakes, which form dense communities that cover the entire lakebed in clear, shallow, closed lakes. In the most isolated lakes (e.g., L520), increased water transparency and sediment fertility allows a shift to dominance by Chara and Ceratophyllum (Squires and Lesack 2003b). Uptake of bicarbonate (HCO3−; represented here as DIC), particularly by Potamogeton, raises the pH (Fig. 3) causing precipitation of calcium carbonate (CaCO3). Concentrations of some ions (K+, Si) are also likely limited by macrophyte growth during summer (Lesack et al. 1998). In our PCA analysis this gradient was correlated to CT, from the lowest elevation closed lake (MD3) to high sill lakes with transparent water and dense Potamogeton communities (L275, MD2). Depth and physical factors in L520 favour a macrophyte community distinct from other HC lakes, and the DIC drawdown associated with Potamogeton-dominated lakes did not occur.

We observed that the 2 kinds of isolated lakes responded differently to alterations in flood regime, particularly contrasting the high-flood year 2013 to subsequent years. Whereas HC lakes did not vary among years on PC1, each LC lake shifted in the positive direction between 2013 and subsequent years (Fig. 4b), indicating greater rooted macrophyte production in years of low or average CT. Differences in CT between 2013 and subsequent years were only ∼5–10 days for HC lakes (Table 1), presumably not enough for variation to affect production of rooted macrophytes in late summer. LC lakes were flooded for 4–7 weeks in 2013, compared to 2–3 weeks in subsequent years, with potential mixing with turbid river water occurring into July 2013. The longer turbid period could delay the onset of macrophyte recruitment, leading to lower productivity in late summer. No such changes occurred in the connected lakes, which mix with turbid river water throughout the growing season and have low macrophyte productivity compared to closed lakes (Squires and Lesack 2003b). Lake metabolism as determined by primary productivity seems to uniquely respond to changes in CT in those lakes for which flood duration and related turbidity is a limiting factor for macrophyte growth.

A dramatic example of rapid change occurred in lake MD4 (Supplemental Fig. S6), although it is unclear whether erosion of near-surface ground ice beneath the lake or failure of the sill was responsible for the water level decline. Low post-drainage concentrations of Mg2+, Na+, and Cl− suggest after 2014 the lake was replenished by dilute sources such as snowmelt (Lesack et al. 1998), and that the increase in particulates and trace metals was not caused by evaporation but rather likely caused by wind-driven resuspension in the shallow basin. The rapid drainage and ultimate disappearance of lakes over permafrost has been observed across the Arctic (Smith et al. 2005), although evidence is conflicting as to whether this phenomenon is increasing (Lantz 2017).

Alterations to the primary productivity of Mackenzie Delta lakes may have implications for the carbon budget of the overall delta. Tank et al. (2009) demonstrated that isolated, non-thermokarst lakes with high macrophyte production were net absorbers of CO2 over the ice-free season, in contrast to thermokarst and connected lakes that emitted CO2 on balance on an annual basis. Because LC lakes account for 51% of total lake area in
the Mackenzie Delta (Marsh et al. 1999), changes in macrophyte-mediated CO$_2$ absorption in this lake type would potentially impact the overall CO$_2$ budget of the delta. However, considerable uncertainty exists in the direction such a change would follow. High elevation lakes in the Mackenzie Delta are becoming less connected to the river as ice-jam flooding declines and the onset of breakup occurs earlier (Lesack et al. 2014), while CTs of low elevation lakes are increasing because of rising baseflow levels and increased storm surges (Lesack and Marsh 2007). The response of LC lakes to climate-mediated changes in hydrology likely depends on their elevation; lakes with sills that remain above their predicted higher river levels could become more productive and therefore stronger CO$_2$ absorbers while those that become inundated through more of the ice-free season would have lower macrophyte productivity and become CO$_2$ emitters. Further research to confirm the emission status of low closure lakes using pCO$_2$ measurements and to model the effect of climate-mediated hydrological changes on low-sill, isolated lakes would greatly refine our understanding of the CO$_2$ balance of the Mackenzie Delta and the potential consequences of alterations due to climate change.

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References

Abbott BW, Jones JB, Schuur EAG, Chapin FS III, Bowden WB, Bret-Harte MS, Epstein HE, Flannigan MD, Harms TK, Hollingsworth TN, et al. 2016. Biomass offsets little or none of permafrost carbon release from soils, streams, and wildfire: an expert assessment. Environ Res Lett. 11(3):034014.

Abdul Aziz OI, Burn DH. 2006. Trends and variability in the hydrological regime of the Mackenzie River basin. J Hydrol. 319:282–294.

Anderson MJ. 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecol. 26:32–46.

Bastviken D, Tranvik LJ, Downing JA, Crill PM, Enrich-Prast A. 2011. Freshwater methane emissions offset the continental carbon sink. Science. 331(6013):50.

Beltaos S. 2013. Hydrodynamic and climatic drivers of ice breakup in the lower Mackenzie River. Cold Reg Sci Technol. 95:39–52.

Beltaos S, Carter T, Rowsell R. 2012. Measurements and analysis of ice breakup and jamming characteristics in the Mackenzie Delta, Canada. Cold Reg Sci Technol. 82:110–123.

Beltaos S, Prowse T. 2009. River-ice hydrology in a shrinking cryosphere. Hydrol Process. 23(1):122–144.

Benjamini Y, Hochberg Y. 1995. Controlling the false discovery rate: a practical and powerful approach to multiple testing. J R Stat Soc B Met. 57(1):289–300.

Bogard MJ, Kuhn CD, Johnston SE, Striegl RG, Holtgrieve GW, Dornblaser MM, Spencer RGM, Wickland KP, Butman DE. 2019. Negligible cycling of terrestrial carbon in many lakes of the arid circumpolar landscape. Nat Geosci. 12(3):180–185.

Brown NJ, Nilsson J, Pemberton P. 2019. Arctic ocean freshwater dynamics: transient response to increasing river runoff and precipitation. J Geophys Res-Oceans. 124:5205–5219.

Carmack E, Macdonald R. 2002. Oceanography of the Canadian shelf of the Beaufort Sea: a setting for marine life. Arctic. 55(Suppl 1):29–45.

Cole JJ, Prairie YT, Caraco NF, McDowell WH, Tranvik LJ, Striegl RG, Duarte CM, Kortelainen P, Downing JA, Middelburg JJ, Melack J. 2007. Plumbing the global carbon cycle: integrating inland waters into the terrestrial carbon budget. Ecosystems. 10(1):172–185.

Colombo N, Salerno F, Gruber S, Freppaz M, Williams M, Fratianni S, Giardino M. 2018. Review: impacts of permafrost degradation on inorganic chemistry of surface fresh water. Global Planet Change. 162:69–83.

Cunada CL, Lesack LFW, Tank SE. 2018. Seasonal dynamics of dissolved methane in lakes of the Mackenzie Delta and the role of carbon substrate quality. J Geophys Res-Biogeo. 123(2):591–609.

Drake TW, Wickland KP, Spencer RGM, McKnight DM, Striegl RG. 2015. Ancient low–molecular-weight organic acids in permafrost fuel rapid carbon dioxide production upon thaw. P Natl Acad Sci USA. 112(45):13946–13951.

Emmerton CA, Lesack LFW, Marsh P. 2007. Lake abundance, potential water storage, and habitat distribution in the
Mackenzie River Delta, western Canadian Arctic. Water Resour Res. 43(5):W05419.

Emmerton CA, Lesack LFW, Vincent WF. 2008. Mackenzie River nutrient delivery to the Arctic Ocean and effects of the Mackenzie Delta during open water conditions. Global Biogeochem Cy. 22(1):GB1024.

Emmerton CA, Louis VL, Lehnherr I, Graydon JA, Kirk JL, Rondeau KJ. 2016. The importance of freshwater systems to the net atmospheric exchange of carbon dioxide and methane with a rapidly changing high Arctic watershed. Biogeosciences. 13(20):5849–5863.

[ECCC] Environment and Climate Change Canada. 2017a. Standard operating procedure for the analysis of carbon and nitrogen in suspended particulate matter and sediment in natural waters via combustion procedure. Burlington (ON): National Laboratory for Environmental Testing. SOP 01-1090.

[ECCC] Environment and Climate Change Canada. 2017b. Standard operating procedure for the analysis of total and dissolved trace metals in water by ‘in bottle digestion’ and inductively coupled plasma-quadrupole mass spectrometry with collision/reaction cell capability (CRC-ICP-QMS). Burlington (ON): National Laboratory for Environmental Testing.

Febria CM, Lesack LFW, Gareis JAL, Bothwell ML. 2006. Patterns of hydrogen peroxide among lakes of the Mackenzie Delta, western Canadian Arctic. Can J Fish Aquat Sci. 63(9):2107–2118.

Feng X, Vonk JE, van Dongen BE, Gustafsson O, Semiletov IP, Dudarev OV, Wang Z, Montlucon DB, Wacker L, Eglinton TG. 2013. Differential mobilization of terrestrial carbon pools in Eurasian Arctic river basins. P Natl Acad Sci USA. 110(35):14168–14173.

Frey KE, McClelland JW. 2009. Impacts of permafrost degradation on Arctic river biogeochemistry. Hydrol Process. 23(1):169–182.

Goulding HL, Prowse TD, Beltaos S. 2009. Spatial and temporal patterns of break-up and ice-jam flooding in the Mackenzie Delta, NWT. Hydrol Process. 23(18):2654–2670.

Graydon JA, Emmerton CA, Lesack LFW, Kelly EN. 2009. Mercury in the Mackenzie River Delta and estuary: concentrations and fluxes during open-water conditions. Sci Total Environ. 407(8):2980–2988.

Haine TWN, Curry B, Gerdes R, Hansen E, Karcher M, Lee C, Rudels B, Sreen G, de Steur L, Stewart KD, Woodgate R. 2015. Arctic freshwater export: status, mechanisms, and prospects. Global Planet Change. 125:13–35.

Hill PR, Lewis CP, Desmarais S, Kauppamuthoo V, Raas H. 2001. The Mackenzie Delta: sedimentary processes and facies of a high-latitude, fine-grained delta. Sedimentology. 48(5):1047–1078.

Holmes RM, McClelland JW, Peterson BJ, Shiklomanov IA, Shiklomanov AI, Zhulidov AV, Gordeev VV, Bobroviatskaya NN. 2002. A circumpolar perspective on fluvial sediment flux to the Arctic Ocean. Global Biogeochem Cy. 16(4):1098.

Holmes RM, McClelland JW, Peterson BJ, Tank SE, Bulygina E, Eglinton TI, Gordeev VV, Gurtovaya TY, Raymond PA, Repeta DJ, et al. 2012. Seasonal and annual fluxes of nutrients and organic matter from large rivers to the Arctic Ocean and surrounding seas. Estuar Coast. 35(2):369–382.

Jammol M, Dengel S, Kettner E, Parmentier FJW, Wik M, Crill P, Friborg T. 2017. Year-round CH4 and CO2 flux dynamics in two contrasting freshwater ecosystems of the subarctic. Biogeosciences. 14(22):5189–5216.

Junk WJ, Bayley PB, Sparks RE. 1989. The flood pulse concept in river–floodplain systems. In: Dodge DP, editor. Proceedings of the International Large River Symposium. Can Spec Publ Fish Aquat Sci. 106:110–127.

Kokelj SV, Lantz TC, Tunnicliffe J, Segal R, Lacelle D. 2017. Climate-driven thaw of permafrost preserved glacial landscapes, northwestern Canada. Geology. 45(4):371–374.

Lantz TC. 2017. Vegetation succession and environmental conditions following catastrophic lake drainage in Old Crow Flats, Yukon. Arctic. 70(2):177–189.

Lehner B, Döll P. 2004. Development and validation of a global database of lakes, reservoirs and wetlands. J Hydrol. 296:1–22.

Lesack LFW, Marsh P, Hecky RE. 1998. Spatial and temporal dynamics of major solute chemistry among Mackenzie Delta lakes. Limnol Oceanogr. 43(7):1530–1543.

Lesack LFW, Marsh P. 2007. Lengthening plus shortening of river-to-lake connection times in the Mackenzie River Delta respectively via two global change mechanisms along the Arctic coast. Geophys Res Lett. 34(23):L23404.

Lesack LFW, Marsh P. 2010. River-to-lake connectivities, water renewal, and aquatic habitat diversity in the Mackenzie River Delta. Water Resour Res. 46(12):W12504.

Lesack LFW, Marsh P, Hicks FE, Forbes DL. 2013. Timing, duration, and magnitude of peak annual water-levels during ice breakup in the Mackenzie Delta and the role of river discharge. Water Resour Res. 49(12):8234–8249.

Lesack LFW, Marsh P, Hicks FE, Forbes DL. 2014. Local spring warming drives earlier river-ice breakup in a large Arctic delta. Geophys Res Lett. 41(5):1560–1567.

Mackay JR. 1963. The Mackenzie Delta area, N.W.T. Ottawa (ON): Geological Survey of Canada. Misc Report 23.

Marsh P, Bigras SC. 1988. Evaporation from Mackenzie Delta Lakes, N.W.T., Canada. Arctic Alpine Res. 20(2):220–229.

Marsh P, Hey M. 1988. Mackenzie River water levels and the flooding of delta lakes. Saskatoon (SK): National Hydrology Research Institute, Environment Canada. Contribution 88013.

Marsh P, Hey M. 1989. The flooding hydrology of Mackenzie Delta lakes near Inuvik, N.W.T., Canada. Arctic. 42:41–49.

Marsh P, Lesack LFW. 1996. The hydrologic regime of perched lakes in the Mackenzie Delta: potential responses to climate change. Limnol Oceanogr. 41:849–856.

Marsh P, Lesack LFW, Roberts A. 1999. Lake sedimentation in the Mackenzie Delta, NWT. Hydrol Process. 13(16):2519–2536.

Oksanen J, Blanchet FG, Friendly M, Kindt R, Legendre P, McGlinn D, Minchin PR, O’Hara RB, Simpson GL, Solymos P, et al. 2019. vegan: Community Ecology package. R package version 2.5-4. https://CRAN.R-project.org/package=vegan

Peterson BJ, Holmes RM, McClelland JW, Vörösmarty CJ, Lammers RB, Shiklomanov AI, Shiklomanov IA, Rahmstorf S. 2002. Increasing river discharge to the Arctic Ocean. Science. 298(5601):2171–2173.

Plaza C, Pegoraro E, Bracho R, Celis G, Crummer KG, Hutchings JA, Hicks Pries CE, Mauritz M, Natali SM, Salmon et al. 2019. Direct observation of permafrost

Rahmstorf S. 2002. Increasing river discharge to the Mackenzie Delta: potential responses to climate change. Limnol Oceanogr. 41:849–856.
degradation and rapid soil carbon loss in tundra. Nat Geosci. 12(8):627–631.

Prowse TD, Culp JM. 2003. Ice breakup: a neglected factor in river ecology. Can J Civ Eng. 30(1):128–144.

Reyes FR, Lougheed VL. 2015. Rapid nutrient release from permafrost thaw in Arctic aquatic ecosystems. Arct Antarct Alp Res. 47(1):35–48.

Rood SB, Kaluhto SA, Philipse DJ, Rood NJ, Zanewich KP. 2017. Increasing discharge from the Mackenzie River system to the Arctic Ocean. Hydrol Process. 31(1):150–160.

Rosenberg DM, Barton DR. 1986. The Mackenzie river system. In: Davies BR, Walker KF, editors. The ecology of river systems. The Netherlands: Dr W. Junk Publishers; p. 425–433.

Rouse WR, Douglas MSV, Hecky RE, Hershey AE, Kling GW, Lesack LFW, Marsh P, Mcdonald M, Nicholson BJ, Roulet NT, Smol JP. 1997. Effects of climate change on the freshwaters of Arctic and subarctic North America. Hydrol Process. 11(8):873–902.

Royston P. 1995. Remark AS R94: a remark on algorithm AS 181: the W-test for normality. Appl Stat. 44(4):547.

Schuur EAG, McGuire AD, Schädel C, Grosse G, Harden JW, Hayes DJ, Hugelius G, Koven CD, Kryp P, Lawrence DM, et al. 2015. Climate change and the permafrost carbon feedback. Nature. 520(7546):171–179.

Segal RA, Lantz TC, Kokelj SV. 2016. Acceleration of thaw slump activity in glaciated landscapes of the western Canadian Arctic. Environ Res Lett. 11(5):054015.

Smith LC, Sheng Y, MacDonald GM, Hinzman LD. 2005. Disappearing Arctic lakes. Science. 308(5727):1439–1442.

Spears BM, Lesack LF. 2006. Bacterioplankton production, abundance, and nutrient limitation among lakes of the Mackenzie Delta (Western Canadian Arctic). Can J Fish Aquat Sci. 63(4):485–487.

Squires MM, Lesack LF. 2002. Water transparency and nutrients as controls on phytoplankton along a flood-frequency gradient among lakes of the Mackenzie Delta, Western Canadian Arctic. Can J Fish Aquat Sci. 59(8):1339–1349.

Squires MM, Lesack LF, Huebert D. 2002. The influence of water transparency on the distribution and abundance of macrophytes among lakes of the Mackenzie Delta, Western Canadian Arctic. Freshwater Biol. 47(11):2123–2135.

Squires MM, Lesack LF. 2003a. Spatial and temporal patterns of light attenuation among lakes of the Mackenzie Delta. Freshwater Biol. 48(1):1–20.

Squires MM, Lesack LF. 2003b. The relation between sediment nutrient content and macrophyte biomass and community structure along a water transparency gradient among lakes of the Mackenzie Delta. Can J Fish Aquat Sci. 60(3):333–343.

Squires MM, Lesack LF, Hecky RE, Guildford SJ, Ramlal P, Higgins SN. 2009. Primary production and carbon dioxide metabolic balance of a lake-rich Arctic river floodplain: partitioning of phytoplankton, epiplankton, macrophyte, and epiphyton production among lakes on the Mackenzie Delta. Ecosystems. 12(5):853–872.

Tank SE, Fellman JB, Hood E, Kritzberg ES. 2018. Beyond respiration: controls on lateral carbon fluxes across the terrestrial–aquatic interface: controls on lateral carbon fluxes. Limnol Oceanogr Lett. 3(3):76–88.

Tank SE, Lesack LF, Hesselin RH. 2009. Northern delta lakes as summertime CO2 absorbers within the Arctic landscape. Ecosystems. 12(1):144–157.

Tank SE, Lesack LF, Gareis JAL, Osburn CL, Hesselin RH. 2011. Multiple tracers demonstrate distinct sources of dissolved organic matter to lakes of the Mackenzie Delta, Western Canadian Arctic. Limnol Oceanogr. 56(4):1297–1309.

Tank SE, Raymond PA, Striegl RG, McClelland JW, Holmes RM, Fiske GJ, Peterson BJ. 2012. A land-to-ocean perspective on the magnitude, source and implication of DIC flux from major Arctic rivers to the Arctic Ocean. Global Biogeochem Cy. 26(4):GB4018.

Tank SE, Striegl RG, McClelland JW, Kokelj SV. 2016. Multidecadal increases in dissolved organic carbon and alkalinity flux from the Mackenzie drainage basin to the Arctic Ocean. Environ Res Lett. 11(5):054015.

Tarnocai C, Canadell JG, Schuur EAG, Mazhitova G, Zimov S. 2009. Soil organic carbon pools in the northern circumpolar permafrost region. Global Biogeochem Cy. 23(2):GB2023.

Tockner K, Malard F, Ward JV. 2000. An extension of the flood pulse concept. Hydrol Process. 14:2861–2883.

Tukey JW. 1949. Comparing individual means in the analysis of variance. Biometrics. 5(2):99–114.

Vonk JE, Mann PJ, Davydov S, Davydkova A, Spencer RGM, Schade J, Sobczak WV, Zimov N, Zimov S, Bulygina E, et al. 2013. High biolability of ancient permafrost carbon upon thaw. Geophys Res Lett. 40(11):2689–2693.

Vonk JE, Tank SE, Bowden WB, Laurion I, Vincent WF, Alekseychik P, Amyot M, Billet MF, Canário J, Cory RM, et al. 2015a. Reviews and syntheses: effects of permafrost thaw on Arctic aquatic ecosystems. Biogeosciences. 12(23):7129–7167.

Vonk JE, Tank SE, Mann PJ, Spencer RGM, Treat CC, Striegl RG, Abbott BW, Wickland KP. 2015b. Biodegradability of dissolved organic carbon in permafrost soils and aquatic systems: a meta-analysis. Biogeosciences. 12(23):6915–6930.

Walvoord MA, Kurylyk BL. 2016. Hydrologic impacts of thawing permafrost—a review. Vadose Zone J. 15(6):1–20.

Wang S, Zhou F, Russell H. 2017. Estimating snow mass and peak river flows for the Mackenzie River basin using GRACE satellite observations. Remote Sens. 9(3):256.

Wild B, Andresson A, Bröder L, Vonk JE, Hugelius G, McClelland JW, Song W, Raymond PA, Gustafsson Ö. 2019. Rivers across the Siberian Arctic unearth the patterns of carbon release from thawing permafrost. P Natl Acad Sci USA. 116(21):10280–10285.

Woo MK, Thorne R. 2003. Streamflow in the Mackenzie Basin, Canada. Arctic. 56(4):328–340.

Yang D, Shi X, Marsh P. 2015. Variability and extreme of Mackenzie River daily discharge during 1973–2011. Quat Int. 380–381:159–168.