A PALEOLIMNOLOGICAL INVESTIGATION INTO THE ECOLOGICAL IMPACTS OF PERMAFROST THAW SLUMPING ON CLADOCERANS (BRANCHIOPODA, CRUSTACEA) IN THE MACKENZIE DELTA REGION (NORTHWEST TERRITORIES, CANADA)

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ABSTRACT

This thesis explores the ecological impacts of retrogressive thaw slumping on cladocerans (Branchiopoda, Crustacea) in Mackenzie Delta Region (Northwest Territories, Canada) lakes using paleolimnological techniques to answer the following questions: 1) Does thaw slumping increase UV exposure of *Chydorus*? 2) Are *Daphnia* absent in slump-affected lakes? I reconstructed UV exposure of *Chydorus* in a reference and slump-affected lake by measuring carapace absorbances at UV wavelengths. Carapace absorbances in the slump-affected lake were comparable to the reference lake, indicating *Chydorus* were not exposed to more UV radiation than in the reference lake. I also examined *Daphnia* ephippia in surface and pre-1850 sediments from 7 slump-affected and 7 unimpacted lakes, to confirm a 2017 zooplankton survey observation that *Daphnia* were absent in slump-affected lakes. *Daphnia* were not observed in slump-impacted lake sediments; however low sediment volumes likely underestimate *Daphnia* presence. My research provides new information on zooplankton biogeography in the Mackenzie Delta region.

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CHAPTER 1: GENERAL INTRODUCTION

Shoreline retrogressive thaw slumps (SRTS) are an example of abrupt permafrost thaw (termed thermokarst) that form landslide-like U-shaped scars on coastal, river and lake shorelines (Lantz & Kokelj, 2008). Retrogressive thaw slumps consist of a retreating headwall that exposes ground ice and a mud slurry slump floor (Lantz & Kokelj, 2008). They have been increasing in intensity and frequency in the western Arctic region as a result of climate warming (Lantz & Kokelj, 2008; Kokelj et al., 2017). There are knowledge gaps regarding the implications of thaw slumps on lake ecological communities, including zooplankton communities that are commonly used as ecological indicators (Jeppesen et al., 2011). The water quality changes (i.e. increased conductivity and decreased nutrient concentrations) that result from shoreline thaw slumping are variables known to structure zooplankton communities in Arctic lakes. Zooplankton are important for the overall health of lake ecosystems, and consequently shift in zooplankton communities that may occur as a result of thaw slumping have the potential to cascade through Arctic lake food webs.

Zooplankton responses to permafrost thaw and retrogressive thaw slumping is poorly understood, in part due to a lack of information on zooplankton biogeography in Arctic regions. This thesis is comprised of two related paleolimnological studies (Chapters 3 and 4) that investigate the impacts of slumping on zooplankton in lakes of the Mackenzie Delta region (Northwest Territories, Canada). The Mackenzie Delta region is lake-rich and is underlain by thick (>100 m) ice-rich continuous permafrost (Burn & Kokelj, 2009). The study region spans the transition from the boreal forest to the low-shrub tundra (Lantz et al., 2010), and thermokarst activity is well-documented in the area (Lantz & Kokelj, 2008; Kokelj et al., 2017).

In Chapter 3, I reconstructed *Chydorus* (a genus of littoral cladocerans) exposure to potentially harmful UV radiation in a pair of neighboring lakes of similar size, where one lake (Lake 9B) has a shoreline thaw slump, and the neighbouring lake (Lake 9A) is a reference lake with no history of thaw slumping. Slump-affected lakes exhibit higher water clarity due to the input of inorganic sediment into the lake from the thaw slump binding to and scouring dissolved organic carbon from the water column, which is then deposited to the lake bottom sediments (Thompson et al., 2008; Houben et al., 2016). Increased exposure to potentially harmful UV radiation in the water column as a result of increased water clarity caused by thaw slumping can be a potential stressor on lake biota. Zooplankton and benthos exposed to harmful UV radiation can experience reduced productivity, growth and development, and increased mutation rates (Häder et al. 2007). Fortunately, zooplankton and benthos produce protective pigments (i.e. melanin) that mitigate the harmful effects of UV radiation (Rautio et al., 2009). Cladocerans (Cladocera; a group of small crustaceans) leave well-preserved subfossil remains (i.e. carapaces, headshields, and post-abdominal claw) in lake sediments that can be used as biological proxies to infer changes in underwater exposure to UV radiation over decades to centuries to millennia. I used melanin absorbance measurements from *Chydorus* carapaces to infer temporal changes in exposure to UV radiation over the last ~100 years.

In Chapter 4, I investigated *Daphnia* presence/absence in 14 lakes in the Mackenzie Delta region along a gradient of thaw slump impacts. This chapter builds upon my Honour's Thesis research from 2017 that compared zooplankton community composition between slump-affected and undisturbed lakes in the Mackenzie Delta region based on a July survey of zooplankton in surface waters. I found that *Daphnia* were mostly absent in slump-affected lakes. *Daphnia* are often considered to be a keystone species, and if they are lost in lakes as a result of thaw slumping this may cascade through the Arctic lake food web. Contemporary zooplankton lake based on one sampling point can miss *Daphnia* as a result of seasonality or vertical migration during the daylight hours to avoid fish predation, potentially resulting in an underrepresentation of their prevalence in a lake (Korhola & Rautio, 2001). Paleolimnological techniques can be used to supplement one-time zooplankton net tows because lake sediments incorporate cladoceran subfossils from all parts of the lake (i.e. littoral and limnetic zones) and several years of sediment accumulation (Korhola & Rautio, 2001). To investigate *Daphnia* presence, I used a 'top-bottom' approach that uses the 'top' interval that is representative of modern conditions and 'bottom' interval that is representative of pre-industrial conditions (~150 years ago). Where encountered, the subfossil remains of *Chaoborus* (Chaoboridae; phantom midges) were also noted, given their role as a predator on *Daphnia* and also as an indicator of fish presence/absence (fish predation is a key control on *Daphnia* presence/absence in shallow lakes; Brooks & Dodson, 1965).

1.1 Thesis objectives

The aim of this study was to advance our knowledge on the impacts of retrogressive thaw slumping on zooplankton in lakes of the Mackenzie Delta uplands region. My objectives are as follows:

1) Reconstruct the recent (last ~300 years) history of *Chydorus* exposure to UV radiation in the water column of two lakes in the Mackenzie Delta uplands and assess whether *Chydorus* in slump-impacted lakes are vulnerable to potentially harmful UV radiation.

2) Assess *Daphnia* presence/absence in surface and pre-industrial sediments in relation to permafrost thaw slumping in the same set of lakes surveyed for zooplankton in surface waters in July, 2017.

CHAPTER 2: LITERATURE REVIEW

2.1 Limnology of Arctic lakes

The Arctic is characterized by numerous lakes, ponds, wetlands, rivers, and streams. For example, approximately 600,000 lakes cover >10% of the Lena River Delta in the North Siberian lake region; lakes occupy 5-10% of the Western Siberian tundra zone and Yana-Indigirka lowland regions; and Karelia (located in Finland and Russia) is comprised of approximately 61,000 lakes covering 21% of its area (Filatov & Gronskaya, 2012). In some Arctic regions, approximately 90% of the total surface area is covered lakes (Pienitz et al., 2008), including the Mackenzie Delta region of the Northwest Territories in Canada that has approximately 45,000 lakes in an area of 13,000 km^2 (Emmerton et al., 2007), and the Yukon Delta that has approximately 200,000 lakes in an area of 80,000 km^2 (Maciolek, 1989). Lake density is higher in previously glaciated terrain compared to non-glaciated terrain, and lake density is also higher in permafrost terrain due to poor drainage of permafrost soils (Pienitz et al., 2008).

Lake thermal regime is a fundamental physical property of lakes that influences ecological and biogeochemical function (Smol et al., 2005; Vincent et al., 2013; Vonk et al., 2015). Thermal stratification is a well-known phenomenon characterized by a change in water temperature with depth that develops as a result of the temperature-density relationship of water (water is most dense at 3.98 °C), typically pronounced in the summer and winter seasons (Woo, 2012; Vonk et al., 2015). In lakes that thermally stratify in the summer, the surface water increases in temperature as a result of heat inputs (i.e. solar radiation). As the surface waters warm, they become less dense than the underlying colder water, eventually thermally stratifying as the density difference presents a barrier to water mixing. In the winter, the surface water cools below 3.98 °C and inverse stratification can occur, where the waters below the ice are a few degrees colder (and less dense) than the underlying hypolimnion (Woo, 2012). During the spring and fall, lakes become isothermal, and the water column mixes (spring and fall overturn). Many Arctic lakes are well-mixed throughout the summer or are thermally stratified for only a short duration of several days (Vonk et al., 2015); however, lakes exhibiting strong thermal stratification have been reported across the circumpolar Arctic and subarctic region (Laurion et al., 2010; Boike et al., 2015; Bouchard et al., 2015; Deshpande et al., 2015; Sepulveda-Jauregui et al., 2015). The frequency and intensity of thermal stratification in Arctic lakes has been increasing in response to climate warming, a trend that is projected to continue into the future (Vonk et al., 2015).

Arctic lakes can be ice-covered for upwards of 8 months of the year or more (Woo, 2012). Lake ice formation initiates when heat loss exceeds heat gains from solar radiation inputs and convection in the lake (Woo, 2012). Ice particles are produced which aggregate into flocs and slush, eventually forming coherent lake ice (Daly, 1984; Woo, 2012). The incorporation of snow into the ice cover along with frazil ice results in the formation of white ice (Woo, 2012). Under calm conditions, lake shorelines often begin to form an ice sheet that is transparent and absent of air bubbles; this is known as black ice (Woo, 2012). Lake ice decay proceeds through both thermal and mechanical processes. Thermal processes include ice melt at the edges of the ice cover, ice-air interface, ice-water interface (bottom of the ice), and internal melt within the ice cover (Woo, 2012). The ice melt at the ice-air interface is determined by net long-wave radiation and heat fluxes (sensible heat, latent heat, and heat from precipitation) (Woo, 2012). Internal ice melt is mainly driven by the penetration of short-wave radiation, where the ice becomes increasingly porous (Woo, 2012). The ice-water interface is driven by a temperature gradient, particularly by the thermal conductivity and temperature of the lake water (Woo, 2012).

Once melt of the ice cover progresses to the formation of floating ice, mechanical breakup simultaneously contributes to ice decay, primarily driven by surface winds (Woo, 2012). The processes of lake ice formation, growth, and decay influences biogeochemical cycling of elements, lake thermal regime, and rates of primary and secondary production during the open-water season (Rautio et al., 2011).

Arctic lakes are often oligotrophic, meaning they are low in nutrients and have low rates of primary production (Lizotte, 2008). There are several factors that influence primary production in Arctic lakes, including nutrient availability, ice phenology, photosynthetic active radiation (PAR) and UV radiation, and grazing pressure by zooplankton (Rautio et al., 2011). Nutrient availability is thought to be the main limiting factor in primary production (Schindler, 1977; Miller et al., 1986; Cole, 2009). Arctic lakes are typically limited by phosphorus concentrations (Schindler et al., 1974), but evidence suggests some Arctic lakes may be limited by nitrogen concentrations (Levine & Whalen, 2001) or co-limited by both phosphorus and nitrogen concentrations (Ogbebo et al., 2009).

Arctic lakes are generally shallow and subjected to 24 hours of daylight during the summer months. This may result in an increase in the risk of exposure of UV radiation to planktonic communities in these small lakes (Rautio et al., 2011). Arctic phytoplankton and zooplankton taxa commonly produce UV-absorbing pigments such as mycosporine-like amino acids and scytonemin as an adaptive mechanism (Garcia-Pichel & Castenholz, 1991; Karentz et al., 1991; Leavitt et al., 1997). The depth at which incoming UV radiation can penetrate the water is dependent on chromophoric dissolved organic carbon (DOC) (Zagarese & Williamson, 2001; Rautio & Korhola, 2002; Vonk et al., 2015). DOC concentrations in Arctic lakes are being modified as a result of climate warming and permafrost thaw (Vonk et al., 2015), with important implications for many limnological processes, including UV penetration (Vonk et al., 2015).

2.2 Impact of climate change on Arctic lakes

The rise in air temperatures in Arctic region over the past several decades has been ~2 times larger than the global average (ACIA, 2005; Solomon et al., 2007; IPCC, 2013). Recorded average surface air temperatures have increased since 1900's, where 2015-2016 was the warmest recorded year followed by 2017-2018 (AMAP, 2019). Arctic lakes have experienced several limnological changes in response to warming Artic air temperatures and permafrost thaw (Smith et al., 2005; Smol et al. 2005). These implications include changes in the abundance, distribution, and properties of Arctic lakes, driving changes in the overall Arctic landscape (Mackay, 1992; Smith et al., 2005).

Arctic lakes are sensitive to air temperatures and shifts in the water balance such that changes in evaporation and precipitation may cause changes in lake depth and even lead to lakes drying up, and/or transitioning into a wetland (Smith et al., 2005; Smol & Douglas, 2007; Vincent, 2009). This is because many Arctic lakes are shallow and have large surface to volume ratios (Vincent, 2009). Glacier melt is also causing increases in lake growth and formation in several Arctic regions (Shugar et al., 2020). An assessment of global glacial lakes between 1990 and 2018 found that glacial lakes have increased in total area by 51% and 53 new lakes formed as a result of inputs of glacial meltwater from glacier retreat caused by climate warming (Shugar et al., 2020). Furthermore, concerns of glacial lake outburst floods may increase in frequency and intensity as a result of glacial meltwater inputs, which may have implications for nearby communities and ecosystems (Shugar et al., 2020).

Surface water temperature, which is influenced by air temperatures, is a major factor influencing biodiversity of Arctic freshwaters (Rhode, 1992). A poleward shift of warm-adapted species to warming regions of the low Arctic is expected, and Arctic species endemic and adapted to the region may decline or become extirpated due to competition and shifts in food web structure (Jeppesen et al., 2010; Vincent et al., 2013). Fish communities in particular are suspected to shift as cold-adapted fish may become physiologically stressed (Jeppesen et al., 2010). The impacts of climate warming on Arctic freshwater fish will be partly determined by climate warming impacts to ice phenology and water quality changes (Jeppesen et al., 2010).

2.2.1 Climate change impacts on lake ice phenology

The well-established correlation between air temperature and ice phenology (Palecki & Barry, 1986) is a useful indicator of regional climate trends and variability, and climate change (Brown & Duguay, 2010; Šmejkalová et al., 2016; Sharma et al., 2019). Many Arctic lakes are experiencing longer open-water seasons via earlier ice break-up in the spring/summer and late ice formation in the fall/winter because of climate warming (Prowse et al., 2011; Duguay & Brown, 2018). Despite the trend of later ice formation and earlier ice decay, there is regional variability where some lakes experience earlier freeze-up and/or later ice break-up (Duguay & Brown, 2018). For example, western Canadian Arctic lakes are experiencing later freeze-up and earlier ice break-up, while many northern Siberian lakes experienced earlier freeze-up but later ice break-up (Duguay & Brown, 2018). In northern Scandinavia, earlier freeze-up occurred, while later freeze-up occurred in southern Finland (Duguay & Brown, 2018). Overall, ice cover duration is 1 to 20 days shorter for western Canadian Arctic lakes and Southern Finland, and in northern Siberia and northern Scandinavia, ice cover duration is approximately 1 to 20 days longer (Duguay & Brown, 2018). The changes in lake ice phenology occurring in response to climate warming are likely to impact biogeochemical cycling and ecological interactions in Arctic lake ecosystems.

When Arctic lakes cross climatic thresholds, this may drive shifts in biotic communities and foodweb structure that can occur within a relatively short period of time (Smol et al., 2005; Prowse et al., 2006; Vincent et al., 2013). Paleolimnological records from subarctic and high Arctic lakes have revealed abrupt shifts in diatom taxa, characterized by a large increase in planktonic diatoms (primarily *Cyclotella*) and a transition in diatom life strategy from benthic to planktonic forms linked to shorter ice cover duration, and thus increased planktonic habitat and light availability, as a result of recent climate warming (Rühland et al., 2003; Smol et al., 2005). Increases in diatom species diversity have also been observed in response to increased growth of aquatic plants and mosses that increase habitat complexity for littoral and benthic forms (Rühland et al., 2003; Smol et al., 2005). Paleolimnological studies have also reported increases in primary production in Arctic lakes linked to a longer ice-free growing season (Overpeck et al., 1997; Douglas & Smol, 1999; Michuletti et al., 2005). Changes in water quality variables like DOC and nutrients with climate warming also play a role in driving increases in Arctic lake primary production (Seekall et al., 2015).

2.2.2 Climate change impacts on Arctic lake water quality

Water quality is a term that refers to parameters that are important for lake ecosystem structure and function, such as DOC, nutrients, and major ions (Adrian et al., 2009). DOC in particular is an important parameter for mediating limnological responses to Artic climate warming (Williamson et al., 1999, 2009; Seekall et al., 2015). DOC plays a significant role in carbon and energy cycling in lakes (Battin et al., 2009). DOC can have both positive and negative effects on primary production. DOC can stimulate primary production by protecting phytoplankton from incoming UV radiation and providing a source of nutrients; however DOC may absorb light, which can limit photosynthesis (Klug, 2002; Vincent, 2009). Factors such as catchment characteristics, regional climate, autochthonous production, and mineralization are key controls on DOC concentrations in lakes (Toming et al., 2020). Increased DOC concentrations have been observed in Arctic lakes partly due to the indirect effects of climate warming (Vincent, 2009; Reyes & Lougheed, 2015; Seekall et al., 2015).

2.3 Permafrost in Arctic and subarctic regions

Permafrost is distributed across the circumpolar north on land and underneath continental shelves (Brown et al., 1997). Approximately 24% of land in the Northern Hemisphere is underlain with permafrost (Zhang et al., 2008). Permafrost is defined as ground that remains frozen (below 0°C) for two or more consecutive years (van Everdingen, 1998). The distribution of permafrost is categorized into continuous and discontinuous permafrost zones (Brown & Péwé, 1973). Permafrost underlies 90-100% of the landscape in the continuous zone while permafrost in the extensive discontinuous permafrost zone underlies 50-90% (Brown & Péwé, 1973; Brown et al. 1997; Shur & Jorgenson, 2007). Sporadic permafrost has 10-50% permafrost extent and isolated permafrost has 0-10% permafrost extent (Brown, 1967).

Permafrost thaw has been accelerating in response to anthropogenic climate change (Schuur & Abbott, 2011). Drivers of permafrost thaw including warming air temperatures, changes in precipitation and snow cover, and fire (Screen & Simmonds, 2010; French et al., 2015; Douglas et al., 2020). Recent findings have shown that Arctic sea ice extent and duration has experienced significant reductions, and that annual precipitation has increased while snow cover duration has decreased (Screen & Simmonds, 2010; Douglas et al., 2020). Reduced sea ice cover may contribute to Arctic amplification resulting from changes in heat and moisture transport inland that may lead to increased rainfall and snowfall (Dai et al., 2019; Vaks et al., 2020). Increased rainfall may cause surface water inundation and as a result provide a heat source for permafrost to thaw, while increased snowfall can insulate the ground from cold winter air temperatures, which may induce thaw during the autumn and winter months (Liu et al., 2012; Douglas et al., 2020; Vaks et al., 2020). Lastly, warmer summers accompanied by long durations of dry periods are expected to occur in some areas, providing a substantial amount fuel for fire ignition (French et al., 2015). Fire can lead to deepening of the active layer and increase soil temperatures at recently burned sites (French et al., 2015). The deepening of the active layer can persist for several years after the disturbance, and talik (area of unfrozen ground in permafrost) formation may occur (Yoshikawa et al., 2002; Gibson et al., 2018).

2.4 Permafrost thaw impacts on Arctic lakes

Thawing permafrost has several implications for Arctic lakes at local to global scales. For example, a 15-year study of two High Arctic lakes documented that the thickening of the soil active layer enhanced subsurface water transport to adjacent streams and rivers, which resulted an increase in lakewater conductivity to streams and downstream lakes (Lamoureux & Lafrenière, 2017). There was also a large increase in sulphate (SO_4) concentrations over time in one of the lakes, indicating that lake chemistry became more similar to the chemistry of the soil active layer, as a result of thawing permafrost (Lamoureux & Lafrenière, 2017).

One dominant feature of the Arctic is thermokarst, used to describe processes and landforms associated with the melting of ground ice (Burn, 2013). In the continuous permafrost zone, thermokarst processes initiate with melting of polygonal and/or ice wedge networks (Czudek & Demek, 1970), where in the discontinuous permafrost zone thermokarst processes begin with the thawing of peat plateaus, palsas and lithalsas (Luoto & Seppälä, 2003).

Thermokarst lakes and ponds are widespread in areas with ice-rich permafrost, such as northern Siberia, western Canadian Arctic, Hudson's Bay lowlands, and Alaska (Côté & Burn, 2002; Walter et al., 2006). Approximately 90% of lakes in the Russian permafrost zone are comprised of thermokarst lakes (Walter et al., 2006). Thermokarst lake depth is dependent on ground ice content and distribution, local topography, and climate; most thermokarst lakes are <1 m in depth while some range up to 10 m in depth (Pienitz et al., 2008; Bouchard et al., 2016). Once thaw depth exceeds maximum ice cover, the lake bottom remains unfrozen annually forming a talik (Grosse et al., 2013; Bouchard et al., 2016). Thermokarst lakes expand laterally by thermal and mechanical erosion into the surrounding permafrost soil which may result in changes to lake characteristics (e.g. lake water chemistry and morphometry) (Grosse et al., 2013; Bouchard et al., 2016). The ultimate fate of thermokarst lakes involve several terminal mechanisms, such as: (1) rapid drainage resulting from shoreline breaching after a large precipitation event (Turner et al., 2010); (2) factors leading to increased evaporation (Bouchard et al., 2013b); (3) revegetation via peat accumulation or infilling (Payette et al., 2004; Roach et al., 2011); and (4) subsurface drainage through an open talik (Yoshikawa & Hinzman, 2003). Thermokarst lakes may fully or partially drain overtime and may occur rapidly and/or slowly (Hinkel et al., 2007).

Recent studies have shown the number of thermokarst lakes have increased in some areas of the continuous permafrost zone (Jones et al., 2011) while thermokarst lakes decreased in some areas of the discontinuous permafrost (Smith et al., 2005). In the discontinuous permafrost zone, many lake drainage events are suspected to occur due to thinner and less contiguous permafrost (Smith et al., 2005), while areas of thicker permafrost and ground ice content in the continuous permafrost zone, thermokarst activity is suspected to increase lake formation (Smith et al., 2005). However, the impact of climate warming on lake drainage in the continuous permafrost zone may vary across regions. For example, one study reported an increase in the number of lakes but a decrease of total lake area in northern Seward Peninsula, Alaska likely due to the partial drainage of several lakes that led to the formation of small remnant lakes (Jones et al., 2011). The changes in lake drainage across the Arctic region are associated with changes in temperature and precipitation, and these changes are a result of climate warming (Vincent, 2009). Changes in air temperatures and precipitation can directly and indirectly affect Arctic lakes through changes in the water balance and ice phenology which may have adverse effects on limnological characteristics of Arctic lakes (Vincent, 2009).

Thermokarst activity may affect limnological properties related to changes in particulate and dissolved constituents delivered to lakes (Kokelj & Jorgenson, 2013). Active layer deepening, operating over longer time scales (i.e. decades or longer), typically enhances delivery of nutrients and DOC to lakes (Vonk et al., 2015). In contrast, thermokarst processes are abrupt and occur on shorter timescales (years to decades). Thermoerosion (i.e. active layer detachments, thermokarst gullies, and thaw slumps) is a thermokarst process that delivers high volumes of particulate and soluble materials to lakes, which may have intense impacts on ecosystem functioning (Vonk et al., 2015). Retrogressive thaw slumps are a type of thermoerosional process that are prominent disturbances on the shorelines of lakes, streams, and coastal areas of Arctic regions with ice-rich permafrost (Burn & Lewkowicz, 1990).

2.5 Retrogressive thaw slumps

Retrogressive thaw slumps are found across the western Canadian Arctic (Lantz & Kokelj, 2008), Alaskan (Jorgenson et al., 2008; Balser et al., 2009), and northern Siberian landscapes (Astakhov et al., 1996; Alexanderson et al., 2002; Leibman et al., 2014). Retrogressive thaw slumps typically develop adjacent to shorelines of lakes, rivers, and coastal areas (Kokelj et al., 2017) to form landslide-like U-shaped scars consisting of a retreating headwall and a mud slurry slump floor (Lantz & Kokelj, 2008). Retrogressive thaw slumps can initiate through various thermal and geomorphic processes that are coupled to local climatic conditions (Kokelj et al., 2005; Lantuit & Pollard, 2005: Segal et al., 2016). Warming air temperatures increase the thickness of the active layer, exposing ice-rich soils that subsequently thaw (Kokelj et al., 2005; Lantuit & Pollard, 2005), while increased precipitation can trigger mass wasting events, forming debris tongues destabilizing the underlying ice-rich permafrost terrain (Kokelj et al., 2015). Thaw slump development can also occur in response to other disturbances, such as fire. For example, one study observed increased thaw slump activity following fire events around Alaskan lakes over the past 3000 years, suggesting that the fires facilitated thermoerosion, while insulative post-fire vegetation slowly deepened the active layer over decades (Chipman & Hu, 2017).

Thaw slumps can remain active for decades as exposed ground ice in the slump headwall ablates, causing the slump headwall to retreat upslope and expand (Lantz & Kokelj, 2008; Lacelle et al., 2010). During the active phase where the slump continues to grow, thaw slump development can impact large areas of terrain, and materials thawed from the retreating headwall are transported as large volumes of eroded sediments into adjacent waterbodies (Burn & Lewkowicz, 1990). Thaw slumps stabilize when no more ground ice is exposed in the headwall, either because all available ground ice has been melted or covered up by debris (Burn & Friele, 1989). Once the slump has stabilized, revegetation of the slump scar occurs (Burn & Friele, 1989). Lake shoreline retrogressive thaw slumps may exhibit polycyclic behaviour, where slump stabilization and reinitiation occurs repeatedly in a cyclical pattern as a result of feedback mechanisms (Kokelj et al., 2009).

Retrogressive thaw slumps have been increasing in frequency and intensity in the Mackenzie Valley and Mackenzie Delta region in recent decades, primarily due to climate warming (Lantz & Kokelj, 2008; Kokelj et al., 2017). Average thaw slump size and growth rates have been increasing from 0.31 to 1.82 ha and 169 to 465 m^2/year, respectively, over the last 50 years (Segal et al., 2016). This has substantial implications for the small lake ecosystems that are abundant across this region.

2.5.1 *Impacts of shoreline retrogressive thaw slumps on lakes of the Mackenzie Delta uplands (Northwest Territories, Canada)*

Small lake ecosystems in the Mackenzie Delta uplands impacted by shoreline thaw slumps typically exhibit elevated concentrations of major ions, conductivity, alkalinity, and pH relative to nearby unimpacted lakes due to large inputs of siliclastic, ion-rich material being eroded from the slumps and transported into the lakes (Kokelj et al., 2005). Slump-impacted lakes also exhibit higher water clarity and lower nutrient (nitrogen and phosphorus) concentrations (Kokelj et al., 2009; Thompson et al., 2012; Houben et al., 2016). This is because inorganic sediments transported into the lake binds to and scours coloured DOC and nutrients from the water column, which is then deposited to the bottom lake sediments (Thompson et al., 2008; Houben et al., 2016). Impacts of shoreline thaw slumps on the physical and chemical characteristics of lakes can persist for years after thaw slumps have stabilized and re-vegetated (Kokelj et al., 2005; Houben et al., 2016), although conductivity and alkalinity concentrations generally decrease while nutrient concentrations increase with increasing slump age (Kokelj et al., 2009).

The physical and chemical alterations that occur in slump-affected lakes have corresponding implications on lake ecology. Within slump-affected lakes, benthic production may increase as a result of increased light penetration and increased nutrients in the sediments that were scoured from the water column (Moquin et al., 2014). This has been linked to higher biomass and well-developed submerged macrophyte communities in slump-impacted lakes, resulting in a more complex benthic habitat to support associated periphyton and macroinvertebrate communities (Mesquita et al., 2010). This was reflected in a paleolimnological investigation of slump-impacted and reference lakes, which showed an increase in periphytic diatoms following the development of shoreline thaw slumps likely tracking increases in macrophyte biomass (Thienpont et al., 2013). Macroinvertebrate abundances have also been shown to be higher in slump-impacted compared to undisturbed lakes, with higher abundances of ostracods and nematodes, and lower abundances of chironomids (Moquin et al., 2014). The differences in macroinvertebrate abundances and composition were related to sediment chemistry (particularly calcium, organic carbon, and nitrogen) and macrophyte biomass (Moquin et al., 2014).

Chlorophyll α, an indicator of overall primary production, has been shown to be lower in slump-affected lakes compared to nearby undisturbed lakes, suggesting that thaw slumping results in a decrease in open-water primary productivity (Thompson et al., 2008; Thompson et al., 2012; Houben et al., 2016). Impacts of thaw slumps on phytoplankton species assemblages have not yet been investigated, but as light and nutrients are known controls on algal species distributions (Peterson et al., 1987; Schindler et al., 2008), thaw slumping likely also drives changes in overall phytoplankton community composition.

2.6 Zooplankton community responses to retrogressive thaw slumps

Zooplankton are small aquatic animals that drift within the water column, occupying important trophic roles in lake ecosystems. In Arctic tundra lakes, zooplankton communities tend to be dominated by copepods and rotifers (Lindsey et al., 1981; Swadling et al., 2000). Copepods in tundra lake ecosystems have relatively high fat stores which make them a preferable prey item for Arctic birds, planktivorous fish, macroinvertebrates, and larger zooplankton (Norrbin, 1991). Rotifers are also an important component of the zooplankton community in Arctic lakes (Rublee, 1998). Rotifers vary in shape and size, and small bodied rotifers provide viable food sources for larger rotifers and other zooplankton, while large-bodied rotifers are food sources for large macroinvertebrates and planktivorous fish (Swadling et al., 2000). Cladocerans, mainly comprised of *Daphnia* and *Bosmina* species in the open-water zone, are essential to lake ecosystem functioning as grazers on phytoplankton, and as a food source for fish and macroinvertebrates (Lindsey et al., 1981; Swadling et al., 2000). *Daphnia* in particular are often considered keystone species in lake ecosystems due to their large body size, which makes them efficient filter feeders and a desired prey source for fish (Lampert, 2011).

Zooplankton are useful ecological indicators for monitoring the health of lake ecosystems, but their response to permafrost thaw and retrogressive thaw slumping is poorly understood. One study observed that zooplankton abundances, diversity, and evenness in lakes in the Mackenzie Valley region were largely structured by water quality parameters projected to change with permafrost thaw (Vucic et al., 2020). The authors' predictive models indicated increased calcium concentrations would drive an increase zooplankton abundances while increased conductivity would drive a decrease in zooplankton diversity and evenness (Vucic et al., 2020). Zooplankton are strongly influenced by water quality variables known to be impacted by thaw slumping, including pH, conductivity, DOC, nutrients, and calcium (Bos et al., 1996; Williamson et al., 1999; Houben et al., 2016). Consequently, the intensification of permafrost thaw slumping projected under future climate warming scenarios (Kokelj et al., 2017) has the potential to be an important driver of future changes in Arctic lake zooplankton communities, in particular across regions predisposed to thaw slump development (Kokelj et al., 2017).

For my undergraduate honour's thesis research, I compared surface water chemistry and zooplankton assemblages from 12 pairs of lakes in the Mackenzie Delta uplands, with one lake impacted by thaw slumping, and a neighboring reference lake unaffected by slumping, to compare zooplankton assemblages between undisturbed and thaw slump-affected lakes. My results showed *Daphnia* were more prevalent in reference lakes which had higher lake colour and phosphorus concentrations, whereas *Keratella* was associated with thaw slump lakes that had higher conductivity and alkalinity (Auger, unpublished data). *Daphnia* were absent in slump-impacted lakes suggesting that *Daphnia* may be particularly sensitive to the limnological changes associated with thaw slumping, but further investigation of other biotic and abiotic factors are still needed better understand the potential mechanisms as to why *Daphnia* were absent in slump-affected lakes.

2.6.1 Impacts of ultraviolet (UV) radiation on lake planktonic communities

A potential stressor on planktonic communities resulting from thaw slumping is the increased exposure of potentially harmful UV radiation within the water column caused by the increase in water clarity in slump-impacted lakes. The increase in direct UV exposure to lake biota can have negative impacts, such as chronic (physiological) effects which may inhibit growth of some biota (i.e. primary producers) and cascade through the food web (Tucker & Williamson, 2011). Phytoplankton and zooplankton exposed to harmful UV radiation can experience reduced productivity, growth and development, and increased mutation rates (Häder et al. 2007). For example, periphyton development was suppressed when exposed to UV radiation caused by the inhibition of photosynthesis (primary production) (Vinebrooke & Leavitt, 1996). Another study reported an increase in *Daphnia* mortality rates when exposed to higher UV radiation as DOC concentrations decreased from 10 to 4.7 mg/L, with the highest mortality rates reported when the DOC concentration was below 4.7 mg/L (Rautio & Korhola, 2002). Small Arctic lakes are generally shallow and subjected to 24 hours of daylight during the summer months. When impacted by a thaw slumps, the increased water clarity and decreased DOC, coupled with 24 hours of daylight and lake shallowness, may increase the risk of exposure of UV radiation to aquatic biota in these small Arctic lakes. Long-term data is needed to understand the impact of thaw slumping on lake biota, including changes in UV radiation. Long-term data are rarely available, especially in Arctic lakes. Fortunately, paleolimnological techniques (reconstructing environmental change using proxy data preserved in lake sediments) can provide the missing long-term perspective.

2.7 Paleolimnology

Paleolimnology is a multidisciplinary science that uses biological, chemical, and physical information preserved in lake sediments to infer past environmental changes (Smol, 2008). Paleolimnological studies use a variety of biological proxies and indicators for inferring various past environmental changes. Numerous studies have used diatoms (Bacillariophyceae; single-cell siliceous algae) (Davis, 1987), cladoceran subfossils (microcrustaceans) (Tolotti et al., 2016), *Chaoborus* mandibles (Chaoboridae; a macroinvertebrate with an aquatic larval stage) (Sweetman & Smol, 2006), and chironomid (Chironomidae; non-biting midges) head capsules (Walker, 1987) as biological indicators and proxies to infer an array of environmental changes related anthropogenic impact and/or climate warming. For example, diatom assemblages have been used to reconstruct past lake acidification based on known pH optima and tolerances of different diatom taxa (Davis, 1987).

The basic paleolimnological method involves selecting a study lake and retrieving sediment from a site using a variety of different corer types (Smol, 2008). Sediment corers optimized for collecting recent sediments (last ~500 years) can be broadly categorized into open-barrel gravity corers and freeze-crust samplers (Smol, 2008). Open-barrel gravity corers (e.g. coring tube) use gravity to vertically penetrate lake sediments. After the tube has sunk into the sediments, the top of the tube is closed using a messenger or close-on-contact mechanism that triggers the seal to close remotely; from here the sediment core sample can be retrieved by pulling the core back up to the surface where the bottom of the core can be plugged. Freeze-crust sampler use a wedge-shaped corer filled with a coolant immersed with alcohol, which is then lowered into the sediments and held in position for a period of time (i.e. 10 minutes) until the sediment freezes onto the sampler (Smol, 2008). After freezing, the sampler is retrieved and the sediment is subsampled (Smol, 2008). Freeze-crust sampler is often used when lake sediments contain high water content and/or gases that may disrupt the sediment profile during the coring processes (Smol, 2008). Both corers typically retrieve cores less than one meter in length representing the last several hundred years of lake histories. of sediment. Piston corers are used to retrieve longer sediment profiles (i.e. > l m) that incorporate millennial time scales, but piston corers are not suited well to collected recently deposited, watery sediments that are easily captured by gravity or freeze corers (Smol, 2008).

Selecting a coring site in the study lake should ideally represent the average accumulation of material for the entire basin; however, determining the average accumulation for the entire basin may be difficult. Coring sites are chosen strategically to avoid lateral sediment movements in steep morphometric gradients, and bioturbation and mixing processes in shallow areas, which would obscure interpretations of environmental change (Smol, 2008). Ideally, coring sites should be located in a flat, central basin in deeper part of the lake to provide the most complete sediment archive that will be representative of the entire basin (Smol, 2008).

Once a sediment core is taken, it is sectioned into intervals often between 0.5-1 cm, which would be representative of typically 2-10 years of accumulation (Smol, 2008). However, analyzing every sediment interval is a detailed approach and allows only a limited number of lakes to be analyzed at high temporal resolution. Some questions of broader, regional interest may require a "before-and-after" perspective of environmental change. A "top-bottom" approach analyzes a large number of lakes by examining only the top and bottom sediment layers to provide insight into recent limnological conditions (top layer) compared to pre-1850 limnological conditions (bottom layer) (Cumming et al., 1992). The "top-bottom" approach provides a snapshot of regional change, however, as a trade-off for analyzing a large number of lakes, the timing of change cannot be determined.

Once a core is sectioned, a chronology is established using different radioisotopes to date the sediment core. ^{210}Pb and ^{137}Cs are used for recent sediments, where ^{210}Pb is used to date sediment cores back to approximately 150 years (pre-1850). ^{137}Cs can delineate specific time periods associated with atmospheric nuclear bomb testing that occurred between 1945 and 1963 (Smol, 2008). ^{14}C is used to date organic deposits in sediments from several hundred years to approximately 40,000 years (Smol, 2018). There are two models that can be used to derive an age-depth model using ^{210}Pb and ^{137}Cs activities: 1) the constant initial concentration (CIC) model and 2) constant rate of supply (CRS) model (Appleby & Oldfield, 1983). The CIC model assumes both sedimentation rate and input of ^{210}Pb into lakes remained constant throughout time (Appleby & Oldfield, 1983), whereas the CRS model assumes a constant input of ^{210}Pb into lake waters and sediments irrespective of varying sedimentation rates (Lubis, 2006). The CRS model is typically applied when sediment inputs have been expected to change throughout time (Appleby & Oldfield, 1983; Lubis, 2006).

2.7.1 Cladoceran subfossils as paleolimnological indicators

Cladocerans (a group of crustacean zooplankton) are ubiquitous in lake ecosystems and leave identifiable subfossil remains preserved in lake sediments (Korosi & Smol, 2012). Cladoceran remains are often well-preserved due to the presence of chitin, a substance resistant to degradation that is a major constituent of the exoskeleton (Frey, 1976). Cladocera have been used as paleoindicators of acidification (Walseng et al., 2003; Jeziorski et al., 2008), food web changes (Jeppesen et al., 2001; Korosi et al., 2012), changes in water depth (Siitonen et al., 2011), and changes in aquatic habitat (Płóciennik et al., 2020).

Cladocerans exposed to UV radiation increase melanin pigment production in the carapace as a mechanism to protect themselves from further damage to UV radiation and increase survival under intensive UV conditions (Rautio et al., 2009). Upon death and disarticulation, melanin pigment remains well-preserved in the carapace subfossils in lake sediments (Nevalainen & Rautio, 2014). Previous studies have used spectral absorbance measurements of cladoceran carapaces to infer past exposure to UV radiation (Nevalainen & Rautio, 2014), elucidating past changes in coloured dissolved organic matter (CDOM) concentrations and exposure to UV radiation (Nevalainen et al., 2020).

Dormancy is a common feature among zooplankton as a mechanism to disperse between habitats and to cope with deteriorating lake conditions (Gyllström & Hansson, 2004). *Daphnia* produce ephippia (resting eggs) when environmental conditions deteriorate as a mechanism to promote genetic diversity within populations and promote dispersion by birds and aquatic insects to colonize new habitats (Allen, 2007). *Daphnia* ephippia can accumulate and remain well-preserved in lake sediments for decades (Jeppesen et al., 2002), allowing for their use in paleolimnological investigations. For example, lake acidification impacts on fish have been reconstructed using *Daphnia* ephippia and *Chaoborus* mandibles, because *Daphnia* and *Chaoborus* are large-bodied biota that are vulnerable to fish predation (Palm et al., 2012).

2.7.2 Previous paleolimnological investigations into retrogressive thaw slumping in the Mackenzie Delta uplands

Paleolimnological investigations can provide critical insights into our understanding of the impacts of retrogressive thaw slumping on lakes. The present-day differences in water quality between slump-affected and reference lakes are well-characterized, (Kokelj et al., 2005; Thompson et al., 2012) but long-term changes associated with thaw slumping are lacking. This is a key knowledge gap as thaw slumps are increasing in frequency and intensity with climate warming, and slump impacts to lake water quality can persist for decades after a slump stabilizes (Kokelj et al., 2005; Lacelle et al., 2010; Houben et al., 2016).

Deison et al. (2012) used paleolimnological techniques to demonstrate that sedimentation rates increased in slump-affected lakes compared to nearby undisturbed lakes, particularly inorganic sediment, diluting organic carbon and mercury contributions to the sediments. Eickmeyer et al. (2016) documented higher concentrations of persistent organic pollutants (POPs) in sediments of slump-impacted lakes when corrected for sediment organic carbon, compared to nearby undisturbed lakes. The authors suggested that solvent switching processes explained the higher concentrations of POPs per gram organic carbon in the sediments, where POPs are adsorbing onto a smaller pool of dissolved and particulate organic carbon. Thienpont et al. (2013) used a diatom-based paleolimnological approach to infer algal responses to thaw slumping. Their study noted an increase in periphytic diatom diversity and abundances associated with the approximate timing of slump initiation, consistent with an increase in water clarity and development of macrophyte communities. My thesis builds on these studies to address whether or not thaw slumping increases the risk of exposure of cladocerans to potentially harmful UV radiation, and whether *Daphnia* are absent in slump-affected lakes.

CHAPTER 3: A COMPARATIVE RECONSTRUCTION OF TRENDS IN *CHYDORUS* UV EXPOSURE ASSOCIATED WITH SHORELINE RETROGRESSIVE THAW SLUMPING IN LAKES OF THE MACKENZIE DELTA UPLANDS (NORTHWEST TERRITORIES, CANADA)

3.1 Abstract

Retrogressive thaw slumping is increasing in intensity and frequency in the Mackenzie Delta uplands as a result of climate warming. Slump-affected lakes have higher water clarity compared to undisturbed lakes, and the increase in water clarity in these small, shallow lakes has the potential to increase the risk of exposure of aquatic biota to potentially harmful UV radiation. *Chydorus* subfossil carapaces were extracted from lake sediment cores and subjected to absorbance measurements at UV wavelengths to reconstruct long-term changes in UV exposure between an undisturbed lake (reference, high DOC) and a slump-affected lake (low DOC). There was an increasing trend of UV exposure over time in Lake 9A (reference lake), but there was not a directional trend in Lake 9B (slump-affected lake). The carapace absorbances in Lake 9B are comparable to that in Lake 9A, indicating *Chydorus* in Lake 9B are not being exposed to higher levels of UV radiation as a result of increases in water clarity. Slump-affected lakes have higher macrophyte biomass that may provide shading from UV radiation. The mechanism underlying the trend in Lake 9A is unclear, and may be related to complex interacting factors like fluctuating lake levels, lake depth, and littoral and near-surface habitat usage by *Chydorus*, changes in cloud cover, or changes in DOC concentrations. My study provides some initial insight into the temporal dynamics of exposure of *Chydorus* to UV radiation in a slump-affected lake and undisturbed lake. Further investigation is needed since there may be differences among slump-affected lakes with different DOC concentrations, as well as the quality of DOC between undisturbed and slump-affected lakes.

3.2 Introduction

Retrogressive thaw slumps are common on lake, river, and coastal margins in ice-rich permafrost regions (Burn & Kokelj, 2009; Kokelj et al., 2017). They are abundant in the western Canadian Arctic (Lantz & Kokelj, 2008), Alaska (Jorgenson et al., 2008; Balser et al., 2009), and northern Siberia (Astakhov et al., 1996; Alexanderson et al., 2002; Leibman et al., 2014). Over the past 50 years, the frequency and intensity of retrogressive thaw slump formation on lake shorelines in the Mackenzie Delta region (Northwest Territories) of the western Canadian Arctic have increased primarily due to climate warming (Lantz & Kokelj, 2008; Kokelj et al., 2017), with implications for lake ecosystem functioning (Kokelj et al., 2005).

Slump-affected lakes in the Mackenzie Delta uplands typically exhibit higher conductivity, alkalinity, and pH, and lower dissolved organic carbon (DOC) and nutrient (nitrogen and phosphorus) concentrations compared to undisturbed lakes (Kokelj et al., 2009; Thompson et al., 2012; Houben et al., 2016). The erosion of inorganic sediments from slumps into lakes binds to DOC and nutrients and scours them from the water column, increasing water clarity (Figure 3.1) as the sediments are deposited to the lake bottom (Kokelj et al., 2005, 2009). Primary production is also reduced in slump-impacted compared to undisturbed lakes, likely due to lower nutrient availability (Houben et al., 2016). The increase in water clarity may increase the risk of exposure of planktonic and benthic organisms to potentially harmful UV radiation because UV radiation may penetrate deeper into the water column without sufficient DOC for attenuation (Williamson et al., 1996; Häder et al., 2007).

Figure 3.1: Water samples collected in July 2017 from an undisturbed lake (7a) and slump-affected lake (7b) in the uplands east of the Mackenzie River Delta (Inuvik, Northwest Territories, Canada). Note the difference in apparent lake colour between 7a (high chromophoric DOC) and 7b (low chromophoric DOC).

Arctic lakes experience seasonal fluctuations in annual solar radiation budget. Snow and ice cover persist on lakes upwards to 8 months of the year and lakes receive minimal to no solar radiation input. In contrast, lakes experience 24 hours of sunlight during the short summer months. The 24 hours of daylight Arctic lakes experience during the summer months may act as a potential stressor on planktonic communities and have an overall negative impact on lake ecosystems (Perin & Lean, 2004). For example, UV exposure experiments revealed *Asplancha* spp., *Ceriodaphnia quadrangula*, and *Bosmina longirostris* to be highly sensitive to UV radiation (Williamson et al., 2001), and suggested exposure to UV radiation plays an important role in distribution and abundances of zooplankton species in Arctic lakes of different ages (Williamson et al., 2001).

The depth at which incoming UV radiation can penetrate the lake water is dependent on the quality and quantity of chromophoric (coloured) DOC (Zagarese & Williamson, 2001; Rautio & Korhola, 2002). Previous studies have identified a DOC threshold concentration of 5 mg/L, below which UV radiation is likely to negatively impact aquatic biota (Williamson et al., 1996; Häder et al., 2007). One study reported UV radiation was able to penetrate several meters deep into the water column of some lakes with DOC concentrations < 2 mg/L (Schindler, 1996). Mackenzie Delta upland lakes impacted by thaw slumps are low in chromophoric DOC, which in combination with generally shallow lake depths (< 5 m), the intensification and increased frequency of thaw slumping related to climate warming (Kokelj et al., 2017) has the potential to increase the risk of exposure of zooplankton and benthos to UV radiation through decreases in chromophoric DOC.

Aquatic biota have several adaptations to mitigate the impacts of UV exposure. For example, zooplankton can increase melanin pigment production in their exoskeleton when exposed to UV radiation (Rautio & Tartarotti, 2010). Cladocerans (crustacean zooplankton) leave well-preserved identifiable remains in lake sediments, and melanin pigment concentrations can be inferred based on spectral analyses of their carapaces (Nevalainen & Rautio, 2014). This allows paleolimnological methods to be applied to reconstruct past exposure of cladocerans to potentially harmful UV radiation within the water column, including open-water *Daphnia* species and the littoral cladocerans *Alona* and *Chydorus* (Rautio & Nevalainen, 2013; Nevalainen & Rautio, 2014). This approach has been used to elucidate long-term changes in cladoceran communities linked to changes in lake water optics attributable to direct UV impact or climate-mediated intensification of photodegradation of allochthonous DOC (Nevalainen et al., 2018).

In this study, I reconstructed past exposure of cladocerans to UV radiation in two lakes in the Mackenzie Delta region (one slump-affected and one undisturbed) by inferring melanin content in the carapaces of *Chydorus* and *Alona* in lake sediment cores. The aim of this study is to determine if thaw slumping increases exposure of cladoceran communities to potentially harmful UV radiation. I hypothesized that: 1) *Chydorus* and *Alona* carapace absorbances will be higher in the slump-affected lake compared to the neighbouring undisturbed lake; and 2) *Chydorus* and *Alona* carapaces absorbances will increase from past to present in the slump-affected lake while carapace absorbances in the neighbouring undisturbed lake will have remained relatively consistent through time. This study provides new insights into the temporal dynamics of cladoceran exposure to UV radiation in Arctic lakes vulnerable to thaw slumping.

3.3 Methods

3.3.1 Study site description

The uplands east of Mackenzie Delta and north of the town of Inuvik, Northwest Territories, Canada are comprised of the Tuktoyaktuk Coastlands and Anderson Plain physiographic regions (Rampton, 1988), and located in the northern Taiga Plains and Tundra Plains ecoregions (Ecosystems Classification Group, 2012). The landscape is lake-rich, and terrain is underlain by thick (>100 m) ice-rich continuous permafrost (Burn & Kokelj, 2009). The study region spans the transition from the boreal forest to the low-shrub tundra (Lantz et al., 2010), and thermokarst activity is well-documented in the area (Lantz & Kokelj, 2008; Kokelj et al., 2017). Mean annual air temperatures at Inuvik are -9.2°C (Environment Canada), and historical climate data indicate significant warming has occurred in the region since the 1960's (Thienpont et al., 2015). Predicted air temperature increases in the region are among the highest in Canada (Prowse et al., 2009), and this has important implications for the stability of ice-rich permafrost in the region (Lantz & Kokelj, 2008; Kokelj et al., 2017).

The Mackenzie Delta region is characterized by numerous small, shallow lakes that have been well-studied in previous investigations into the physical and limnological effects of thaw slumping (Kokelj et al., 2009; Thompson et al., 2012; Houben et al., 2016). I selected a pair of neighbouring study lakes (Lakes 9A and 9B) west of Parsons Lake (Figures 3.2 and 3.3). Lake 9B has a shoreline thaw slump with a polycyclic slump history, meaning that it has experienced multiple episodes of slump initiation, stabilization, and re-initiation over its history (Kokelj et al., 2005; Thienpont et al., 2013). The exact age of the thaw slump on Lake 9B is unknown but previous studies indicated the approximate timing of the most recent slump initiation to be pre-1950 (Kokelj et al., 2005; Thienpont et al., 2013). Lake 9A is a reference lake with no history of thaw slumping (Figure 3.3). Both study lakes have similar lake depths but differ based on DOC concentrations, apparent lake colour, conductivity, total phosphorus, calcium, and total dissolved solids (TDS) (Table 3.1). Consistent with previous studies on water quality in slump-impacted and reference lakes of the Mackenzie Delta uplands (Kokelj et al., 2009; Thompson et al., 2012; Houben et al., 2016), Lake 9A has higher DOC, phosphorus, and apparent colour, and lower conductivity, calcium, and TDS compared to Lake 9B (Table 3.1).

3.3.2 Field and laboratory methods

Lake sediment cores were collected from Lakes 9A and 9B in July, 2013. Lakes were accessed by an inflatable boat and sediment cores were collected near the centre of the lake using a UWITEC^{TM} gravity corer with hammer action to penetrate deep into the thick, clay-rich sediment. Sediment cores were sectioned at 0.5 cm resolution using a modified Glew (Glew, 1988) vertical extruder and stored frozen at the University of Ottawa (Canada).

Sediment core chronologies were established for Lakes 9A and 9B using ^{210}Pb and ^{137}Cs radioisotopes at the Laboratory for the Analysis of Natural and Synthetic Environmental Toxicants, at the University of Ottawa using an Ortec High Purity Germanium Gamma Spectrometer (Oak Ridge, TN, USA). Certified reference materials obtained from International Atomic Energy Association (Vienna, Austria) were used for efficiency corrections, and results were analyzed using ScienTissiME (Barry's Bay, ON, Canada). ^{137}Cs dating is used to delineate sediment deposited from the atmospheric nuclear bomb testing in 1950's and 1960's (Longmore & Duerden, 1982). ^{137}Cs is typically used to corroborate sediment ages derived from ^{210}Pb radioisotope activity. An age-depth model was developed, and sedimentation rates were inferred using the constant rate of supply (CRS) model (Appleby & Oldfield, 1978).

Chydorus and *Alona* carapaces were extracted from Lakes 9A/B sediments using methods described in Korosi & Smol (2012). Approximately 1 g of wet sediment was deflocculated in 100 mL of 10% KOH solution and heated to 70 °C on a hotplate for 20 minutes. The sediment mixture was sieved through a 64 µm mesh and rinsed with deionized water before being transferred to a beaker. Approximately 1 mL aliquots from each sub-sample were transferred to a Bogorov chamber where *Chydorus* and *Alona* carapaces were hand-picked with tweezers using a dissection microscope at 25x magnification (Figure 3.4). *Alona* carapaces were rare or absent in most sediment intervals, and so were not used for further analysis. *Chydorus* carapaces were transferred to centrifuge tubes and preserved with 3-4 drops of 10% ethanol for each sediment interval and shipped to the Nevalainen Laboratory at University of Helsinki (Finland) for absorbance measurements.

Chydorus carapace absorbance were measured at UVR wavelengths 305 nm and 340 nm using a Shimadzu UV/VIS-2401PC dual-beam spectrophotometer with an adapter, following methods from Nevalainen & Rautio (2014). Carapace absorbances were measured in triplicates for QA/QC assurance, and the standard error was calculated for each carapace. An average of the triplicate absorbance measurements was calculated for each carapace, and an average of *Chydorus* carapace absorbance values was calculated for each sediment interval.

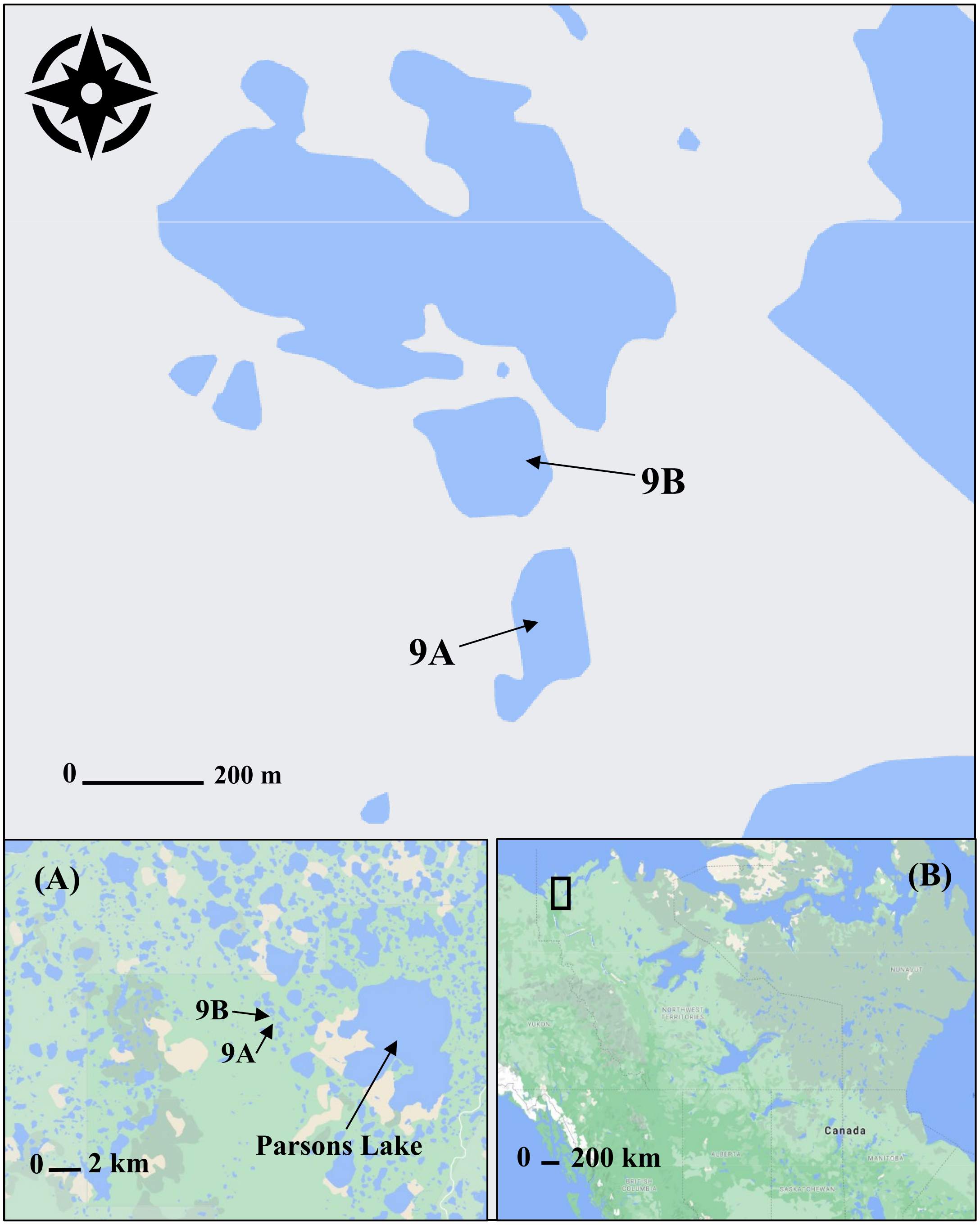

Figure 3.2: Map of study Lakes 9A and 9B in the Mackenzie Delta region (Northwest Territories, Canada). Inset (A) shows the location of Lakes 9A/B west of Parsons Lake in the Mackenzie Delta region. Inset (B) shows the location of the study region in the Northwest Territories, Canada.

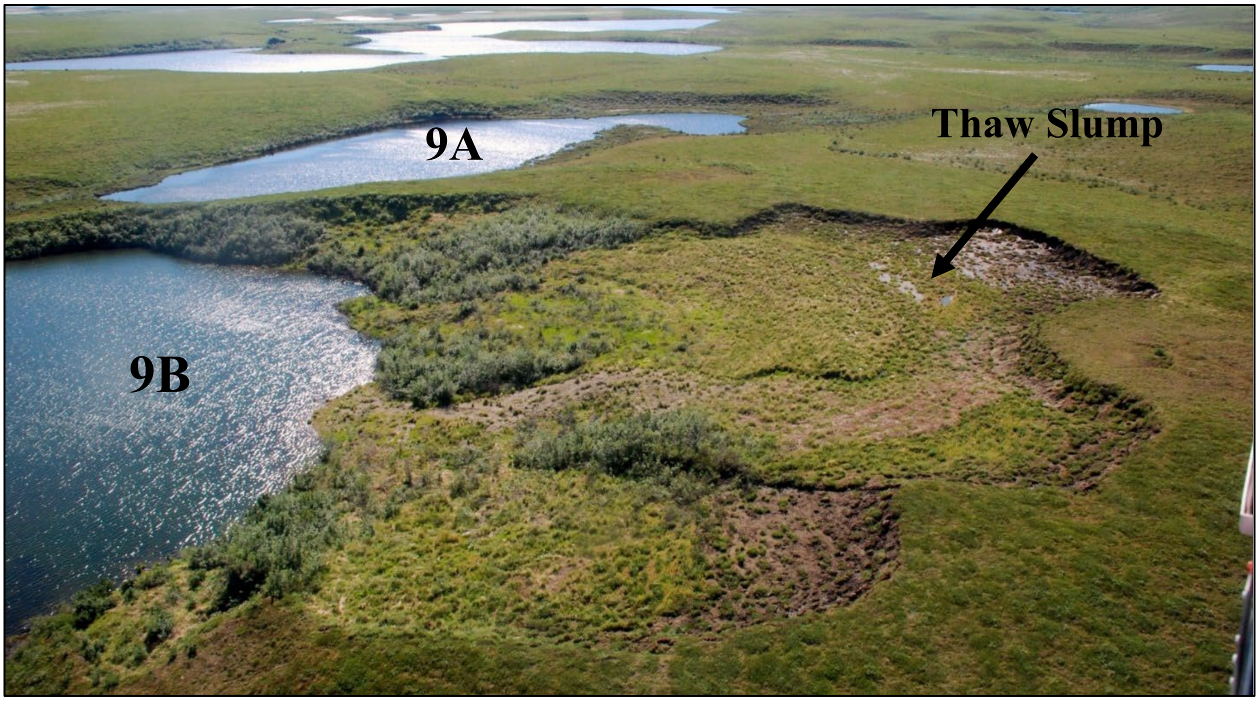

Figure 3.3: Photo of study Lakes 9A (reference) and 9B (slump-affected), located in the uplands east of the Mackenzie River Delta and north of the town of Inuvik (Northwest Territories, Canada). Photo taken by Dr. Joshua Thienpont, York University, in July 2008.

Table 3.1: Summary of lake locations, maximum water depth, and water chemistry data analyzed at the Taiga Environmental Labs in Yellowknife (a CALA accredited facility) based on surface water samples collected in July, 2017. Water chemistry variables analyzed are dissolved organic carbon (DOC), apparent colour (Colour), conductivity (Cond), filtered total phosphorus (TP), calcium (Ca), and total dissolved solids (TDS). "Avg A" and "Avg B" are the average values for reference and slump-affected lakes, respectively, from 60 lakes in the Mackenzie Delta region based on lake morphometry and water chemistry samples collected in 2017 (J. Korosi, York University, unpublished data).

Lake ID	Latitude (N)	Longitude (W)	Depth (m)	DOC (mg/L)	Colour (CU)	Cond (µS/cm)	TP (mg/L)	Ca (mg/L)	TDS (mg/L)
9A	68°58.072	133°53.875	2.5	20.3	181	44.3	0.019	3.8	55
9B	68°58.220	133°53.968	2.1	7.4	22	463.0	0.004	55.6	282
Avg A	-	-	4.1	20.7	169	97.2	0.022	10.4	90
Avg B	-	-	3.5	11.1	52	508.8	0.014	62.7	351

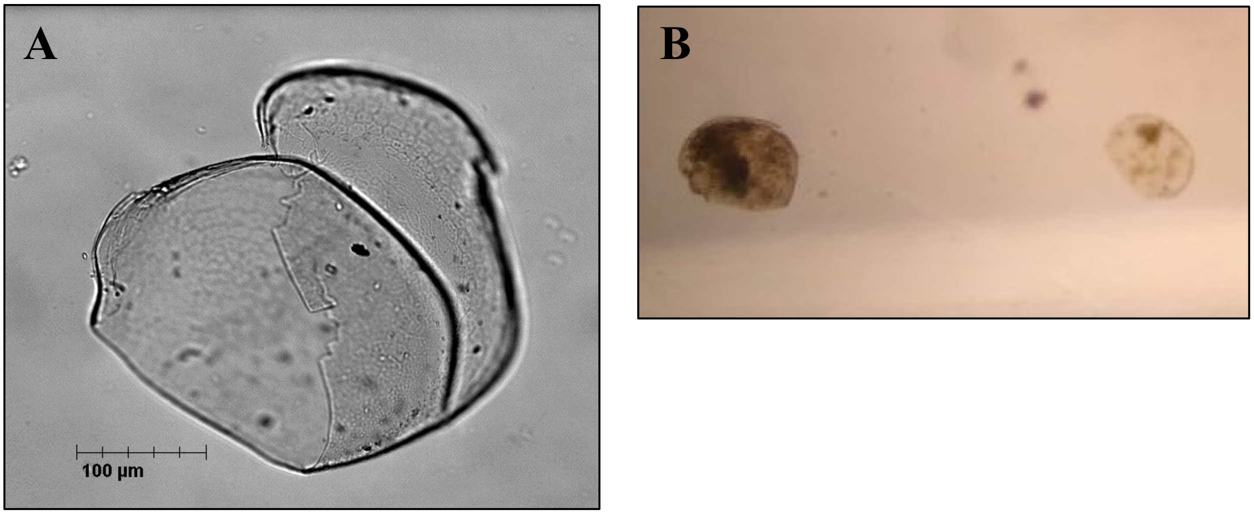

Figure 3.4: A) Photo of a *Chydorus* carapace taken using an inverted microscope at 250x magnification. Photo taken by Dr. Jennifer Korosi, York University. B) Photo of two *Chydorus* carapaces taken under a dissection microscope at 25x magnification. Note the difference in pigmentation between the two carapaces; the carapace on the left has higher pigmentation than the carapace on the right.

3.4 Results

3.4.1 Sediment core chronologies

In Lake 9A, there was a decline in ^{210}Pb isotopic activity from approximately 1000 to 150 Bq/kg, then generally declined to 0 Bq/kg at 12 cm, where background ^{210}Pb was reached (Figure 3.5). There was a peak in ^{137}Cs isotopic activity at approximately 3.25 cm indicating ~1963 in Lake 9A. The ^{137}Cs dates roughly correspond with dates produced by the CRS model for ^{210}Pb activity, where 1953 dates back at a core depth of approximately 3.5-4 cm (Figure 3.5). Based on the CRS model, 1943 ± 23 years occurred at a core depth of 4.25 cm, and below this depth the error margins of the CRS model increased substantially (Figure 3.5).

In Lake 9B, there was a decline in ^{210}Pb isotopic activity from 228 to 82.49 Bq/kg from 1 cm to 5 cm, respectively, followed by a general declined to 0 Bq/kg at 14 cm, where background ^{210}Pb was reached (Figure 3.5). There was a peak in ^{137}Cs isotopic activity at a depth of 7.25 cm indicating ~1963. This is consistent with the CRS model, where 1963 also corresponds to a core depth of approximately 7 cm. Based on the CRS model, 1927 ± 11 years occurred at a core depth of 10.25 cm, and below this depth the error margins of the CRS model increased substantially (Figure 3.5).

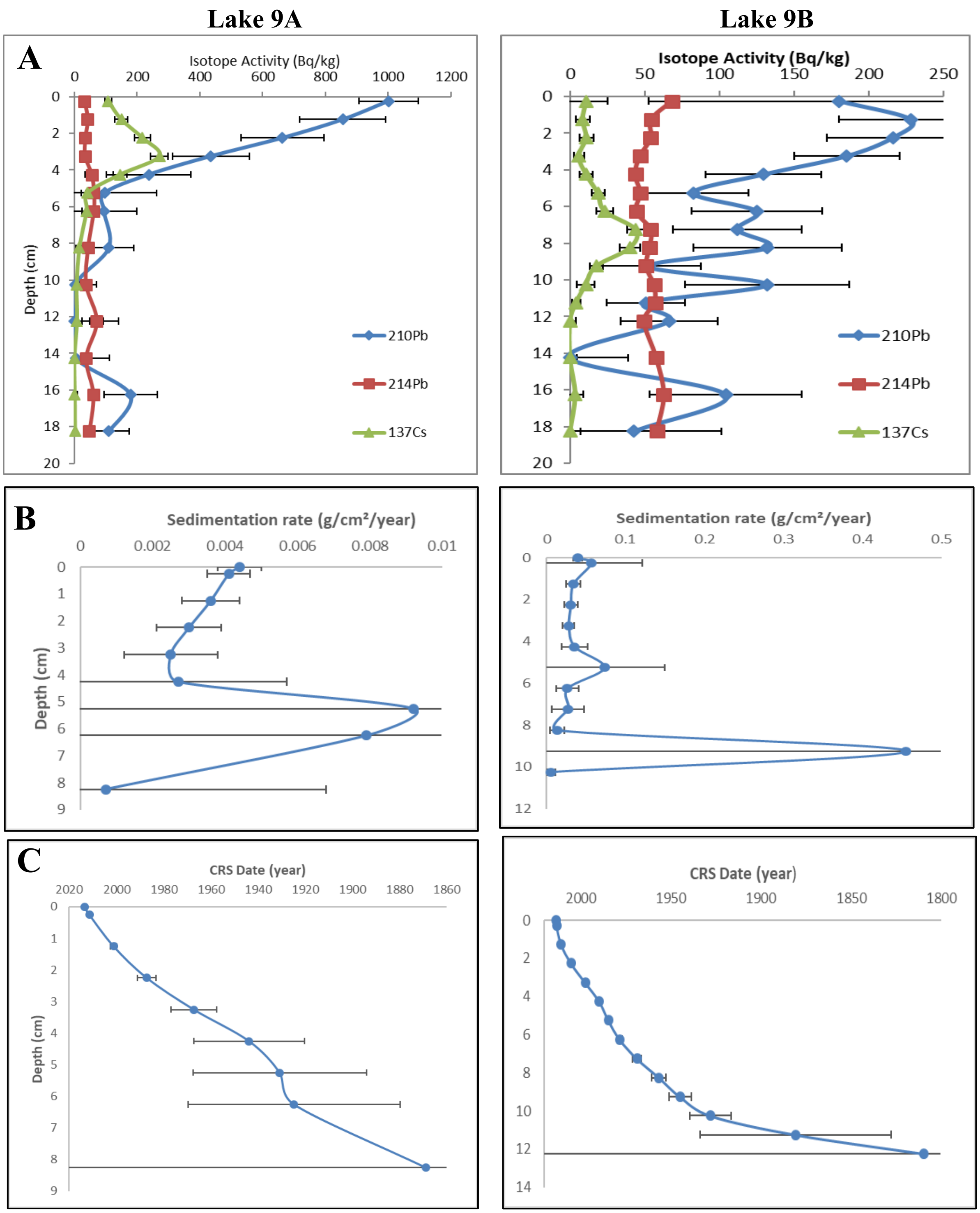

Figure 3.5: The results of ^{210}Pb dating for Lakes 9A and 9B. (A) Radioisotopic activities for ^{210}Pb, ^{214}Pb, and ^{137}Cs. (B) The sedimentation rate based on ^{210}Pb dating CRS model. (C) Sediment core depth versus date based on the CRS model.

3.4.2 Cladoceran carapace absorbances

In Lake 9A, the maximum average absorbances of individual carapaces are 2.68 at 305 nm and 2.26 at 340 nm at a depth of 1.25 cm (Figure 3.6). The minimum average absorbances of individual carapaces are 0.41 at 305 nm and 0.38 at 340 nm at a depth of 10.25 cm (Figure 3.6). The standard error for individual carapaces (based on triplicate measures) ranged from 0.001-0.375 at 305 nm and 0.002-0.123 at 340 nm (Table 3.2). The average *Chydorus* carapace absorbance values for sediment intervals ranged between 0.4-2.6 at 305 nm and 0.3-2.5 at 340 nm (Figure 3.7). The average *Chydorus* carapace absorbance values generally increased from ~1943 to ~2000, followed by a decrease in average absorbances from ~2000 to 2013 (Figure 3.7).

In Lake 9B, the maximum average absorbances of individual carapaces are 2.79 at 305 nm and 2.32 at 340 nm, both at a depth of 4.25 cm (Figure 3.8). The minimum average absorbances of individual carapaces are 0.24 at 305 nm and 0.23 at 340 nm, both at a depth of 12.25 cm (Figure 3.8). The standard error for individual carapaces (based on triplicate measures) ranged from 0.002-0.411 at 305 nm and 0.002-0.496 at 340 nm (Table 3.2). The average *Chydorus* carapace absorbance values for sediment intervals ranged from 0.2-2.8 at 305 nm and 0.2-2.3 at 340 nm (Figure 3.7). Only one carapace was recovered in the 10.25 cm interval, with an average absorbance values of 1.4 and 1.3 at 305 nm and 340 nm, respectively. Most carapace absorbance values remained ≥ 1 at both 305 and 340 nm wavelengths throughout the sediment core, except at the depth of 12.25 cm (Figure 3.7). Average carapace absorbance values fluctuated throughout time; average carapace absorbance values decreased from ~1927 to ~1975, followed by an increase from ~1975 to ~1990. From ~1990 to 2013, the average carapace absorbance values decreased again (Figure 3.7).

The carapace absorbances in both lakes were higher in the 305 nm (UVB range; 280-315 nm) wavelength than the 340 nm (UVA range; 315-400 nm) wavelength, consistent with other studies which documented lower carapace absorbances in the UVA range (340 nm) and higher carapace absorbances in the UVB range (305 nm) (Rautio & Nevalainen, 2013; Nevalainen & Rautio, 2014). The higher carapace absorbances in the UVB range than the UVA range, and higher absorbance in the UVR range (280-400 nm) in general, is also consistent with the melanin absorption characteristics from other studies (Rautio & Nevalainen, 2013; Nevalainen & Rautio, 2014).

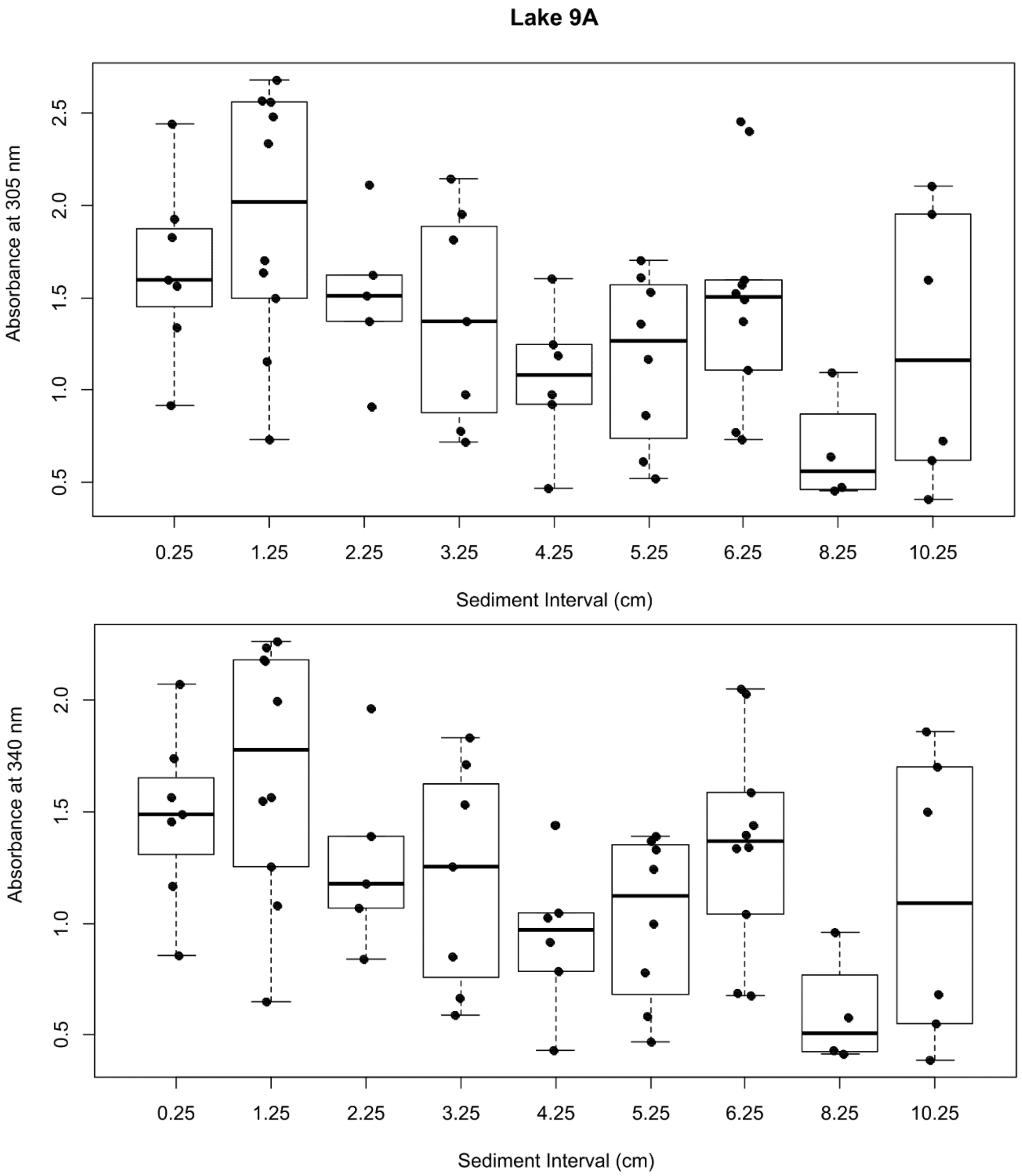

Figure 3.6: Boxplots comparing the average of the triplicate absorbance measures for each carapace at 305 nm and 340 nm wavelengths for each sediment interval for Lake 9A.

Table 3.2: Summary of triplicate *Chydorus* carapace absorbances for Lakes 9A and 9B. The ranges of triplicate absorbance measurements, the ranges of standard errors of each set of triplicate absorbance measurements, and number of carapaces extracted for absorbance measurements from each sediment interval. “(305)” and “(340)” indicate the wavelengths (nm) the absorbance measurements were taken at.

	Ranges of triplicate absorbances				**Ranges of standard errors**				**Number of carapaces**	
Interval (cm)	**9A (305)**	**9A (340)**	**9B (305)**	**9B (340)**	**9A (305)**	**9A (340)**	**9B (305)**	**9B (340)**	**9A**	**9B**
0.25	0.9-2.5	0.8-2.1	0.5-2.5	0.4-2.1	0.001-0.089	0.003-0.067	0.002-0.229	0.004-0.276	7	6
1.25	0.6-2.7	0.6-2.3	-	-	0.015-0.265	0.015-0.123	-	-	10	-
2.25	0.9-2.2	0.8-1.9	0.6-2	0.5-1.8	0.013-0.375	0.009-0.121	0.006-0.133	0.004-0.110	5	4
3.25	0.6-2.2	0.5-1.9	-	-	0.027-0.091	0.017-0.074	-	-	7	-
4.25	0.4-1.7	0.4-1.5	0.7-2.9	0.6-2.4	0.007-0.185	0.005-0.105	0.012-0.289	0.002-0.112	6	10
5.25	0.5-1.8	0.4-1.5	-	-	0.003-0.118	0.004-0.094	-	-	8	-
6.25	0.7-2.6	0.6-2.2	0.6-1.7	0.6-1.4	0.009-0.160	0.006-0.092	0.005-0.143	0.003-0.056	10	6
8.25	0.4-1.1	0.3-1	0.5-2.7	0.4-2.5	0.002-0.019	0.002-0.039	0.004-0.411	0.002-0.383	4	5
10.25	0.4-2.1	0.3-1.9	1.4-1.5	1.3-1.4	0.008-0.061	0.009-0.070	0.029	0.019	6	1
12.25	-	-	0.2-1.2	0.2-0.9	-	-	0.007-0.075	0.003-0.083	-	3
14.25	-	-	0.5-1.8	0.5-1.6	-	-	0.003-0.162	0.007-0.496	-	6
16.25	-	-	0.6-2.5	0.6-2.2	-	-	0.002-0.348	0.002-0.332	-	10
18.25	-	-	0.3-2.4	0.3-2	-	-	0.018-0.286	0.011-0.064	-	10
20.25	-	-	0.7-1.9	0.6-1.8	-	-	0.005-0.133	0.016-0.132	-	10

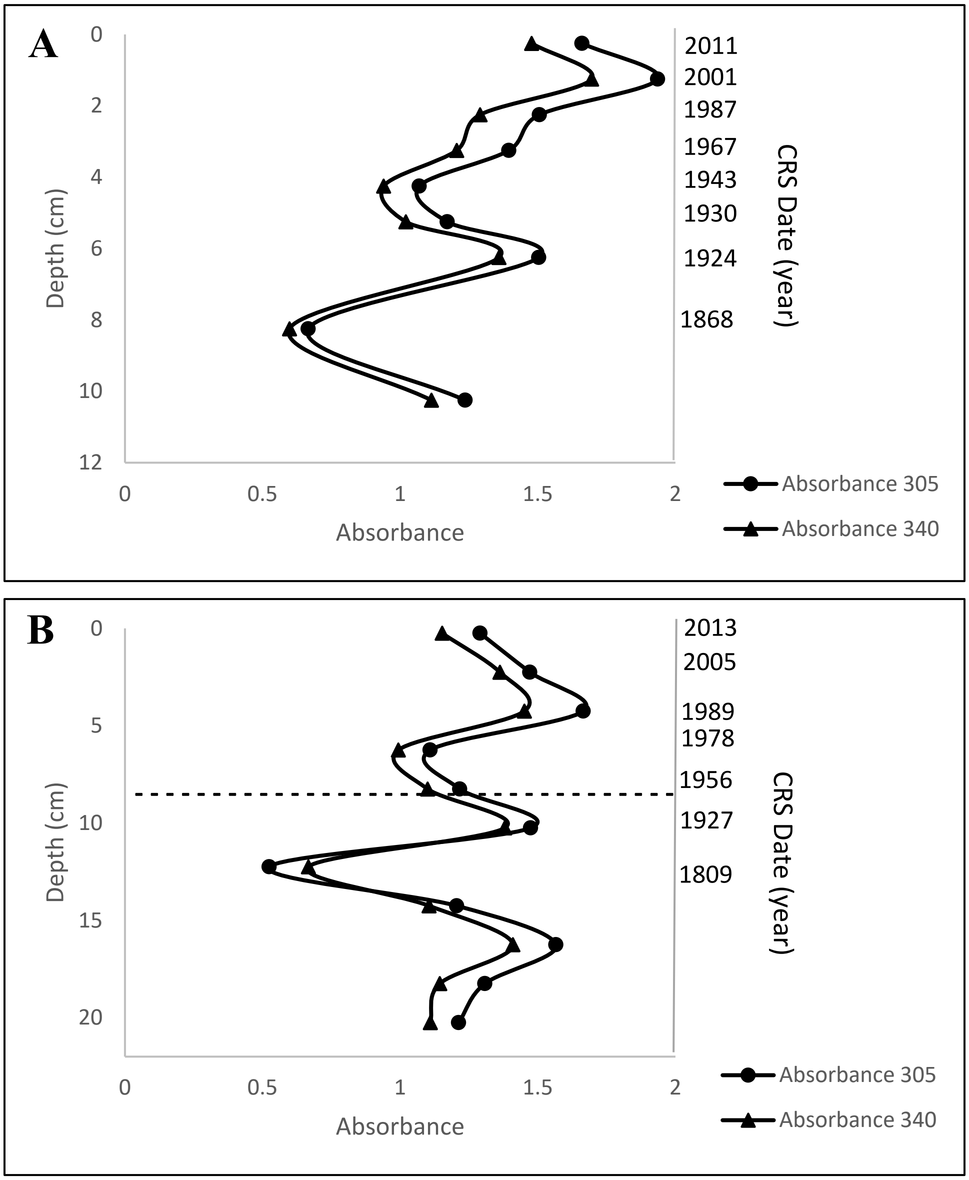

Figure 3.7: The average *Chydorus* carapace absorbance values for 305 nm and 340 nm wavelengths with sediment core depth for Lakes 9A (A) and 9B (B). For Lake 9B, horizontal dashed line represents known approximate timing of slump initiation (pre-1950) (Thienpont et al., 2013).

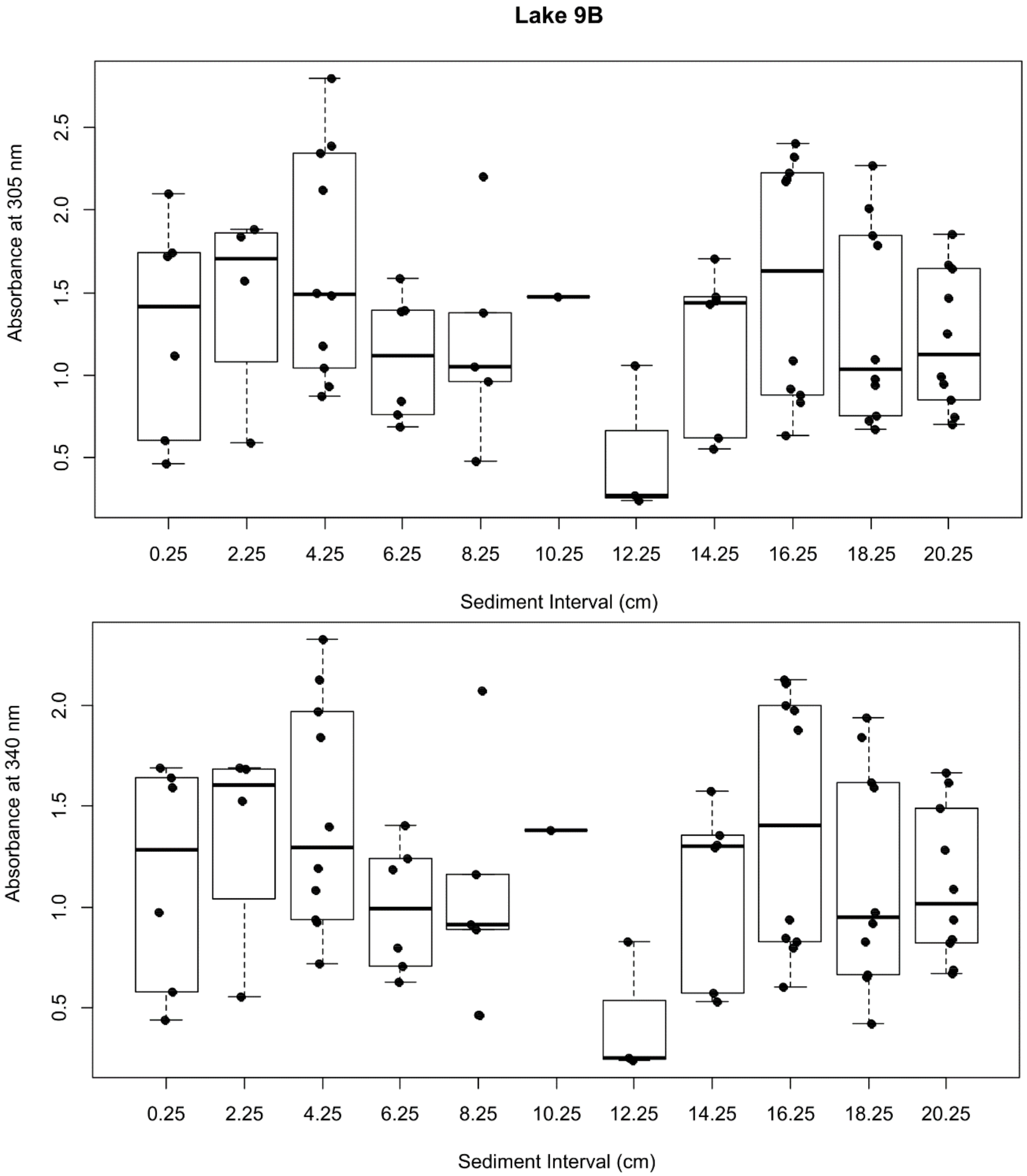

Figure 3.8: Boxplots comparing the average of the triplicate absorbance measures for each carapace at 305 nm and 340 nm wavelengths for each sediment interval for Lake 9B.

3.5 Discussion

The aim of the study was to measure the absorbances of carapaces at UVR wavelengths to infer if thaw slumping increases the risk of exposure of cladocerans to potentially harmfully UV radiation in small, shallow western Arctic lakes of the Mackenzie Delta region. The increase in water transparency that occurs as a result of shoreline thaw slumping may increase the exposure of aquatic biota to UV radiation (Rautio & Nevalainen, 2013; Nevalainen & Rautio, 2014). The high range of variability of the standard errors for individual carapaces is likely due to some small carapaces (< 300 µm) that did not fully cover the shutter of the adapter on the Shimadzu UV/VIS-2401PC dual-beam spectrophotometer, and thus, the interpretations of the carapace absorbances in Lakes 9A and 9B are done with caution.

DOC concentrations in lake waters are a result of allochthonous inputs from terrestrial catchments and autochthonous sources produced by algae and macrophytes (Kellerman et al., 2020), combined with in-lake processes such as microbial respiration, photolytic processes, and flocculation that processes DOC in the water column (Molot & Dillon, 1997; Cory et al., 2007). Chromophoric (i.e. coloured) DOC compounds in lakes attenuates UV radiation in the water column, limiting the depth to which UV radiation can penetrate in the water column (Nevalainen et al., 2015). Drivers such as solar radiation, catchment disturbance, and lake productivity, are all known to alter DOC concentrations in Arctic lakes (Vincent et al., 2007; Cory et al., 2014; Nevalainen & Rautio, 2014).

3.5.1 Are Chydorus *in Lake 9B exposed to high levels of UV radiation?*

The range of carapace absorbances for Lake 9B (slump-affected) are comparable to the range of the carapace absorbances in Lake 9A (reference), despite large differences in DOC and apparent colour. In addition, the average *Chydorus* carapace absorbance values in Lakes 9A and

9B were mostly between 1 and 1.5 at both 305 and 340 nm wavelengths throughout the entire cores (Figure 3.7). Similar carapace absorbance measurements have been reported in a subarctic boreal lake with high DOC (approximately 1.5 at 305 nm and 1.2-1.3 at 340 nm) and in an alpine lake with a highly transparent water column (approximately 1.8-2.4 at 305 nm and 1.5-2 at 340 nm) (Nevalainen & Rautio, 2014). Some of the carapace absorbance measurements in the alpine lake were higher than some of the carapace absorbances in Lakes 9A and 9B. Collectively, this indicates *Chydorus* in Lake 9B are not being exposed to higher levels of UV radiation as a result of slump-induced increases in water clarity.

Slump-affected lakes have higher macrophyte biomass as a result of changes in sediment chemistry and increases in light penetration to the benthic zone (Mesquita et al., 2010). *Chydorus* commonly live associated with benthic and littoral substrates, although they exhibit weak swimming ability (Frey, 1988; Hann, 1989; Thienpont et al., 2015). *Chydorus* in Lake 9B may live amongst the dense macrophyte beds, which can provide shading from UV radiation (Gareis et al., 2010). Alternatively, DOC concentrations may have remained high enough to prevent UV exposure in cladocerans. Previous studies have suggested a DOC concentration below 5 mg/L UV radiation is known to negatively impact aquatic biota (Williamson et al., 1996; Häder et al., 2007). The DOC concentration in Lake 9B is 7.4 mg/L (Table 3.1) which likely includes non-chromophoric DOC sources, whereas chromophoric DOM is the main UV-absorbing constituent (Gareis et al., 2010).

3.5.2 How has Chydorus *exposure to UV radiation changed over time?*

In Lake 9A, the average *Chydorus* carapace absorbance values increased from ~1943 to ~2000 (1.4 to 1.9 at 305 nm; 1.2 to 1.7 at 340 nm), which indicated exposure to UV radiation increased over this time period. The subsequent decrease in carapace absorbance values from

~2000 to 2013 (1.9 to 1.6 at 305 nm; 1.7 to 1.5 at 340 nm), indicated a slight decrease in exposure to UV radiation. Long-term trends of diatom assemblages from Lake 9A showed a decrease in the relative abundance of benthic *Fragilaria* spp. from ~1970 to 2008, a response likely due to increased air temperatures and shorter durations of lake ice cover (Thienpont et al., 2013). Additionally, there was an increase of chrysophyte scales relative to diatom valves from ~1990 to 2008, a response that may be due to intensifying thermal stratification as a result of climate warming (Thienpont et al., 2013). Lastly, there was no increase in the relative abundance of planktonic diatoms throughout time (Thienpont et al., 2013). Shorter durations of lake ice cover may increase the exposure of *Chydorus* to UV radiation which may partially explain the increase in carapace absorbance values from ~1970 to ~2000. However, shorter durations of lake ice cover cannot explain the carapace absorbance values fluctuations between ~1943 and ~1970 because the increase in carapace absorbance values occurred before the rapid increase of local air temperatures starting 1970 (Thienpont et al., 2013), and also between ~2001 and 2013 because carapace absorbance values decreased.

The increase in the average *Chydorus* carapace absorbance values may be related to changes in chromophoric DOM concentrations, lake shallowness, and near-surface habitats of *Chydorus* that may increase exposure to UV radiation despite the high DOC concentration in Lake 9A (Table 3.1). For example, a survey of UV radiation in thermokarst lakes of the Mackenzie Delta region showed approximately 19% and 31% of the water columns were exposed to UVB and UVA radiation, respectively, despite high chromophoric DOM concentrations (Gareis et al., 2010). Furthermore, small shifts in chromophoric DOM can increase the depth UV radiation penetrates in the water column (Forsström et al., 2015), and thus may increase the exposure of *Chydorus* to UV radiation. Alternatively, there may have been a shift in habitat usage by *Chydorus* occupying shallower habitats throughout time that may expose *Chydorus* to UV radiation near the surface. Similarly, Nevalainen et al. (2015) suggested the increased UV exposure of *Alona affinis* carapaces (high degree of melanisation) in two of their study lakes was probably due to lake shallowness (1.4 and 1.6 m) and near-surface habitats of *Alona affinis* despite the high UV-attenuating properties (K_{d305} = 22.3 and 27.6 m; K_{d340} = 15.2 m).

In contrast to Lake 9A, no directional change in UV exposure of *Chydorus* was apparent in Lake 9B, and average carapace absorbance values generally fluctuated between 1 and 1.6 (Figure 3.7). The changes in UV exposure of *Chydorus* could potentially be related to polycyclic slump activity and associated limnological changes. The thaw slump(s) on Lake 9B exhibits polycyclic slump behaviour which periodically increases inorganic sediments to the lake. Thienpont et al. (2013) observed a general increase in the relative abundance of non-fragilarioid periphytic diatom taxa from early 1950's to 2008, a biological response probably due to polycyclic slump activity that corresponds well with the known recent timing of slump initiation. In addition, the slump was active in 2004 (Kokelj et al., 2005) but had recently stabilized in 2008 (Thienpont et al., 2013). UV exposure of *Chydorus* decreased from ~2005 to 2013, which may suggest carapace absorbances does not reflect polycyclic thaw slump behaviour. Overall, the fluctuating *Chydorus* carapace absorbance values between ~1927 and 2008 may be the interaction of complex ecological and limnological processes that prevent a clear association of UV absorbances with polycyclic slump activity.

3.5.3 Conclusions and future directions

The average carapace absorbances were similar between Lakes 9A and 9B (Figure 3.8) despite large differences between DOC and lake colour (Table 3.1), which indicates that thaw slumping did not increase exposure of *Chydorus* to UV radiation. I suggest that the development of dense macrophyte beds in response to thaw slumping (Mesquita et al., 2010) may provide shading for aquatic biota and mitigate increased UV exposure that would result from the decrease in chromophoric DOC (Gareis et al., 2010). In contrast to my initial hypothesis, carapace absorbances in Lake 9B did not increase over time in response to increased slump activity but did increase over time in the reference lake (Lake 9A). The mechanisms behind the temporal changes in *Chydorus* carapace absorbances through time in Lakes 9A and 9B may be the result of interacting limnological and ecological processes that reflect changes in cloud cover, water clarity, ice cover, littoral and near-surface habitat usage by *Chydorus*.

My study provides some initial insight into the temporal dynamics of exposure of biota to UV radiation in a slump-affected lake and undisturbed lake. Further investigation of UV radiation and thaw slumping and its potential ecological impacts is needed. To further investigate long-term changes in UV exposure using the paired-lake study design, *Chydorus* and *Alona* carapaces should be extracted and absorbances measured from more study lakes because there may be long-term differences between slump-affected lakes with different DOC concentrations. For example, unpublished data from my Honour's Thesis showed Lake 12B, a heavily slump-affected lake, to have a DOC concentration of 4.3 mg/L, which may be below the DOC threshold (~5 mg/L) (Williamson et al., 1996; Häder et al., 2007) and allow UV radiation to penetrate deeper in the water column. Also, measuring the diffuse attenuation coefficients (K_d) and absorptivity of DOC at 305 and 340 nm wavelengths between slump-affected and reference lakes would provide insight into the quality of DOC and the depths UV radiation is able to penetrate the water column. In addition, collecting zooplankton (i.e. *Daphnia*, *Bosmina*, copepods, and rotifers) from slump-affected and reference lakes and analyzing other UV protective pigments and photoprotectants such as carotenoids and mycosporine-like amino acids, would provide insight into factors (i.e. DOC and lake depth) that may be important for UV exposure of zooplankton in the open-water zone.

CHAPTER 4: A PALEOLIMNOLOGICAL APPROACH TO INVESTIGATING *DAPHNIA* PRESENCE/ABSENCE IN SUB-ARCTIC LAKES IN THE MACKENZIE DELTA REGION

4.1 Abstract

In July 2017, a survey of zooplankton in surface waters from lakes in the Mackenzie Delta uplands (Northwest Territories, Canada) observed that *Daphnia* were absent in slump-affected lakes but relatively common in unimpacted lakes. The objective of this study was to use paleolimnological techniques to examine *Daphnia* presence/absence in a subset of 14 lakes included in the 2017 survey inferred from subfossil *Daphnia* ephippia. *Daphnia* ephippia and *Chaoborus* mandibles were extracted from the top (recent) and bottom (pre-1850) sediment intervals of 14 lakes. *Chaoborus* mandibles were identified because they are predators on *Daphnia* and can also indicate fish presence/absence, providing useful biogeographic information on biotic controls on *Daphnia* in Mackenzie Delta region lakes. *Daphnia* presence/absence in the study lakes were mostly consistent with the 2017 zooplankton survey. *Daphnia* ephippia were not encountered in Lakes 14A and 5B, despite *Daphnia* being present in the 2017 zooplankton survey; this likely indicates higher volume of sediment needs to be analyzed to confirm *Daphnia* absence. *Daphnia* ephippia were encountered in the sediments in Lake 5A, but not in the 2017 zooplankton survey. *Chaoborus trivittatus* mandibles were detected in Lakes 5A, 14A, and 20A. In the 2017 zooplankton survey, *C. trivittatus* was present in Lakes 5A and 14A, and *C. americanus* individuals were present in Lakes 5B and 20A. Based on the sediment data and zooplankton survey, *Daphnia* and *Chaoborus* coexist in Lakes 5A/B, 14A, and 20A. This information can assist with study lake selection for future studies examining the ecological impacts of thaw slumping. For example, Lakes 5A/B, 14A/B, and 20A/B can be

selected for the reconstruction of *Daphnia* and *Chaoborus* assemblages to elucidate the impact of retrogressive thaw slumping on *Daphnia* and *Chaoborus* throughout time.

4.2 Introduction

Permafrost thaw is widespread throughout the Arctic region and is manifested in various forms. Retrogressive thaw slumping is one form of permafrost thaw that is a common geomorphic disturbance that forms along coast lines, river shorelines, and lake shorelines across the western Arctic and northern Siberia (Burn & Lewkowicz, 1990; Kokelj et al., 2017) (Figure 4.1). Retrogressive thaw slumping has been increasing in frequency and intensity the past 50 years as a result of increasing air temperatures due to climate warming (Lantz & Kokelj, 2008). This has implications for water quality in Arctic lake and stream ecosystems (Kokelj et al., 2009; Thienpont et al., 2013).

Lakes with shoreline thaw slumps in the Mackenzie Delta region (Northwest Territories, Canada) exhibit higher conductivity and alkalinity, and lower chromophoric DOC and nutrient concentrations (Kokelj et al., 2009; Thompson et al., 2012; Houben et al., 2016). The input of siliclastic slump material into adjacent lakes binds to and scours DOC-nutrient complexes from the water column, which is then deposited to the bottom lake sediment, resulting in a highly transparent water column and nutrient-poor lake conditions (Thompson et al., 2008; Houben et al., 2016). Slump-affected lakes also exhibit lower chlorophyll α concentrations, suggesting low nutrient concentrations results in decreased primary production in slump-affect lakes compared to undisturbed lakes (Thompson et al., 2008; Thompson et al., 2012; Houben et al., 2016). Macrophyte biomass is typically higher in slump-affected lakes than undisturbed lakes due to increased exposure to photosynthetically active radiation (PAR) via higher water transparency, and this may promote a complex benthic habitat (Mesquita et al., 2010). For example, one study reported slump-affected lakes to have higher abundances of ostracods and nematodes, and lower abundances of chironomids than undisturbed lakes (Moquin et al., 2014). The differences in macroinvertebrate abundance and community composition were attributed to differences in sediment chemistry and macrophyte biomass.

The implications of thaw slumping on zooplankton communities and higher trophic levels have not been well-documented, despite high relevance for the health of Arctic lake food webs. My undergraduate honour's thesis (York University Department of Geography, 2017; unpublished data) surveyed zooplankton in lakes in the uplands east of the Mackenzie Delta in 2017 and compared zooplankton assemblages between slump-affected lakes and undisturbed lakes. I found that *Daphnia* were mostly absent in slump-affected lakes, suggesting *Daphnia* may be particularly sensitive to the limnological changes associated with thaw slumping, but further investigation is still needed since my zooplankton surveys were based on a single horizontal net tow from each study lake. *Daphnia* exhibit seasonality and may have been abundant prior to (and/or following) sampling, and may also have been living amongst macrophytes that are not easily sampled, thus it is possible that *Daphnia* absence was overestimated.

Figure 4.1: Photo of a shoreline retrogressive thaw slump on a small lake located in the uplands east of the Mackenzie Delta, and north of the town Inuvik, Northwest Territories. Photo taken in July 2017.

Daphnia (Daphniidae) are large-bodied pelagic zooplankton taxa that are commonly used as a model organism in ecology and evolutionary biology. *Daphnia* are considered keystone species in lake ecosystems because of their role in energy transfer in lake food webs (Korhola & Rautio, 2001; Stollewerk, 2010). They occupy an intermediate role in the food web, grazing upon algae and detritus, and in turn serve as an important food source to facilitate energy transfer to higher trophic organisms such as planktivorous fish, macroinvertebrates, and large-bodied zooplankton (Lindsey et al. 1981; Swadling et al. 2000). *Daphnia* are influenced by biotic and abiotic factors, such as primary production, fish predation, nutrient concentrations, and water quality (i.e. conductivity and pH) (Korhola & Rautio, 2001). For example, if algal production decreased this can act as bottom-up control on *Daphnia* by limiting their growth (DeMott & Kerfoot, 1982; Hessen & Alstad Rukke, 2000). *Daphnia* often inhabit lakes with low conductivity and major ions concentrations because high conductivity may increase mortality rates (Sarma et al., 2006; Bogart et al., 2016). Consequently, water quality variables known to be affected by permafrost thaw slumping, such as increased conductivity, decreased DOC, and decreased nutrients, are important for structuring *Daphnia* which, in turn, may negatively affect *Daphnia* abundances and may cascade through the food web.

The purpose of this study is to corroborate the findings of the previous 2017 zooplankton surveys using paleolimnological techniques. Contemporary sampling methods using one-time plankton net tows can miss zooplankton during sampling and may result in an underrepresentation of their importance in zooplankton assemblages (Korhola & Rautio, 2001). Lake sediment located in deeper parts in lakes incorporate cladocerans from offshore (littoral zone) as a result of passive transport (i.e. water currents generated by wind) to deep parts of the lake that mixes with limnetic taxa (i.e. open-water cladocerans) before being deposited to the lake sediment (Korhola & Rautio, 2001). For example, Frey (1960) found six cladoceran species present in lake sediments that were not identified in contemporary sampling. *Daphnia* shift to the production of diapausing eggs (eggs that undergo dormancy) encapsulated in the ephippia, a chitinous shell that preserves well in lake sediments, when environmental conditions deteriorate (Schultz, 1977). Because of this, *Daphnia* ephippia in lake sediments are easily identifiable, allowing for their use in paleolimnological studies.

The objective of my study is to investigate *Daphnia* presence/absence in 14 lakes of the Mackenzie Delta uplands region that span a gradient in thaw slump impact, including many of the lakes originally sampled for zooplankton in 2017. Specifically, my aim is to determine if *Daphnia* presence could be confirmed in any additional lakes where they were not present in the zooplankton net tow. To accomplish this, I used a "top-bottom" approach that examines the top (recently deposited) and bottom (deposited ~1850) sediment layers (Cumming et al., 1992). For ten of the fourteen study lakes, I had to use archived sediment cores collected in 2013 because I was unable to complete planned fieldwork activities in April 2020 as a result of the COVID-19 pandemic. As a result, sediment availability was limited to only 2 g per interval and estimates of *Daphnia* absence based on ephippia remains are likely overestimated as a result of the low sediment availability. Where encountered, I noted and identified *Chaoborus* (phantom midges that are predators on *Daphnia*) mandibles in lake sediments to infer predator-prey relationships and fish status. In particular, *Chaoborus americanus* does not coexist with fish (unlike other chaoborid species) due to lack of diel vertical migration behaviour that makes *C. americanus* easily preyed upon by fish (von Ende, 1979). Thus, fish absence can be inferred if *C. americanus* mandibles are present in lake sediment intervals (Uutala, 1990).

4.3 Methods

4.3.1 Study site description

I chose lakes in the uplands east of the Mackenzie Delta (Northwest Territories, Canada) that have been extensively studied in past investigations into the limnological effects of thaw slumping (Kokelj et al., 2009; Thompson et al., 2012; Houben et al., 2016). Lakes are located near Noel Lake (68°31'37.16"N, 133°34'5.39"W) and Parsons Lake (68°58'28.40"N, 133°37'58.58"W), and span a transition from boreal forest to low-shrub tundra (Figure 4.2) (Lantz et al., 2010). They are categorized based on the presence or absence of shoreline thaw slumps. The “A” lakes are undisturbed, with no evidence of shoreline thaw slumping. “B” lakes are slump-affected but differ in the magnitude of slump impact (the percent of catchment impacted by slumping), and whether slumps are active or stabilized (Table 4.1). Lakes are all located within the continuous permafrost zone that typically has thick (>100 m) ice-rich permafrost (Burn & Kokelj, 2009). All lakes were included in the 2017 zooplankton survey with the exception of Lake 15A and Lake 15B. I observed that Lake 12A had a new slump forming on it during sampling in 2017, indicating it is no longer a reference lake. However, water chemistry analysis showed that the lake was not chemically impacted, and lake water chemistry was similar to that of an undisturbed lake (Table 4.2).

In July 2017, surface water samples (~0-1m depth) were collected from all study lakes except Lake 15A and Lake 15B using a vertical Van Dorn water sampler, and shipped to Taiga Environmental Labs in Yellowknife, a CALA accredited facility, for water chemistry analysis. Zooplankton were collected from a water depth of 0-2 m using horizontal net tows (net size of 63µm). Zooplankton were identified using a Sedgewick chamber under an inverted microscope at 200x magnification, and the entire chamber was analyzed. A minimum of 300 individuals were counted per sample. Zooplankton were identified to the species or genus using Balcer et al., (1984) and Haney et al., (2013) as taxonomic guides. Based on the results of this survey, *Daphnia* are present in Lakes 5B, 9A, 10A, 14A, 20A, and 35B, and absent in Lakes 5A, 9B, 10B, 12A/B, and 14B.

4.3.2 Field and laboratory methods

I collected sediment cores from Lakes 10A, 10B, 20A, and 35B in July, 2019. Sediment cores from Lakes 5A/B, 9A/B, 12A/B, 14A/B, and 15A/B were collected by the Blais Lab at the University of Ottawa in July 2013 (Table 3.1). I had intended to revisit the lakes in April, 2020, to collect larger volumes of surface sediment, but fieldwork was cancelled due to COVID-19. Our remote study lakes were accessed by helicopter. Sediment cores were collected from approximately the centre of the lake (from an inflatable boat) using a UWITEC^{TM} gravity corer. Lake sediment cores were sectioned at 0.5 cm intervals using a Glew-style (Glew, 1988) vertical extruder.

Sediment core chronologies had previously been established for the lakes cored in 2013 (5A/B, 9A/B, 12A/B, 14A/B, and 15A/B). Chronologies were established using an Ortec High Purity Germanium Gamma Spectrometer (Oak Ridge, TN, USA) to measure ^{210}Pb and ^{137}Cs radioisotopic activities, and the constant rate of supply (CRS) model was applied (Appleby & Oldfield, 1978). ^{210}Pb and ^{137}Cs dating was completed at the Laboratory for the Analysis of Natural and Synthetic Environmental Toxicants, at the University of Ottawa, Canada. The bottom intervals were chosen to correspond to ~1850, which is roughly consistent with the depth at which background levels of ^{210}Pb are reached in the core (Appleby & Oldfield, 1978). No ^{210}Pb dating was available for the four lakes cored in 2019, and so bottom intervals were selected based on the range of ^{210}Pb depths based on the 2013 cores and previous paleolimnological studies in the Mackenzie Delta uplands region (Eickmeyer, 2013; Thienpont et al., 2013).

Daphnia ephippia and *Chaoborus* mandibles were extracted from lake sediments following methods from Walker (2001) (Figure 4.3). Approximately 2 g of wet sediment from the ‘top’ (0-2 cm) and ‘bottom’ intervals was used for the lakes cored in 2013 (5A/B, 9A/B, 12A/B, 14A/B, and 15A/B) (Table 4.3). This is a lower volume of sediment than what is ideal, but I was limited by sediment availability. For Lakes 10A/B, 20A, and 35B, I used approximately 6 g of wet sediment from the ‘top’ (0.5-1 cm) and ‘bottom’ intervals (Table 4.3). The volume of lake sediment for detecting *Daphnia* ephippia and *Chaoborus* mandibles vary. For example, anywhere from 1-10 g of sediment may be required to encounter an appropriate number of *Daphnia* ephippia for analysis (Jeppesen et al., 2002; Szeroczyńska & Sarmaja-Korjonen, 2007; Nevalainen et al., 2013). For *Chaoborus* mandibles, anywhere from 5-52 g of sediment may be required to meet the minimum count recommendation of 10 individuals for assemblage characterization (Quinlan & Smol, 2001, 2010).

Lake sediment sub-samples were deflocculated in 100 mL of 10% KOH solution on a 70 ºC hotplate for 20 minutes then sieved through an 80 µm mesh, rinsed with deionized water, and transferred to a beaker. The entire sediment sub-sample was analyzed in ~1 mL aliquots in a Bogorov chamber under a dissection microscope at 25x magnification. *Daphnia* ephippia and *Chaoborus* mandibles were hand-picked with tweezers, transferred to separate glass slides and mounted with Entellan. *Chaoborus* mandibles were identified on the slide using an inverted microscope at 100x magnification following Uutala (1990) as a species identification guide.

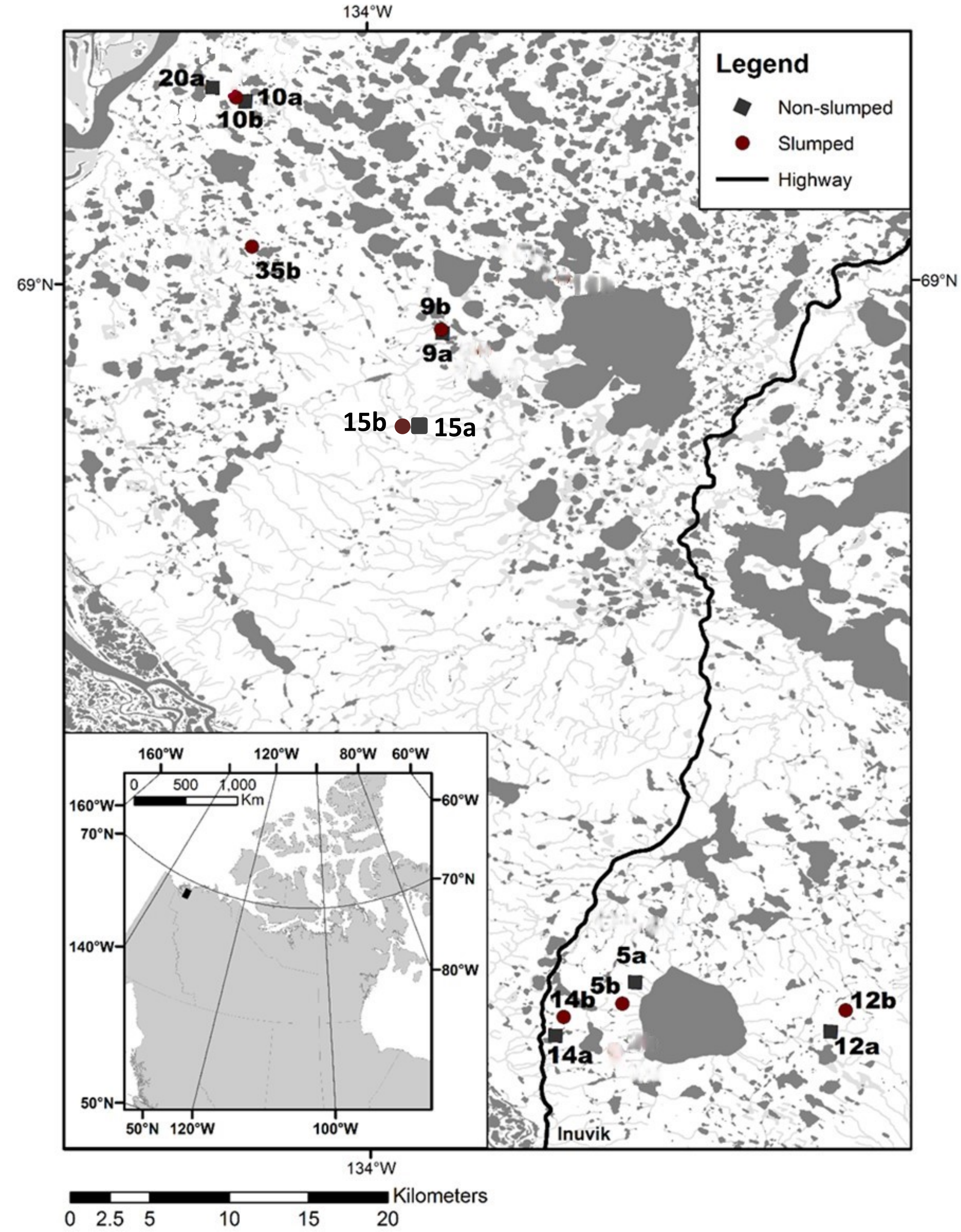

Figure 4.2: Map of study lakes in the Mackenzie Delta region, and north of Inuvik (NWT, Canada). "A" lakes are reference lakes and "B" lakes are slump-affected lakes.

Table 4.1: Summary of lake locations, maximum water depth, and slump status. Lakes labelled with “A” are reference lakes and lakes labelled with “B” are slump-affected lakes. Notes on fish presence is from D’Onofrio (2014)[a]. *Chaoborus americanus*[b] observed in 2017 zooplankton net tows was used as an indicator of fish absence (Uutala, 1990). *Daphnia*[c] with extended head helmet observed in 2017 zooplankton net tows was used as an indicator fish presence (Haney et al., 2013). Extended helmets are a morphological adaptation to avoid fish predation (O’Brien et al., 1979). Relative abundances of *Daphnia* spp. data (% *Daphnia* spp.), and number of individuals of *C. americanus* and *C. trivittatus* are based on the 2017 zooplankton survey. Lakes with “-” indicated information was not available.

Lake ID	**Latitude (N)**	**Longitude (W)**	**Depth (m)**	**Slump Status**	**Notes on Fish**	**% *Daphnia* spp.**	***C. americanus***	***C. trivittatus***
5A	68°33.067	133°38.411	8.0	No slump	Fishless[a]	0	-	1
5B	68°32.243	133°39.489	6.8	Slump stabilizing	Fishless[ab]	0.88	1	-
9A	68°58.072	133°53.875	2.5	No slump	Fishless[a]	3.31	-	-
9B	68°58.220	133°53.968	2.1	Active slump	Fish present[a]	0	-	-
10A	69°07.116	134°10.161	3.0	No slump	Fish present[c]	7.19	-	-
10B	69°07.252	134°10.993	3.6	Large active slumps	Fishless[a]	0	-	-
12A	68°31.091	133°22.482	7.2	New slump present	-	0	-	-
12B	68°31.960	133°21.258	4.0	Slump reactivation	-	0	-	-
14A	68°31.048	133°44.975	8.0	No slump	-	0.80	-	1
14B	68°31.757	133°44.232	5.2	New and active slumps	-	0	-	-
15A	68°87.924	133°94.787	2.9	No slump	-	-	-	-
15B	68°86.087	133°93.471	7.0		-	-	-	-
20A	69°07.660	134°12.861	2.0	No slump	Fishless[b]	10.9	14	-
35B	69°01.485	134°09.625	3.0	Old stable large slump	-	20.9	-	-

Table 4.2: Summary of lake water chemistry data based on lake surface water samples collected in 2017. Lakes are categorized by reference and slump-affected lakes ("A" are reference lakes and "B" are slump-affected lakes). Variables analyzed are total nitrogen (TN), dissolved organic carbon (DOC), total phosphorus filtered (TP), conductivity (Cond), total dissolved solids (TDS), apparent colour, alkalinity, pH, and water temperature (Temp).

Lake ID	TN (mg/L)	DOC (mg/L)	TP (mg/L)	Alkalinity (mg/L)	Colour (CU)	Cond (µS/cm)	TDS (mg/L)	pH	Temp (ºC)
5A	0.42	13.6	0.006	12.4	98	67.5	51	7.15	12.99
5B	0.40	13.9	0.004	38.6	71	329.0	210	7.60	13.92
9A	0.69	20.3	0.019	9.6	181	44.3	55	7.01	17.45
9B	0.45	7.4	0.004	126.0	22	463.0	282	8.37	16.97
10A	0.74	22.4	0.032	11.4	219	67.7	78	7.06	17.08
10B	0.38	7.7	0.003	159.0	25	1380.0	1110	8.23	16.14
12A	0.38	11.9	0.005	2.4	121	51.4	63	6.26	12.90
12B	0.36	4.3	0.003	80.5	38	548.0	325	8.05	12.61
14A	0.98	16.0	0.011	64.5	66	258.0	176	7.88	15.09
14B	0.35	8.3	0.003	68.3	32	1050.0	823	7.92	14.32
20A	0.72	21.3	0.051	10.8	223	53.5	75	7.08	17.08
35B	1.06	24.4	0.115	26.1	203	96.5	102	7.48	18.20

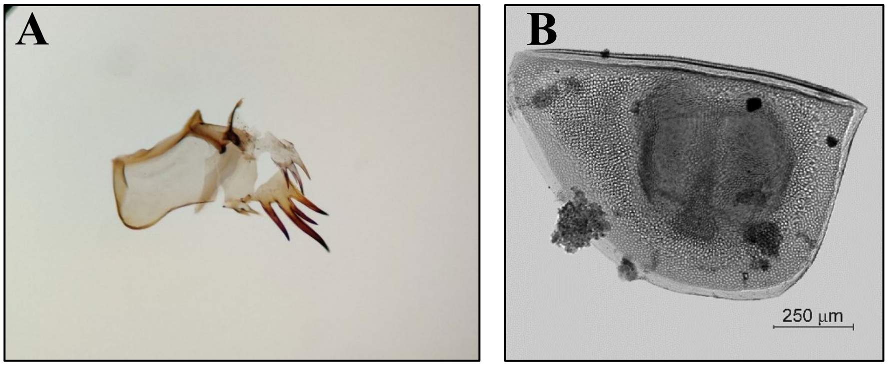

Figure 4.3: Picture of (A) *Chaoborus trivittatus* mandibles under an inverted microscope at 100x magnification and (B) *Daphnia* ephippia under a light microscope at 250x magnification.

Table 4.3: Summary of the wet sediment weight analyzed for the top and bottom sediment intervals from the study lakes. Lakes are categorized by reference and slump-affected lakes ("A" are reference lakes and "B" are slump-affected lakes). "Sediment Interval (cm)" indicate the core depth for the "Top" and "Bottom" sediment intervals from each study lake.

Lake ID	**Sediment analyzed (g)**	**Sediment Interval (cm)**
5A Top	2	0-2
5B Top	2	0-2
5A Bottom	2	20-21
5B Bottom	2	15-16
9A Top	2	0-2
9B Top	2	0-2
9A Bottom	2	12.5-13
9B Bottom	2	22-22.5
10A Top	6	0.5-1
10B Top	6	0.5-1
10A Bottom	6	22-22.5
10B Bottom	6	31.5-32
12A Top	2	0-2
12B Top	2	0-2
12A Bottom	2	18-19
12B Bottom	2	18-19
14A Top	2	0-2
14B Top	2	0-2
14A Bottom	2	24-25
14B Bottom	2	24-25
15A Top	2	0-2
15B Top	2	0-2
15A Bottom	2	20-20.5
15B Bottom	2	24-25
20A Top	6	0.5-1
20A Bottom	6	21-21.5
35B Top	6	0.5-1
35B Bottom	6	25-25.5

4.4 Results and Discussion

The aim of this study was to determine if *Daphnia* presence could be confirmed in any of the study lakes where they were not present in the 2017 zooplankton net tows. In addition, *Chaoborus* presence was also noted since they are predators on *Daphnia*, and presence of *Chaoborus americanus* is an indicator of fish absence. Fish are also predators on *Daphnia* and are known to contribute to the absence of *Daphnia* in lakes (Galbraith Jr, 1967; Černý & Bytel, 1991). To achieve this, I examined the 'top' and 'bottom' sediment intervals of lake sediment cores for *Daphnia* ephippia and *Chaoborus* mandibles in 14 lake sediment cores. The low amount of sediment analyzed likely overestimates *Daphnia*/*Chaoborus* absence, especially for 10 of the 14 study lakes where I had to use remaining sediment from cores collected in 2013. For this reason, I limit my analysis and interpretation of the results to focus on *Daphnia*/*Chaoborus* presence in lakes rather than absence.

Daphnia ephippia were detected in 6 of 14 lakes, and the number of ephippia extracted from sediment intervals ranged from 1 to 10 (Figure 4.4). *Daphnia* ephippia were detected in the top intervals of Lakes 5A and 9A, and in the bottom intervals of Lakes 10A, 15A, 20A, and 35B (Figure 4.4). The presence/absence of *Daphnia* in the study lakes are mostly consistent with the 2017 zooplankton survey. *Daphnia* were detected in Lakes 10A, 20A, and 35B in the 2017 zooplankton survey, as well as the bottom intervals. Lake 35B has a stabilized, revegetated slump scar and is chemically similar to an undisturbed lake, indicating that the slump has been stable long enough for lake recovery to occur. *Daphnia* ephippia were detected in the top interval of Lake 5A but were not detected in the 2017 net tow. *Daphnia* ephippia were not detected in the 'top' or 'bottom' sediment intervals for Lakes 14A and 5B, despite *Daphnia* being present in the 2017 zooplankton survey (Figure 4.4). The amount of sediment analyzed for each 'top' and ‘bottom’ intervals for both lakes were 2 g, and so this is likely the result of the low sediment volume analyzed (Table 4.3). I do not report any new instances of *Daphnia* presence in slump-affected lakes based on my sediment data, which is consistent with my observation of the absence of *Daphnia* in slump-affected lakes from the 2017 zooplankton survey. Ephippia were not detected in the top or bottom sediments of Lakes 9B, 10B, 12B, 14B, and 15B.

Chaoborus mandibles were detected in 3 of 14 lakes, and the number of mandibles extracted from sediment intervals ranged from 1 to 9 (Figure 4.4). *Chaoborus* mandibles were detected in the ‘top’ and ‘bottom’ intervals of Lakes 5A and 20A, while mandibles were only detected in the ‘bottom’ interval of Lake 14A (Figure 4.4). *Chaoborus trivittatus* was the only *Chaoborus* species identified. In the 2017 zooplankton survey, *C. trivittatus* was present in Lakes 5A and 14A, and *C. americanus* (an indicator of fishless conditions) individuals were present in Lakes 5B and 20A (Table 4.1). I do not report any new instances of *Chaoborus* presence in any additional lakes based on my sediment data.

Based on my sediment data and 2017 zooplankton survey, *Chaoborus* and *Daphnia* coexist in Lakes 5A/B, 14A, and 20A (Figure 4.4). *Chaoborus* are well-known predators on *Daphnia* that has potential to influence *Daphnia* abundances and assemblages in Mackenzie Delta upland lakes. My study also reports the first record of *Chaoborus* presence in tundra lakes of the Mackenzie Delta region. *Chaoborus* is typically associated with the boreal zone (below the treeline) (Dupuis et al., 2008; Luoto & Nevalainen, 2009) and *Chaoborus* in tundra lakes are at the edge of their biogeographic range (Taylor et al., 2016). This information can assist with study lake selection for future studies examining the ecological impacts of thaw slumping. For example, Lakes 5A/B, 14A/B, and 20A/B can be selected for long-term reconstruction of *Daphnia* and *Chaoborus* assemblages to elucidate the impact of retrogressive thaw slumping on

Daphnia and *Chaoborus* throughout time. This information can be used in conjunction with Thienpont et al. (2013), providing the long-term changes in diatom assemblages from the same study lakes to assist with interpretations of *Daphnia* and *Chaoborus* assemblages throughout time.

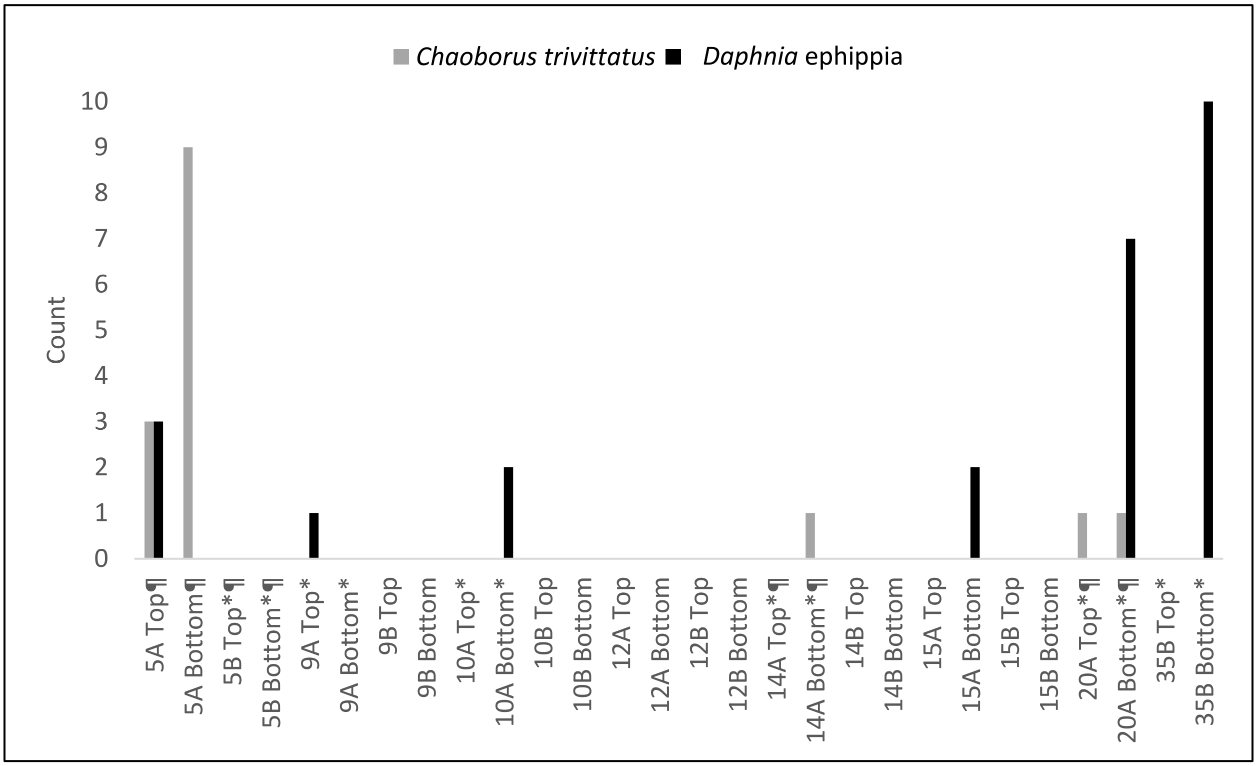

Figure 4.4: Individual *Daphnia* ephippia and *C. trivittatus* mandible(s) counts from each top and bottom sediment intervals for all study lakes. Presence of (*) *Daphnia* and (¶) *Chaoborus* based on the zooplankton survey from 2017.

CHAPTER 5: GENERAL CONCLUSIONS AND FUTURE DIRECTIONS

5.1 Thesis summary and Conclusions

Retrogressive thaw slumping in the western Canadian Arctic has been increasing in frequency and intensity over the past few decades, primarily due to increased air temperatures over the region (Lantz & Kokelj, 2008; Kokelj et al., 2017). Thaw slumping has been extensively studied in the Mackenzie Delta region, particularly its implications on Arctic lake water chemistry (Kokelj et al., 2009; Thompson et al., 2012; Houben et al., 2016). Some studies investigated the biological responses to retrogressive thaw slumping (e.g. Thienpont et al., 2013); however, little is known about the implications of thaw slumping and zooplankton. I identified two potential concerns related to zooplankton and thaw slumping that I explored in this thesis. 1) The increase in water clarity as a result of DOC being scoured from the water column may increase the risk of exposure of zooplankton to potentially harmful UV radiation. 2) It is unknown if *Daphnia* have always been absent in slump-affected or if thaw slumping is a potential mechanism responsible for *Daphnia* absence. To investigate each concern, I compared *Chydorus* carapace absorbances trends between Lakes 9A (undisturbed) and 9B (slump-affected) to infer changes in exposure to UV radiation over time. I also examined top and bottom sediment intervals in 14 lakes (7 undisturbed lakes and 7 slump-affected lakes) for *Daphnia* ephippia and *Chaoborus* mandibles to provide information on their biogeographic distribution in the uplands region. These concerns were explored separately in Chapters 3 and 4.

5.1.1 Does retrogressive thaw slumping lead to an increased risk of exposure of aquatic biota to potentially harmful UV radiation?

Sediment cores from two lakes (one slump-affected lake and one lake with no history of thaw slumping) were collected in the Mackenzie Delta region to examine changes in exposure of *Chydorus* to UV radiation throughout time inferred from carapace absorbances. To reconstruct long-term changes in UV exposure, *Chydorus* carapaces were hand-picked from sediment intervals and then subjected to absorbance measurements at UVR wavelengths (305 nm and 340 nm). Overall, *Chydorus* carapaces absorbances in Lake 9B were similar to the carapace absorbances in Lake 9A; carapace absorbances were mostly between 1 and 1.5 in both lakes. Collectively, this indicates *Chydorus* in Lake 9B were not exposed to higher levels of UV radiation despite the higher water clarity as a result of thaw slumping. I suggest *Chydorus* may be living amongst dense macrophyte beds that develop as a result of increased water clarity (Mesquita et al., 2010), providing shade from UV radiation. Alternatively, DOC concentrations may have remained above a threshold limit (studies suggest a DOC concentration threshold of 5 mg/L for UV exposure; Williamson et al., 1996; Häder et al., 2007) and increased melanin as an adaption was not required. The DOC concentration in Lake 9B is 7.4 mg/L, although this likely includes a substantial proportion of non-chromophoric DOC given the noted clarity of the water column. These findings highlight the need to further explore the relationship between DOC quality and quantity, light dynamics, and aquatic biota in slump-affected lakes.

5.1.2 Are Daphnia *present in slump-affected lakes?*

My undergraduate Honour's Thesis compared zooplankton assemblages between slump-affected lakes and undisturbed lakes in the Mackenzie Delta region and found *Daphnia* to be absent in slump-affected lakes. This suggested *Daphnia* may be particularly sensitive to the implications of thaw slumping (i.e. increased conductivity and decreased DOC and nutrients). I examined *Daphnia* subfossil ephippia remains in the top and bottom lake sediment intervals to determine if *Daphnia* could be identified in slump-affected lakes where they were absent in the 2017 zooplankton survey. Lake 5A is the only study lake to detect *Daphnia* presence in lake sediments (top interval) where they were not encountered in the zooplankton survey. My sediment-based investigation into *Daphnia* presence/absence did not report any new instances of *Daphnia* presence in slump-affected lakes. However, *Daphnia* ephippia were not detected in the 'top' or 'bottom' sediment intervals for Lakes 14A and 5B, despite *Daphnia* being present in the 2017 zooplankton survey. This indicates that an insufficient volume was analyzed and interpretations about *Daphnia* absence based on the sediment data should be viewed with caution.

5.2 Future directions

The findings presented in this thesis highlight the complex responses of cladocerans to permafrost thaw slumping in lakes of the Mackenzie Delta region. There is much more research needed to examine the impacts of thaw slumping on cladocerans and zooplankton in general. Future studies can use the paired-lake study design to compare temporal trends in *Chydorus* and *Alona* carapace absorbances between slump-affected lakes and reference lakes to expand on my study of Lakes 9A/9B, focusing on a gradient of lake DOC concentrations. For example, some slump-affected lakes are below the DOC threshold (~5 mg/L; Williamson et al., 1996; Häder et al., 2007) that would allow UV radiation to penetrate deep in the water column. Future studies should also compare diffuse attenuation coefficients and absorptivity of DOC at 305 nm and 340 nm wavelengths between slump-affected and undisturbed lakes to provide insight into the quality of DOC and the depth UV radiation penetrates in the water column, similar to Gareis et al.

(2010) that examined UV penetration in delta lakes of the Mackenzie Delta Region where thaw slumping does not occur.

The top-bottom study showed *Daphnia* ephippia were not encountered in slump-affected lakes, tentatively corroborating with the findings from the 2017 zooplankton survey where *Daphnia* was absent in slump-affected lakes. It is worth revisiting this study to examine larger sediment volumes, especially for the lakes sampled in 2013 where only limited sediment volumes remained. Ideally, 10 g of wet sediment should be used for *Daphnia* (Jeppesen et al., 2002; Szeroczyńska & Sarmaja-Korjonen, 2007; Nevalainen et al., 2013), and up to 50 g of wet sediment should be used to characterize *Chaoborus* assemblages (Quinlan & Smol, 2001, 2010). Large volumes of surface sediments can be collected quickly using an Eckman dredge. In addition, retrieving environmental DNA (eDNA) from lake sediments (and sediment cores) can be used to examine *Daphnia* presence/absence in relation to predation factors as well as water quality factors, providing stronger characterization of the biotic and abiotic controls on *Daphnia* biogeography in the Mackenzie Delta uplands. This will help to disentangle why (and if) *Daphnia* are absent in slump-affected lakes.

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