

**Environmental change in Yellowknife Bay,
Northwest Territories, over the last two
centuries inferred through diatom
paleolimnological analysis**

By

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ABSTRACT

Great Slave Lake is one of the largest subarctic lakes in the world and the deepest lake in Canada. The duration and strength of thermal stratification is increasing in many subarctic lakes across the northern hemisphere, but little information is available for larger and deeper systems such as Great Slave Lake. This thesis uses paleolimnological techniques to investigate ecological changes occurring in Yellowknife Bay, Great Slave Lake, over approximately the last two centuries based on diatom and *Chla* analysis of a dated sediment core. The results of this thesis showed a sharp increase in the relative abundance of the smaller planktonic *Discostella stelligera* from <10% to ~25% diatoms and a decline in heavier *Aulacoseira* sp. These data suggest that over the last ~two centuries thermal stratification is increasing in Yellowknife Bay with potentially important ecological consequences for this large subarctic lake.

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Chapter 1.0 Introduction

Yellowknife, Northwest Territories, is located in the Canadian subarctic and is experiencing stress caused by climate change. The subarctic is a region that is warming at a faster pace compared to the rest of the world (Huang et al., 2017). In Yellowknife, the last few decades have brought noticeable changes to the surrounding environment that have directly impacted the city and surrounding communities such as: infrastructure problems with changing permafrost layers; problems with drinking water supply from higher turbidity levels than normal due to increased erosion; animals from the south migrating further north altering traditional hunting activities; introductions of invasive species, and more (Hitt, 2010). Many subarctic lakes are changing due to climate change as well, with longer ice-free seasons due to warmer weather and surface waters, causing longer and stronger periods of thermal stratification (Rühland et al., 2008). Increased duration and strength of thermal stratification in lakes can alter limnological characteristics that are important for the ecosystem. Decreases in deep water dissolved oxygen caused by increased stratification is a major issue for lake ecosystems as it can negatively impact fish habitat and alter chemical interactions at the sediment water interface (Boehrer & Schultze, 2008; Sivarajah et al., 2020). Evidence supporting increased thermal stratification has already been observed in small lakes around the city of Yellowknife (Sivarajah et al., 2020), although there is a lack of research exploring whether or not similar changes are occurring in Yellowknife Bay, an inlet which extends northwards to Yellowknife from the open waters of Great Slave Lake.

Legacy pollution is a major threat to ecosystems worldwide with legacy arsenic contamination being a common result of historical mining activities (Little et al., 2020; Sivarajah et al., 2020; Sprague & Vermaire, 2018; Thienpont et al., 2016). During the second half of the

20th century, Yellowknife was the centre of one of the most productive and profitable gold districts in Canada. The city expanded rapidly during this period mainly due to its mining boom. Several long-running gold mines were operational surrounding the City of Yellowknife during this period. Giant Mine was the largest of the nearby mines, generating over 7 million ounces of gold throughout its operational period (1948-2004), making it one of the single most productive mines in Canadian history (Palmer et al., 2015). Another notable mine in the region that was operational during the same period was Con Mine. Gold mining in the Yellowknife region resulted in the generation of massive amounts of arsenic-bearing waste as a by-product of the gold extraction process. Gold within Yellowknife's deposits is hosted within the mineral structure of arsenic bearing minerals and is known as refractory gold. Roasting the ore is one of the main steps needed to extract gold in this form (Fawcett et al., 2015).

Through the ore-roasting process, Giant Mine produced over 237,000 tonnes of arsenic trioxide (As_2O_3) waste (Government of Canada, 2018). Most of this arsenic waste is currently stored in underground chambers, however, in the early days of Giant Mine's operations, much of the arsenic waste was released directly into the environment (about 7,400 kg of arsenic trioxide waste was released per day into the atmosphere in the early 1950's) (Government of Canada, 2018). Anthropogenic metal contamination of aquatic ecosystems as a result of mining activities has been seen worldwide, with Yellowknife being a dramatic example of widespread metal contamination across subarctic landscapes and freshwater ecosystems (Galloway et al., 2017; Palmer et al., 2015). Because of this, many lakes within the Yellowknife region still exhibit arsenic concentrations in surface waters well above federal drinking water guidelines along with high levels of arsenic still present in lake sediments (van den Berghe et al., 2018; Chetelat et al., 2019; Galloway et al., 2017; Palmer et al., 2015; Schuh et al., 2018).

Arsenic has been identified as a toxic long-term pollutant and can often be distributed in aquatic ecosystems adjacent to historical mining sites (Houben et al., 2016; Little et al., 2020; Sprague & Vermaire, 2018; Thienpont et al., 2016). Because arsenic is a redox sensitive element, it can be mobilized from lake sediments under specific conditions, which can have toxic effects on biota (Little et al., 2020; Palmer et al., 2019; Sprague et al., 2016; Wang & Mulligan, 2006). Due to its high toxicity and mobility, arsenic in the environment is of particular concern (Little et al., 2020; Sprague et al., 2016). The long-term impacts of arsenic in aquatic ecosystems are relatively poorly understood as the majority of data currently available are from short-term laboratory-based studies (Little et al., 2020).

Yellowknife Bay is a body of water that is of significant importance to the city of Yellowknife and its surrounding communities. To this day, the bay retains high levels of legacy arsenic contamination within its lake sediments (Andrade et al., 2010). Low oxygen levels in deep water can promote mineral-bound arsenic in sediment to dissolve and diffuse upwards into overlying water which can result in greater arsenic exposure to aquatic biota (Andrade et al., 2010). The response to increased temperatures in the subarctic due to climate change will be longer ice-free seasons for lakes, which will increase the length of the stratified seasons and could increase the depth of mixing and will ultimately lead to lower oxygen concentrations in the hypolimnion of lakes (Rouse et al., 1997). Should this be the case in Yellowknife Bay, current stores of legacy arsenic contamination from past gold mining activities may become more mobile as a result of climate change, and dissolve and diffuse into the waters of the bay (Jackson & Miller, 2000; McDonald et al., 2015).

Understanding how the environmental conditions in Yellowknife Bay have changed over the past few centuries is crucial information for understanding long-term ecological changes

resulting from climate change and also the future fate of legacy arsenic contamination in the bay. The objective of this thesis was to quantify environmental change over the last ~two centuries, if any, by studying a sediment core from Yellowknife Bay through the reconstruction of past diatom assemblages and primary-productivity as a proxy for environmental change. Diatoms are excellent bio-indicators for identifying changes in the environment when monitoring data is absent and have been particularly useful in understanding climate change impacts and anthropogenic stressors on subarctic lakes (Hamilton et al., 2021; Rühland et al., 2008; Rühland & Smol, 2005; Sivarajah et al., 2019, 2020; Sorvari et al., 2002). They are one of the most commonly used bio-indicators in paleolimnological studies due to their role as primary producers, because their observable environmental sensitivities allow for the identification of environmental change to a high degree of certainty, and because they are abundant and well preserved in lake sediments (Dixit et al., 1992; Little et al., 2020; Smol, 2009). The diatoms within this sediment core were studied at a high resolution (1 cm intervals) in order to temporally pin point when changes began to occur within Yellowknife Bay. Results from this thesis will provide further knowledge on the impacts of climate change on deep lakes in the subarctic and will explore how these impacts are connected with the long term fate of an environmental contaminant released into the ecosystem by gold mining. This thesis is part of a larger ongoing research program in collaboration with Environment and Climate Change Canada to investigate the legacy mining contamination in the Yellowknife area of Northwest Territories.

Chapter 2.0 Literature Review

2.1 Geography of Yellowknife, NWT, Canada

Yellowknife (62°27' N, 114°22' W) is the capital city of the Northwest Territories, Canada (Fig.1). It has a population of just under 20,000 and is located on the western shore of Yellowknife Bay which extends northwards from Great Slave Lake (City of Yellowknife, 2021). The city is situated in Canada's subarctic, about 400 km south of the Arctic circle, amongst spruce-dominated boreal forests (CDENTO, 2021). The land around Yellowknife has long been inhabited by the Yellowknives Dene First Nation's people (YKDFN, 2021). There are two Yellowknives Dene First Nations communities in close proximity to Yellowknife; Detah and N'dilo. The area surrounding Yellowknife is located within the Canadian Shield Slave Structural Province and lies within the Great Slave Uplands and Great Slave Lowlands of the Taiga Shield High Boreal Ecoregion where exposed bedrock plains and undulating uplands are the dominant features (Government of Northwest Territories, 2019).

There are three major geologic groups within the immediate vicinity of the city: to the west of Yellowknife is the *Anton Complex* that is characterized by younger, intrusive granitoids; to the east of Yellowknife, on the opposite shore of Yellowknife Bay, is the *Burwash Formation* that consists of Archean metasedimentary rocks; and running north to south along the western shore of Yellowknife Bay is the *Yellowknife Greenstone Belt* that consists of Archean metavolcanics rocks (Cousens, 2000). The gold deposits exploited by major mines in the region, such as Giant mine and Con mine, were located within the *Yellowknife Greenstone Belt* (Palmer et al., 2015). Due to its high mineral potential, this greenstone belt has become one of the most well-studied in the world and has been known to contain high concentrations of arsenic (As) (Ootes, L., 2004).

Climate change is occurring worldwide, however, amplified warming within the globe's Arctic and subarctic regions, where Yellowknife is located, has been observed with annual temperatures rising at alarming rates (Huang et al., 2017). This warming has already had substantial effects on the subarctic landscape within Canada and the rest of the world. Holocene warming in the central Canadian subarctic is believed to have already resulted in the migration of the treeline north by 50 km or more (Sulphur et al., 2016). Yellowknife typically experiences a continental subarctic climate, having short summers (mean temperature in July being 17.0 °C) and long winters (mean temperature in January being -25.6 °C) with temperatures generally remaining below freezing from early October to late April (ECCC, 2018). However, the Northwest Territories are currently warming at about three times the global rate (Yellowknife has seen an average temperature increase of 2°C from 1958 to 2012) and is causing significant changes to the natural environment, such as shifting ice conditions and longer ice-free seasons (Government of Northwest Territories, 2019; Spence, 2002). Environmental changes in the region have already led to infrastructure problems such as buckling roads from changing permafrost, and problems with the city's drinking water from increased erosion (Hitt, 2010) and could possibly contribute to greater diffusion of arsenic from sediments in freshwater lakes (Galloway et al., 2017; McDonald et al., 2015). Moving further into the 21st century, these issues related to climate change are expected to only increase. Lake-ice cover in the Northern Hemisphere is projected to decrease both in thickness (10-50cm) and duration (15-50 days) over the period of 2040-2079 when compared to 1960-1999 (Dibike et al., 2011; D. Mullan et al., 2017; D. J. Mullan et al., 2021). Future warming will also result in an overall increase in lake-water temperature, with summer stratification starting earlier and extending later into the year (Dibike et al., 2011).

Great Slave Lake is one of three large lakes in the Mackenzie River Basin. The lake has a surface area of 27,200 km², a volume of 1,070 km³, and a maximum depth of 614 m (Rawson, 1950; Schertzer et al., 2008). Due to its large size, depth, and cold climate, Great Slave Lake is oligotrophic (Rawson, 1950). Great Slave Lake is a dimictic lake meaning that it experiences mixing twice a year, in the spring and the fall (Schertzer et al., 2008). Schertzer et al. (2008) found that there was considerable inter-annual variability in the dates of spring and fall overturn and that spatial and temporal variability in air temperature and wind speed can result in highly dynamic responses of deep-water vertical mixing affecting thermal stratification characteristics. The shallower inshore waters, such as Yellowknife Bay, heat more rapidly and are less exposed to winds compared to the open waters of Great Slave Lake and therefore less heat is lost by circulation into the depths in these areas (Rawson, 1950). This suggests that climate change will likely have a considerable impact on the stratification regime of Great Slave Lake, especially within smaller and more isolated reaches of it, such as Yellowknife Bay, where warmer temperatures could increase the strength and duration of the summer stratification period that would in turn reduce deep water oxygen levels that could then influence the release of arsenic from lake sediments (Galloway et al., 2017; MacDonald et al., 2016; Miller et al., 2020). Increasing temperatures from climate change could also lead to increased primary production in subarctic lakes which could alter ecological communities and further contribute to reduction of deep water oxygen levels (Hayden et al., 2017). Reduction in the duration of winter ice-cover in subarctic regions will lengthen the prime period of primary productivity (Hampton et al., 2017). Increased precipitation, as predicted to occur with climate change, and land-use intensification increase nutrient inputs from terrestrial environments as well, leading to elevated dissolved organic matter and enhanced productivity in lakes (Hayden et al., 2017).

Arsenic concentrations in the near-surface lake sediments within a 30 km radius of Giant Mine ranged from 5 $\mu\text{g/g}$ to greater than 10,000 $\mu\text{g/g}$ (median of 81.2 $\mu\text{g/g}$, $n = 105$) (Galloway et al., 2018). The concentration of arsenic in lake sediments was typically the highest in lakes within 11 km from Giant Mine, with levels decreasing the further the lakes were from the mine (Galloway et al., 2017). Surface water arsenic concentrations within Yellowknife Bay is higher in the north end of the bay ($\sim 3 \mu\text{g/L}$) when compared to the main body of Great Slave Lake ($< 1 \mu\text{g/L}$) (Ch  telat et al., 2017, 2019). Andrade et al (2010) measured arsenic levels of approximately 50 mg/kg in Yellowknife Bay sediments prior to Giant Mine’s opening, with levels increasing to 1300 mg/kg during peak years of the mine’s operation.

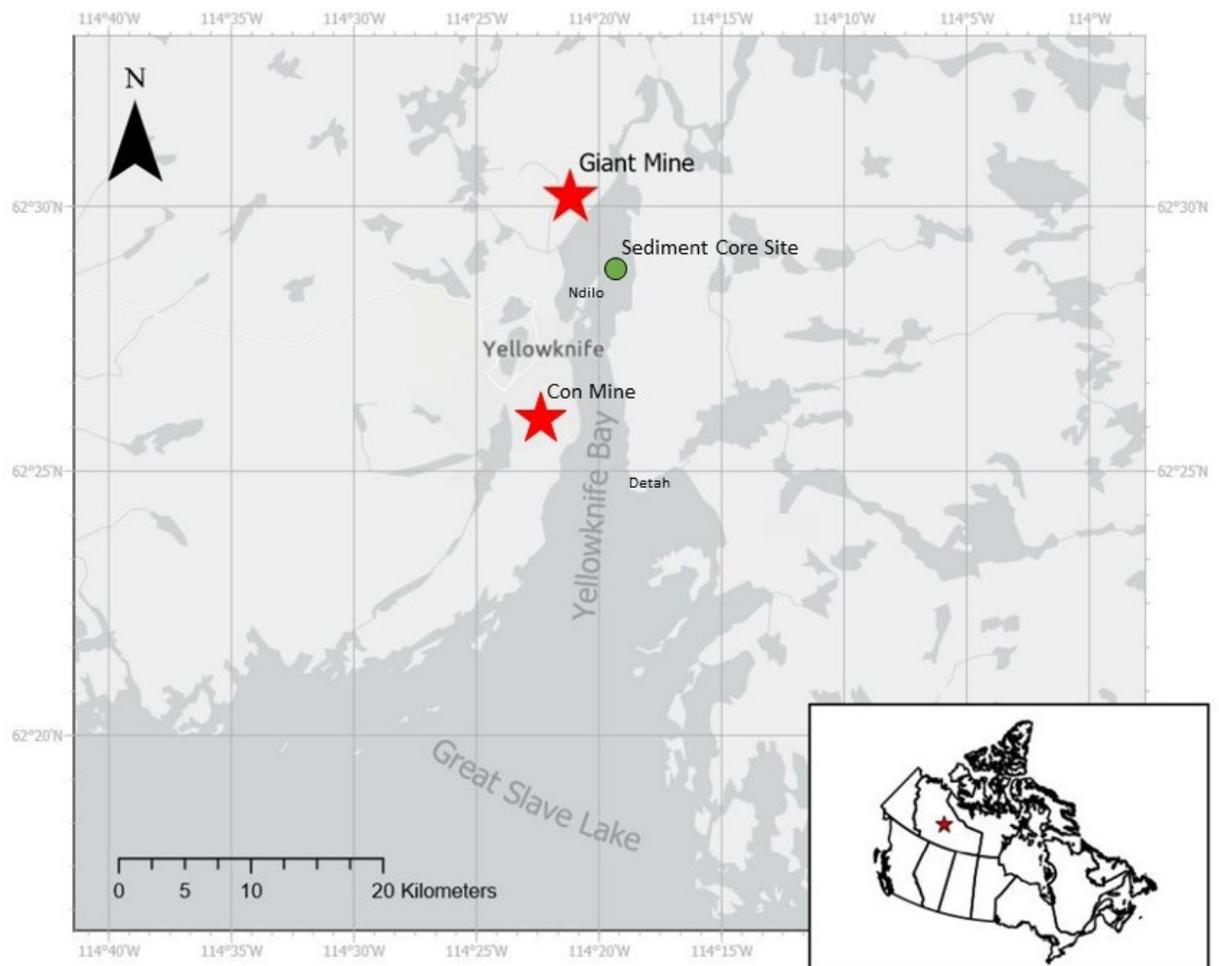


Figure 1. Yellowknife Bay in Great Slave Lake with the location of where the sediment core was taken from near the former Giant Mine.

2.2 Mining history & legacy contamination

The economic growth and development of the Canadian north has largely been driven by the exploration and extraction of mineral resources (Silke, 2013). Gold was first discovered in the Yellowknife area by prospectors in 1896 however nothing came of the discovery at that time as the region was considered to be inaccessible (Government of Canada, 2018). It wasn't until 1935, with the arrival of commercial aircraft (bush planes), that the area became more accessible.

It was at this time that the Yellowknife gold boom began and many eager prospectors returned to the area in search of precious metals along the shores of Great Slave Lake. With the breakout of World War II a few years later, gold was not considered a priority and there was a shortage of available men; mining companies therefore struggled to establish themselves in the Yellowknife area until after the war ended in 1945 (Government of Canada, 2018).

During the second half of the 20th century, Yellowknife was the centre of one of the most productive and profitable gold districts in Canada. Several long-running gold mines were operational surrounding the City of Yellowknife during this period. Giant Mine was the largest of the nearby mines, generating over 7 million ounces of gold throughout its operational period (1948-2004), making it one of the single most productive mines in Canadian history (Palmer et al., 2015).

The extraction of gold in the Yellowknife region resulted in the generation of large amounts of arsenic-bearing waste as a by-product of the ore roasting process. Gold within the deposits of the *Yellowknife Greenstone Belt* geologic region is predominantly hosted within the mineral structure of arsenopyrite (FeAsS), and to a lesser degree arsenian pyrite (FeS_2), in quartz-carbonate veins (Bromstad et al., 2017). Gold occurring in this form, known as refractory gold, requires four steps in order to be extracted: crushing, floatation, roasting, and cyanidation (Fawcett et al., 2015). The ore must be roasted in order for it to be more amenable to cyanidation; this roasting process produces arsenic vapour in which arsenic-bearing iron oxides and sulfur dioxide are emitted into the atmosphere via a roasting stack (Palmer et al., 2015). Once in the atmosphere, the arsenic vapour precipitates to form solid phase arsenic trioxide (As_2O_3) (Palmer et al., 2015). At Giant Mine, over 20,000 tonnes of arsenic in the form of

arsenic trioxide (As_2O_3) was inputted directly into the local environment through its roasting stack (Galloway et al., 2018).

Most of the direct input of arsenic contamination into the local environment occurred in the early years of Giant Mine's operations (1949-1951) when no environmental regulations or safety precautions were at play and roaster emissions were uncontrolled (Silke, 2013). It wasn't until 1952 when the first generation of arsenic trioxide (As_2O_3) capturing technology (an electrostatic precipitator) was implemented as a result of raising concern for worker health following the deaths of several local children due to accidental arsenic poisoning (Bromstad et al., 2017). In 1958, technological advancements allowed for the installation of a baghouse which dramatically reduced emissions (approximately 85% of As_2O_3 emissions from Giant Mine were released pre-1958) (Bromstad et al., 2017). Even with these later precautions set in place, the arsenic contamination that was directly inputted into the environment in the early years of operation is still very present within the modern landscape. Lake sediments in Yellowknife Bay, for example, still retain high levels of arsenic (Fawcett et al., 2015). It is worth mentioning that some mining waste was discharged directly into Yellowknife Bay (Mudroch et al., 1989) and that persistent arsenic pollution of regional soils are an on-going source of arsenic to regional lakes (Palmer et al., 2021).

In addition to the arsenic trioxide (As_2O_3) that was directly released into the environment via the Giant Mine roasting stack, the Government of Canada is now responsible for managing the 237,000 tonnes of arsenic trioxide (As_2O_3) waste that was successfully collected from the roasting operation's capturing technology (Government of Canada, 2018). Currently, this arsenic trioxide (As_2O_3) waste is safely contained and managed as it is stored in the former underground mining chambers of Giant Mine and frozen to keep it immobile (Government of Canada, 2018).

As arsenic trioxide (As_2O_3) does not decompose or disappear over time, there is currently no “walk-away” solution that exists for the Government of Canada; the costly operation of actively freezing and monitoring the arsenic waste is still the safest and best option to control this problem for the unforeseeable future (Government of Canada, 2018).

2.3 Arsenic in aquatic environments

It is not well understood what the influence of widespread distribution of arsenic-bearing emissions across a landscape is on the water quality of lakes (Palmer et al., 2015). However, it is known that arsenic exposure has toxic effects on humans as well as most other living organisms (Smith et al., 2000). The main sources of arsenic in the environment include weathering of rocks and soils, and anthropogenic inputs such as tailings and roaster emissions from gold and silver mines, wood preservatives, herbicides, pharmaceuticals, and glass manufacturing by-products, as well as coal-fired power plants (Little et al., 2020).

There is little doubt about the toxic effects of some arsenic species on biota, including humans (Matschullat & Deschamps, 2011). Inorganic arsenic is an internationally recognized Class 1, non-threshold carcinogen (Kabay et al., 2010). While arsenic exposure can occur in food, air, and water, most major chronic arsenic poisonings in humans have stemmed from water as this is the predominant route of exposure (Kapaj et al., 2006). Exposure to arsenic leads to the accumulation of arsenic in tissues which can result in the development of skin, organ, and lung cancers, cardiovascular disease, neuropathy, memory loss, hormone suppression, and birth complications (Kapaj et al., 2006). Health Canada (2006) has currently set guidelines for arsenic levels to not exceed 10 micrograms per litre in drinking water and has stated that they will

continuously review this guideline in order to reflect new information on health risks as they become available. There is an estimated 200 million people worldwide who are currently exposed to drinking water that exceeds the recommended limit of 10 µg/L (Naujokas et al., 2013).

Biologically, arsenic exerts a toxic effect through several mechanisms, the most significant of which is a reversible combination with sulfhydryl groups where it binds to the pyruvate dehydrogenase complex, blocks the Krebs cycle and interrupts oxidative phosphorylation, resulting in a marked depletion of cellular ATP and eventually the death of the metabolizing cell (Hughes, 2002). A second major form of toxicity is termed arsenolysis where arsenate (a phosphate analogue) affects oxidative phosphorylation by competition with phosphate and as a consequence, diminishes the formation of ATP and uncouples many reactions which results in the loss of energy for the cell (Hughes, 2002; Kabay et al., 2010).

Arsenic (As) is a relatively common, naturally occurring metalloid that exists in the environment in either organic or inorganic form (Eisler, 1988). Typically, arsenic exists in natural systems in one of four oxidative states: As(-III), As(0), As(III), and As(V). Because As(III) and As(V) appear most frequently in nature, they are of main concern when studying the impacts of arsenic in the environment; these two forms are commonly referred to as arsenite and arsenate, respectively (A. Little, 2019). Typically, inorganic arsenic is considered more toxic than its organic counterpart, and the arsenite (As(III)) species is more toxic than arsenate (As(V)) (Eisler, 1988; Razo et al., 1990).

Once arsenic has been deposited in lake sediments within the environment, it undergoes many post-depositional sedimentary processes such as remobilization, changes in speciation, and

the formation of new oxide compounds (van den Berghe et al., 2018; A. Little, 2019). Although the reactions driving arsenic release and sequestration by lake sediments is not fully understood, it is well established that when the hypolimnion (bottom water) is oxygenated, arsenic is recycled with chemical compounds such as iron (Fe) oxyhydroxides which controls and greatly reduces arsenic's mobility as insoluble precipitates are formed (Couture et al., 2009). If anoxia develops in the hypolimnion, reducing conditions within the sediments drive the dissolution of the oxyhydroxides and the release of arsenic into the water (Couture et al., 2009). When bound to oxyhydroxides under aerobic conditions, arsenic tends to be found in the As(V) state whereas under anaerobic conditions arsenic becomes reduced to its As(III) state (A. Little, 2019). Because of this behaviour, it is of concern to ecosystem health should a lake that has high levels of arsenic within its sediments develop anoxic conditions within the hypolimnion as this would promote higher levels of the more dangerous arsenite [As(III)] in the lake water. Dissolved oxygen levels are therefore very important to study within contaminated lakes.

2.4 Lake processes that influence dissolved oxygen

2.4.1 Water temperature

Water temperature plays an important role in several limnological and biological processes, including ice-cover break up, species distribution, and the growth and survival of many aquatic organisms including phytoplankton and fishes (Sharma et al., 2008). Water temperature is influenced by lake morphology, water chemistry, surrounding topography, and climate variables such as air temperature and solar radiation (Sharma et al., 2008). Large and deep lakes, such as Great Slave Lake, tend to be cooler than smaller and shallower lakes due to thermal inertia (Sharma et al., 2008). Oxygen solubility is directly impacted by water temperature and the actual

amount of dissolved oxygen will vary depending on the temperature (Fondriest Environmental, 2013). The solubility of oxygen decreases as water temperature increases. This means that lakes that are warmed as a result of climate change are unable to contain the same amount of dissolved oxygen within their waters. Rubalcaba et al. (2020) also showed that warmer water increases biological oxygen demand which could further limit oxygen supply within aquatic ecosystems that are impacted by climate change and warming.

2.4.2 Thermal mixing regimes & lake stratification

Lake water temperature, along with dissolved substances, contributes to density differences within the water; colder water is denser than warm water (Boehrer & Schultze, 2008). Density is one of the crucial physical quantities that controls flows such as vertical circulation of lakes; whether a lake will experience a full overturn or not depends on the density differences within the water (Boehrer et al., 2010). The circulation of water within lakes has a decisive impact on the redistribution of dissolved substances, such as nutrients and oxygen, throughout the water column (Boehrer & Schultze, 2008). Lake stratification and vertical mixing are very important processes for chemical and biological dynamics in deep freshwater lakes in the temperate zones (Ficker et al., 2017).

A lake becomes stratified when the surface layer of the lake warms up relative to the deeper layers of the lake. This usually occurs in the summer when the sun is able to heat up to very top layer of water, called the epilimnion, which ultimately causes it to become less dense than the colder water below it where sunlight is unable to penetrate (RMB Environmental Laboratories, 2021). The cooler, bottom layer of a stratified lake is called the hypolimnion and because it does not receive sunlight, it remains cold relative to the epilimnion (RMB Environmental Laboratories, 2021). Since the epilimnion is much less dense than the

hypolimnion, it circulates independently from the deeper, colder, and denser water; the thermocline is the dividing zone between these top and bottom layers of a lake (RMB Environmental Laboratories, 2021). As mentioned earlier, this water circulation is essential for the distribution of dissolved oxygen throughout the water column; when a lake is stratified, the water does not fully overturn as the hypolimnion is separated from the epilimnion and the atmospheric exchange of oxygen which can therefore cause the hypolimnion to become anoxic (Boehrer & Schultze, 2008; RMB Environmental Laboratories, 2021).

Many lakes in Canada are classified as dimictic lakes which means that they mix twice a year in the spring and fall (NALMS, 1996). When surface water temperatures are cooler and ice-free, the density difference between the epilimnion and the hypolimnion decreases, the water from the epilimnion becomes denser and therefore sinks deeper into the lake to mix the layers. Wind can also aid in this mixing process. These cooler water temperatures and mixing now allow for oxygen and nutrients to be distributed again throughout the entire water column all the way down to the deepest depths of the lake (RMB Environmental Laboratories, 2021). These ice-free mixing periods usually occur in the fall and spring when the surface water is cooler, but the lake is not covered by ice. When the lake is covered again by ice, the water cannot mix because it is not exposed to wind and the hypolimnion can become anoxic as it does during summer stratification (RMB Environmental Laboratories, 2021). In the Yellowknife area, under ice dysoxia or anoxia can occur in smaller lakes during this time (Palmer et al., 2019).

2.4.3 Aquatic Productivity

Primary productivity is the conversion of the sun's energy into organic material through photosynthesis. In aquatic ecosystems, primary productivity is driven by a number of factors such as temperature and the availability of nutrients and light (Biodivcanada, 2010). Lakes are

commonly classified based on their productivity. Oligotrophic lakes are generally deep and clear with little aquatic plant growth; these lakes maintain sufficient dissolved oxygen in the cool, deep-bottom water (Michigan State University, 2008). Eutrophic lakes are highly productive, turbid, and support abundant aquatic plant and algal growth. In deep eutrophic lakes, the bottom waters usually contain little or no dissolved oxygen (Michigan State University, 2008).

Mesotrophic lakes are those which fall somewhere in between oligotrophic and eutrophic (Michigan State University, 2008).

Climate warming in the 20th century has had profound effects on the limnology of many lakes worldwide, and has been linked to increases in primary productivity in aquatic ecosystems; especially in lakes at higher latitudes (Outridge et al., 2007; Prowse et al., 2006). Hypolimnetic oxygen concentration has long been considered to be an important indicator of eutrophication. Warming and longer ice-free seasons can increase the growing season for aquatic plants and algae which then could increase overall biomass, ultimately increasing the amount of organic material injected into the hypolimnion as well (NALMS, 1996; Zhang et al., 2019). When a lake is stratified, the greater organic loads increase the decomposition rates in the hypolimnion and, subsequently, increase the rate of oxygen depletion (NALMS, 1996). A chemical phenomenon called internal loading may occur if phosphorus is bound to iron hydroxide complexes in lake sediments when oxygen within the hypolimnion decreases (RMB Environmental Laboratories, 2021). Phosphorus may be chemically released from lake sediments into overlying water in anoxic conditions and find its way through the thermocline where it will fuel plant growth within the epilimnion, thus creating a positive feedback system which greatly contributes to oxygen depletion within the hypolimnion (NALMS, 1996; RMB Environmental Laboratories, 2021).

2.5 Diatoms as paleolimnological bioindicators

Environments across the world are constantly changing; some of these changes are due to natural causes, while others are related to anthropogenic influences. For some aquatic ecosystems, only short-term monitoring data are typically available, while no monitoring data have been collected for most lakes and rivers (Smol, 2009). Despite this, every day sediments are accumulating at the bottom of lakes. These accumulated sediments store a vast amount of chemical, physical, and biological information. The field of paleolimnology investigates the information stored within lake sediments in order to reconstruct past histories of lakes without any previous long-term monitoring data. Because to the law of superposition, sediments that are closer to the surface represent more recent time periods than those buried deeper beneath the surface. Sediment cores can be taken from an undisturbed location within a lake and the chemical, physical, and biological proxies found within these accumulated lake sediments can then be used to reconstruct past environmental conditions. Biological proxies are used to determine ecological responses to known or potential changes in lake systems (A. Little, 2019). Various bio-indicators can be used in order to reconstruct historical trends in phenomenon such as thermal stratification, water temperature, and bottom water oxygen in paleolimnological studies as particular organisms preserve well within lake sediments allowing researchers to identify former species assemblages based on microfossils and can then infer environmental conditions and changes based on well-established species-environmental relationships (A. Little, 2019; Smol, 2009).

Diatoms are single-celled algae that live nearly everywhere in the world where there is water; in streams, ponds, lakes, oceans, and even in soils (Spaulding, et al., 2021). Diatoms are a commonly used as biological proxy in paleolimnology studies because of their silica cell walls, called frustules, which preserve in lake sediments and can be used to identify their taxonomy.

They are a widely distributed algal group in freshwater systems, can migrate and colonize habitats quickly, and have unique environmental optima and tolerances which therefore make diatoms excellent indicators for environmental conditions and ecological change (Smol, 2009). Changes in diatom species assemblages over time and space, along with the identification of species of known tolerance to environmental variables, provide information on the magnitude and trajectory of potential environmental change (A. Little, 2019; Smol, 2009). *Discostella stelligera* diatoms, for example, are planktonic and are typically more abundant within the water column of lakes during thermally stratified periods thanks to their centric shape and small size which allow them to float in less turbid waters compared to heavier, non-centric diatoms (Boeff et al., 2016).

2.6 Previous work in Yellowknife

Climate change may be enhancing the negative impacts from long-term arsenic contamination within lakes in the region as shifts in diatom assemblages have lined up with recent warmer periods (Stewart et al., 2018). Warmer temperatures across the subarctic, especially in winter months, have resulted in earlier ice-out dates, earlier onsets of thermal stratification, and earlier development of algal blooms in both Europe and North America (Rühland et al., 2008). All of these changes can result in decreasing oxygen levels at the sediment-water interface. Should these changes be occurring in Yellowknife Bay, they could promote the diffusion of arsenic from sediments into the overlying water. Great Slave Lake is a dimictic lake meaning that it mixes from the surface to the bottom twice per year, once during the fall and once during the spring (Adrian et al., 2009). Changes to the duration and strength of thermal stratification, in particular, could potentially have major impacts on the lake ecosystem including the release of arsenic

contamination from lake sediments to overlying water. Paleolimnological studies in the northern hemisphere have already shown that high-latitude and high-altitude lake ecosystems are experiencing longer ice-free seasons and increased stratification (Winder et al., 2009). Increased warming of Yellowknife's epilimnion throughout ice-free months may not only increase the length of stratification, but it may also strengthen the stratification as well (Winder et al., 2009). Strengthened stratification would mean a greater density difference between epilimnion and hypolimnion and could further interrupt the distribution of dissolved oxygen throughout the water column, causing anoxic conditions in the hypolimnion.

Several studies have conducted paleolimnological research methods focused on using biological proxy from lake sediment cores taken from lakes within the general proximity of Yellowknife which have analyzed the effects of multiple stressors on these aquatic ecosystems (Hamilton et al., 2021; Sivarajah et al., 2019, 2020, 2021; Stewart et al., 2018; Thienpont et al., 2016). There have not, however, been many published paleolimnological studies using biological proxy to reconstruct past lake conditions within Great Slave Lake itself. The nearby studies have observed that stressors such as mining contamination, climate change, and urbanization are impacting the freshwater systems in the Yellowknife region. Results from a 2016 Pocket Lake study showed that the peak periods of arsenic input into the environment corresponded with synchronous changes in the biological community in the lake, including the near extirpation of cladocera and planktonic diatom taxa to which there still has been no indication that biological recovery has occurred (Thienpont et al., 2016). More recently, Hamilton et al (2021) identified a substantive increase in planktonic diatoms within Pocket Lake, such as *Discostella* species, which is indicative of warmer temperatures. Arsenic contamination to this degree can have major impacts on aquatic ecosystems as many of the affected biological indicators used in these

paleolimnology studies make up the bases of the food webs in these systems (Chen et al., 2015; Thienpont et al., 2016).

With some smaller lakes in the region, such as Pocket Lake, showing ecological changes that correspond with direct mining impacts, many lakes in the area are experiencing cumulative effects from multiple stressors (Sivarajah et al., 2019, 2020; Stewart et al., 2018). Sivarajah et al. (2020) explored the long-term impacts of multiple environmental stressors on deep subarctic lakes around Yellowknife. These stressors included climate change, mining contamination, and local land-use changes with the expansion of the city of Yellowknife over the second half of the 20th century. Sivarajah et al. (2020) found that the overall primary productivity increased in two deep water lakes near Yellowknife, Alexie and Grace lakes, and that the relative abundance of the smaller planktonic diatom species *Discostella stelligera* increased in both lakes over time as well. Sivarajah et al. (2020) suggest that the increases in primary productivity and *Discostella stelligera* are influenced by changes in thermal stratification at both sites as this has been documented in a wide range of lakes across the Northern Hemisphere. Sivarajah et al. (2020) does also link the increase in primary productivity of Grace Lake (the closer lake to Yellowknife/Con Mine) to increased urbanization and anthropogenic land-use change which would have likely brought in additional nutrients from the landscape into the lake. These land-use changes would have likely contributed to increases in Chl-*a* and in eutrophic *Stephanodiscus* taxa within the sediment records (Sivarajah et al., 2020). Prior to this study, Sivarajah et al. (2019) assessed the influence of multiple environmental variables across an arsenic gradient in 33 subarctic lakes near Yellowknife and similarly found that planktonic *Discostella stelligera* now dominate in deep lakes within the region. The results from Sivarajah et al. (2019) indicate that subarctic climatic conditions are important factors shaping diatom assemblage composition

in lakes that have been impacted as well both by mining activities and land-use changes near Yellowknife.

One of the only published studies of a paleolimnology study using diatoms in Great Slave Lake was undertaken by Stoermer et al. (1990) in McLeod Bay which is in the remote and far north-eastern arm of Great Slave Lake. Stoermer et al. (1990) found that the most substantial diatom changes over the last ~5000 years in the bay occurred within the past ~200 years, with the most obvious increases in planktonic abundance occurring in post-industrial times. There was also a general increase in planktonic species such as *Discostella* (formerly referred to primarily as *Cyclotella* species) and general overall increase in microfossil abundance in the most recent intervals of the sediment core (Stoermer et al., 1990). The results from Stoermer et al. (1990) portray that the lake environment was still oligotrophic but has undergone subtle changes overtime towards more biomass within the bay. Results from this study were also not at a high enough resolution to pinpoint exactly when environmental changes occurred.

Arsenic concentrations in lake sediments near Yellowknife have also been studied and are still elevated and have persisted in lake sediments for more than 60 years following the atmospheric deposition of roaster emitted arsenic trioxide (As_2O_3) from Giant mine (van den Berghe et al., 2018; Galloway et al., 2018; Schuh et al., 2018). A 2016 study examined the impacts of mining in Pocket Lake near Yellowknife where it was observed that a 1700% increase in arsenic concentration in the sediment profile coincided with known historical mining activities in the Yellowknife region (Thienpont et al., 2016). Because of the potential ecological and human health risks associated with the large volumes of arsenic-bearing waste produced by mining operations around Yellowknife, there has been detailed research in the area on the nature, extent, and behaviour of arsenic in the aquatic environment on-site and within the immediate

vicinity of Giant mine, Con mine, and the City of Yellowknife (Palmer et al., 2015). A number of studies have analyzed the geochemical aspects of lake sediments in the area and investigated the mobility of arsenic (Andrade et al., 2010; Houben et al., 2016; Menard, 2018), all studies have shown a spike in arsenic concentrations associated with the sediments corresponding with the time period of Giant mine's early operational years and have also shown that the legacy arsenic contamination is not being buried effectively overtime in Yellowknife Bay and the surrounding lakes.

Although bio-indicators, such as diatoms, have been analyzed from lake sediments in multiple studies of smaller lakes within the region, there haven't been any previous paleolimnological studies on Great Slave Lake that have examined environmental change over the last two centuries. Yellowknife Bay is an important body of water for surrounding communities with many people relying on it for sustenance and transportation. Great Slave Lake itself also has a commercial fishery. Understanding how conditions have changed within Yellowknife Bay could provide valuable knowledge with regards to the long-term fate of legacy arsenic contamination. This thesis aims to fill this gap in literature by analyzing biological indicators within a sediment core retrieved from Yellowknife Bay in order to reconstruct its environmental history over the last two centuries.

Chapter 3.0 Methods

3.1 Laboratory analysis

A sediment core retrieved from the north end of Yellowknife Bay (62°28.850 N, 114°19.368 W) using a Glew core (Glew et al., 2001) in 19 meters of water in the summer of 2018 was used for this analysis. The sediment core was 32.5 cm long and was subsampled at 0.5 cm intervals. The samples were refrigerated in the field and shipped to Carleton University where they were stored at 4°C until analysis could be completed.

To calculate the organic content of the sediment a 1 cm³ subsample of sediment was taken from each sediment interval for loss-on-ignition (LOI) analysis following standard protocol (Dean, 1974; Heiri et al., 2001). Samples were weighed for wet weight before being placed in a furnace at 110°C for a minimum of 12 hours to be dried. Once dried, samples were weighed for dry weight before being placed in a furnace at 550°C for four hours to burn the organic matter present in the sediment. Samples were then weighed a final time and the water and organic contents of the sediments were calculated.

An age-depth model for the Yellowknife Bay core was developed using ²¹⁰Pb and ¹³⁷Cs measured using Gamma Spectrometry at the PEARL lab at Queen's University (Kingston, Ontario, Canada). The age-depth model was created using a Constant Rate of Supply (CRS) model (Appleby & Oldfieldz, 1983). Chlorophyll-*a* was also recorded throughout the core at the PEARL lab at Queen's University. Spectral analysis was used to infer trends in sedimentary chlorophyll *a* based on a unique trough found in the 650-700 nm range of the spectral profile of the sediments (Hadley et al., 2019; Michelutti et al., 2010). The area of this trough has been correlated to the concentration of chlorophyll *a* and its major derivatives in the sediment,

providing a rapid, non-destructive method for estimating primary production (Hadley et al., 2019).

Diatom slides were prepared at 1 cm intervals for the sediment core, following standard procedures as outlined by Battarbee et al. (2001). Approximately 0.5 g of wet sediment from each sample was placed in a glass scintillation vial and treated with 10% hydrochloric acid (HCl) to remove any carbonate material. Each sample was then rinsed with deionized water five times and left to sit for 24 hours between each rinse to allow settling of diatoms. Once this step was complete, each sample was treated with 30% hydrogen peroxide (H₂O₂) and then placed in a hot water bath (~70°C) for 8 hours to oxidize any organic material in the sample. Once the samples had cooled, they were refilled with deionized water and rinsed seven more times, and again allowed to settle for 24h between rinses. Once completed, well-mixed aliquots from each vial were deposited onto glass cover slips and were left to dry at room temperature on a covered slide warmer overnight. These cover slips were mounted onto microscope slides using Naphrax[®]. A minimum of 400 diatom valves were identified per sample using a Leica DM 2500 LED microscope under 1000x/1.32 oil immersion lens. Diatom identifications were made to the lowest possible taxonomic level using guides by Krammer and Lange-Bertalot (1986-1991a).

3.2 Statistical Analysis

Diatom species counts were converted to relative abundance and species that were considered rare (<2% relative abundance in at least one sample) were removed from the dataset (Patterson & Fishbein, 1989). Stratigraphic plots showing diatom species relative abundance data throughout the core were produced using the r package 'rioja' (Juggins, 2020) in order to observe any major

shifts in species assemblages over time. Cluster analysis was determined using square-root transformed abundance data and the constrained hierarchical clustering using the 'rioja' package (Juggins, 2020). In order to further explore any major trends in the diatom data, Principal Component Analysis (PCA) was performed using the R package 'vegan' (Oksanen et al., 2020) and a biplot was generated to represent PC1 and PC2 using the 'ggbiplot' package (Vincent, 2011). Ellipses representing the zones based on the constrained hierarchical clustering of the stratigraphic plot were added to the PCA biplot to further visualize and analyze how the sediment core zones related to each other.

The most complete and up to date adjusted and homogenized Canadian climate data were also retrieved from the Government of Canada's Yellowknife station (2021). Mean annual and seasonal temperatures were converted to a 5-year moving average and compared to PC1 scores and the relative abundance changes of *Discostella stelligera* overtime.

Chapter 4.0 Results & Discussion

4.1 Results

^{210}Pb data from the Yellowknife Bay core indicated that the top 20 cm represented a time period from the beginning of the 20th century to present (Fig.2) based on the constant rate of supply (CRS) model (Appleby & Oldfieldz, 1983). The ^{137}Cs peak occurred at approximately 18 cm sediment core depth which according to the CRS model would correspond to the mid-1920's. Ideally, the ^{137}Cs levels within sediments would peak around the height of atmospheric nuclear bomb testing around 1960 (~14 cm based on the CRS model), however, often this peak can substantially precede this time period as a result of post-depositional ^{137}Cs mobility (Smol, 2009) which may explain this discrepancy of approximately 4 cm.

The Yellowknife Bay core had relatively low organic matter. The bottom of the core (32.5-10 cm) showed fairly consistent percentages of organic matter, ranging between approximately 5-6 % (Fig.2). From 10 cm to 2.5 cm there was a relatively constant increase of organic matter from 4.8 % in the late 1970's to 7.5 % in 2008. Following 2008, there was a short but abrupt decrease in organic matter back to 6.5 % in 2010 before levels consistently remained between 6.5-7 % to present day.

Chlorophyll-*a* levels very gradually increased from approximately 0.0025 mg/g from the bottom of the core to the 15 cm mark, where they reached a maximum of 0.0047 mg/g by the mid 20th century. Following 1950, there was a gradual decrease in Chlorophyll-*a* until the early 1980's, where levels reached approximately 0.0030 mg/g before steadily rising once again to 0.0046 mg/g around 2003. From approximately 2004 to 2013, there was an abrupt decrease in

Chlorophyll-*a* to a low of 0.00023 mg/g before a sharp increase once again to 0.0034 mg/g at 0.5 cm sediment depth.

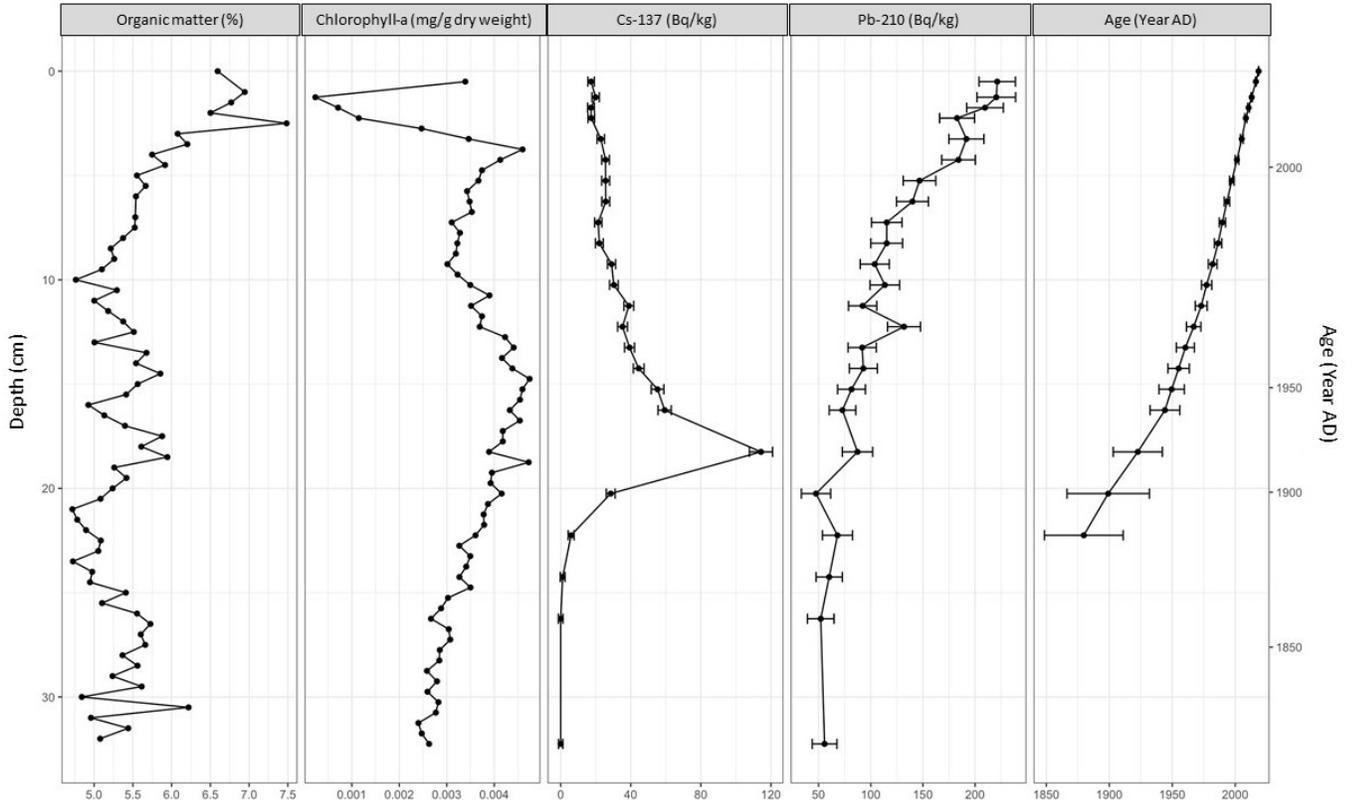


Figure 2. Age-depth profile of the Yellowknife Bay core based on ^{210}Pb activities and developed using a Constant Rate of Supply model (Appleby & Oldfieldz, 1983). ^{137}Cs activity, Chlorophyll-*a*, and organic matter are also plotted in relation to the sediment core depths.

The diatom assemblage from the Yellowknife Bay core contained 44 taxa with 24 taxa having a greater relative abundance than 2% in at least one sample (Fig.3). The stratigraphic plot was divided into three separate zones based on the cluster analysis. Each zone represented a different diatom community assemblage. Three zones were chosen based on the sum of squares equalling 5 as these divisions clearly took into account the increase in *Discostella stelligera* which was a species of particular interest. Zone 1 (from 32.5 cm to 20 cm) represented the deepest and oldest portion of the core, dating back to pre-1900 based on the CRS model, and was

mainly dominated by *Aulocoseira subarctica*, with relative abundances averaging 20 % (between 10-28 %). *Achnanthes minutissima* and *Tabellaria flocculosa* were the next most dominant species, with relative abundances around 10% throughout Zone 1. Zone 2 was between core depths of 20 cm to 9 cm and represented approximately 1900 to 1980 based on the CRS model. Zone 2 saw a slight reduction in relative abundance of *Tabellaria flocculosa* (from ~10% to 5%) and *Achnanthes lanceolata* (~8% to 5%) and an increased relative abundance of both *Fragillaria ulna* (~5% to 10%), *Fragillaria capucina* (~3% to 8%), and *Asterionella Formosa* (~0-5% to 8%). *Aulocoseira subarctica* remained as the dominant species in zone 2 (up to 25-30% relative abundance) with its relative abundances remaining relatively unchanged compared to zone 1. Zone 3 was the uppermost zone in the Yellowknife Bay core, from 9-0 cm which represented ~1980 to the present day based on the CRS model, and was defined by an abrupt increase in the relative abundance of planktonic *Discostella stelligera* (~15-20% relative abundance). In zone 3, *Aulocoseira subarctica* became the second most dominant diatom species (~20% relative abundance) as it was overtaken by *Discostella stelligera*, which became the most dominant species in the zone, with its highest relative abundances being found in the upper most intervals of zone 3, reaching a maximum relative abundance of 25%. *Tabellaria flocculosa* relative abundances also slightly increased again back to similar levels which were found in zone 1 (~8-10%). Both benthic *Fragillaria ulna* and *Fragillaria capucina* also saw a slight overall decrease in relative abundance (to ~5% or less).

Organic matter and chlorophyll-*a* remained relatively constant throughout the zones with some slight increases in zone 3. Organic matter especially increased in the intervals with the highest *Discostella stelligera* relative abundances.

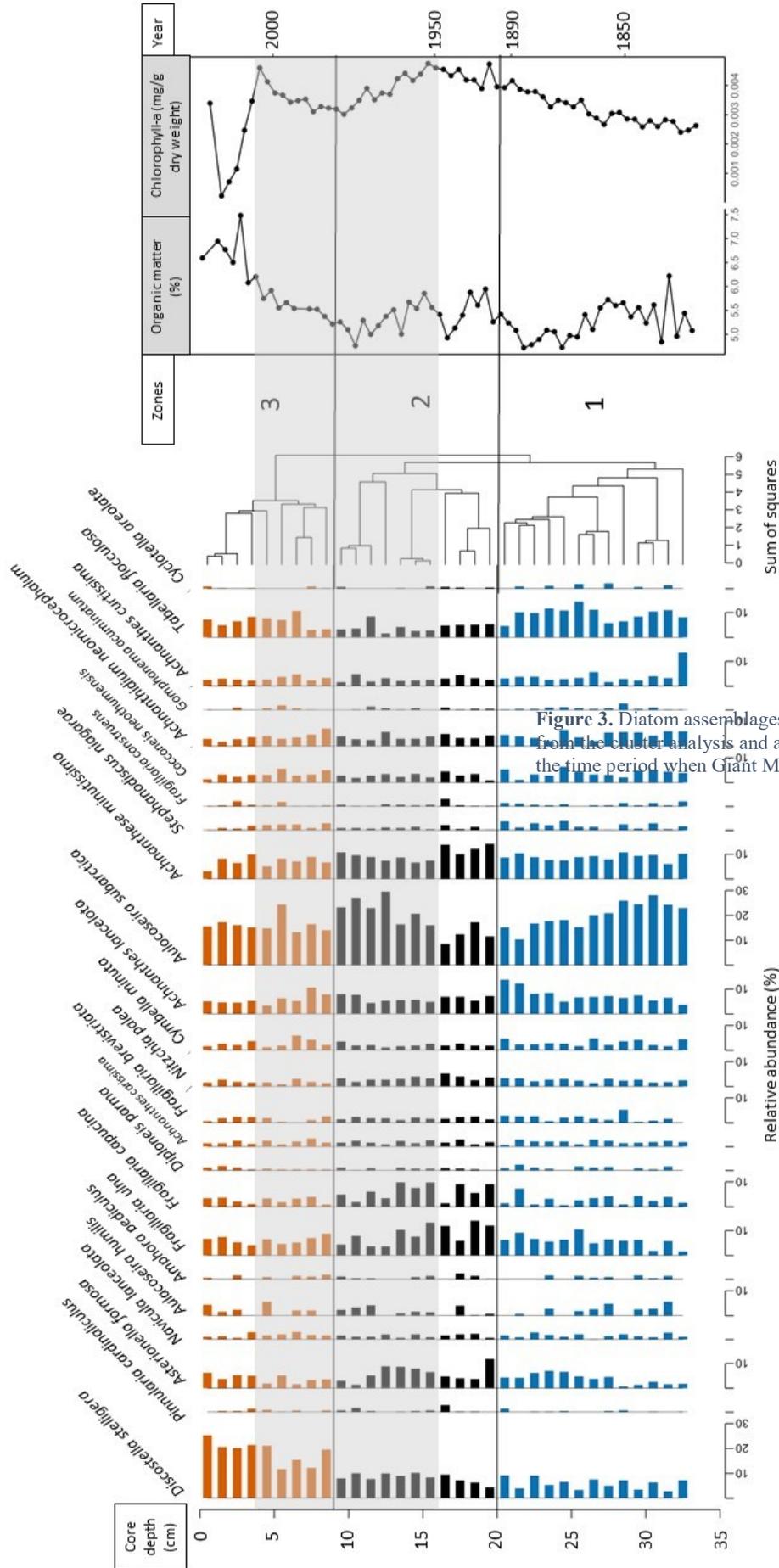


Figure 3. Diatom assemblages from the 2018 Yellowknife Bay core. Three zones from the cluster analysis and are represented by the different colours. The grey shaded area represents the time period when Giant Mine was operational.

The Principal Component Analysis of the Yellowknife Bay core showed that PC1 explained 34.5% of the variation in the diatom assemblage and was largely driven by a shift in samples dominated by the tychoplanktonic *Aulacoseira subarctica* to those dominated by the planktonic *Discotella stelligera* (Fig.4). Ellipses corresponding to the three zones derived from the constrained hierarchical clustering clearly display that zones 1 and 2 are closely related while zone 3, which represents the most recent sediments, has a more unique diatom assemblage than the rest of the core (Fig.4). PC1 has a largely negative association based on the Eigenvalues with *Discotella stelligera* (-0.85) and a positive association with *Aulacoseira subarctica* (0.47), showing that these two species are the main drivers of variability within the diatom assemblage over time (Table 1). An abrupt increase in the relative abundance of *Discotella stelligera* in zone 3 was the main driver of diatom assemblage change throughout the entire sediment core. *Discotella stelligera* relative abundance was significantly correlated ($p = 6.3e-15$) with PC1 scores throughout the core (Fig.5). PC2 represents 25.8% of the variance in the diatom assemblage and shows that samples with higher *Fragillaria ulna* abundances often have lower *Aulacoseira subarctica* relative to the rest of the core.

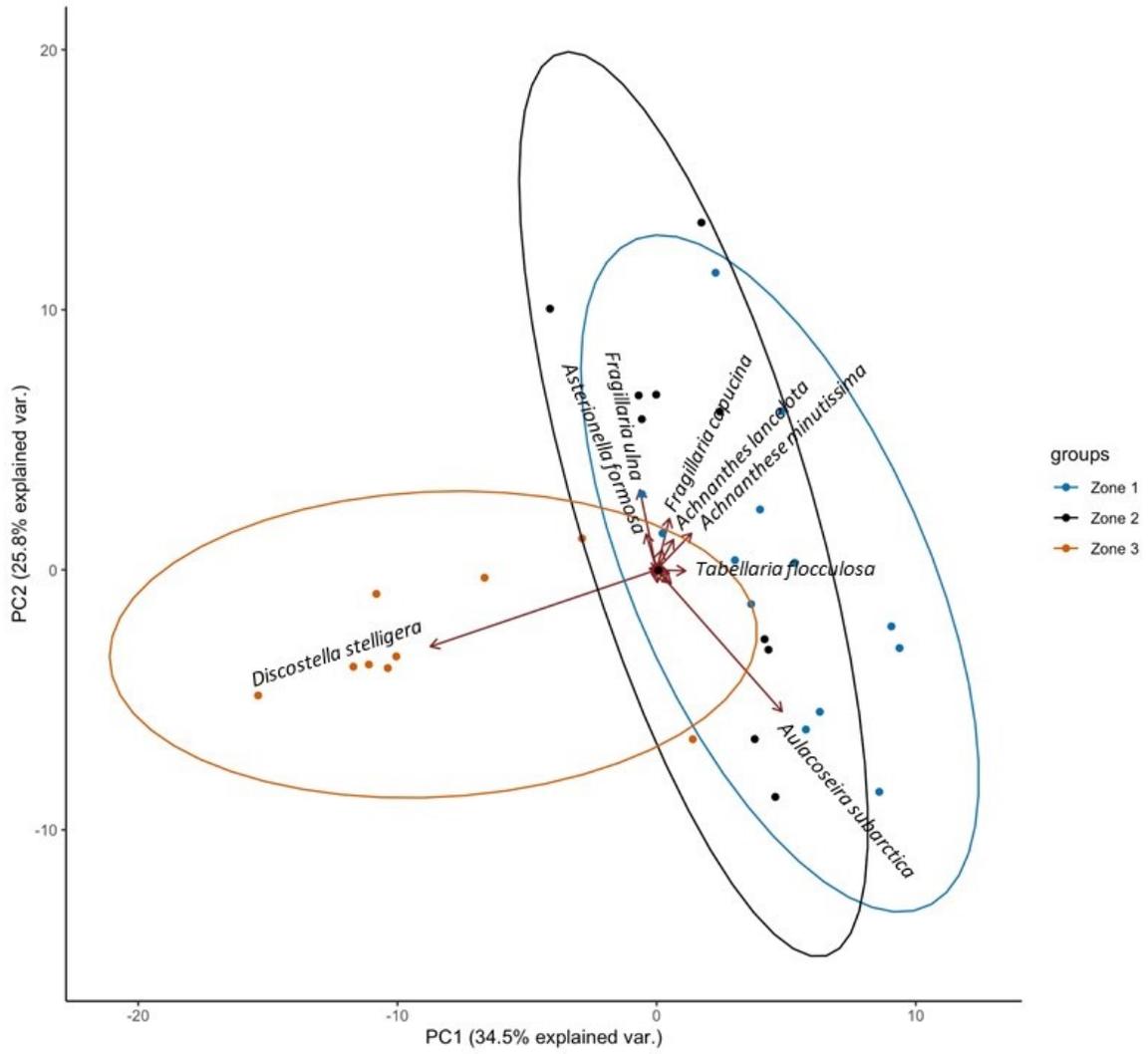


Figure 4. Principal component analysis of the Yellowknife Bay core diatom assemblages. The red points correspond to zone three from the stratigraphic plot, green points correspond to zone two, and blue corresponds to zone one.

Table 1. Eigenvalues for principal components 1 & 2 based on the principal component analysis of the Yellowknife Bay sediment core.

	PC1	PC2
<i>Aulacoseira subarctica</i>	0.473	-0.712
<i>Aulacoseira humilis</i>	0.00255	-0.0633
<i>Achnantheidium neomicrocephalum</i>	0.0481	-0.0178
<i>Asterionella formosa</i>	-0.0395	0.179
<i>Amphora pediculus</i>	-0.00598	0.0147
<i>Achnanthes lanceolata</i>	0.0634	0.152
<i>Achnanthes curtissima</i>	0.0547	-0.0689
<i>Achnanthes carissima</i>	0.00185	0.0249
<i>Achnanthese minutissima</i>	0.132	0.183
<i>Cymbella minuta</i>	0.00866	-0.0151
<i>Discostella stelligera</i>	-0.852	-0.386
<i>Cyclotella areolata</i>	0.0163	0.0358
<i>Cocconeis neothumensis</i>	0.0417	-0.0573
<i>Diploneis parma</i>	-0.00721	0.0274
<i>Fragillaria construens</i>	0.000359	-0.00767
<i>Fragillaria capucina</i>	0.0484	0.258
<i>Fragilaria brevistriata</i>	-0.00151	0.0198
<i>Fragillaria ulna</i>	-0.0634	0.401
<i>Gomphonema acuminatum</i>	0.0239	-0.0216
<i>Navicula lanceolata</i>	-0.00669	-0.0136
<i>Nitzchia palea</i>	0.0209	0.101
<i>Pinnularia cardinaliculus</i>	-0.0266	0.00685
<i>Tabellaria flocculosa</i>	0.109	-0.00517
<i>Stephanodiscus niagarae</i>	-0.00611	-0.0236

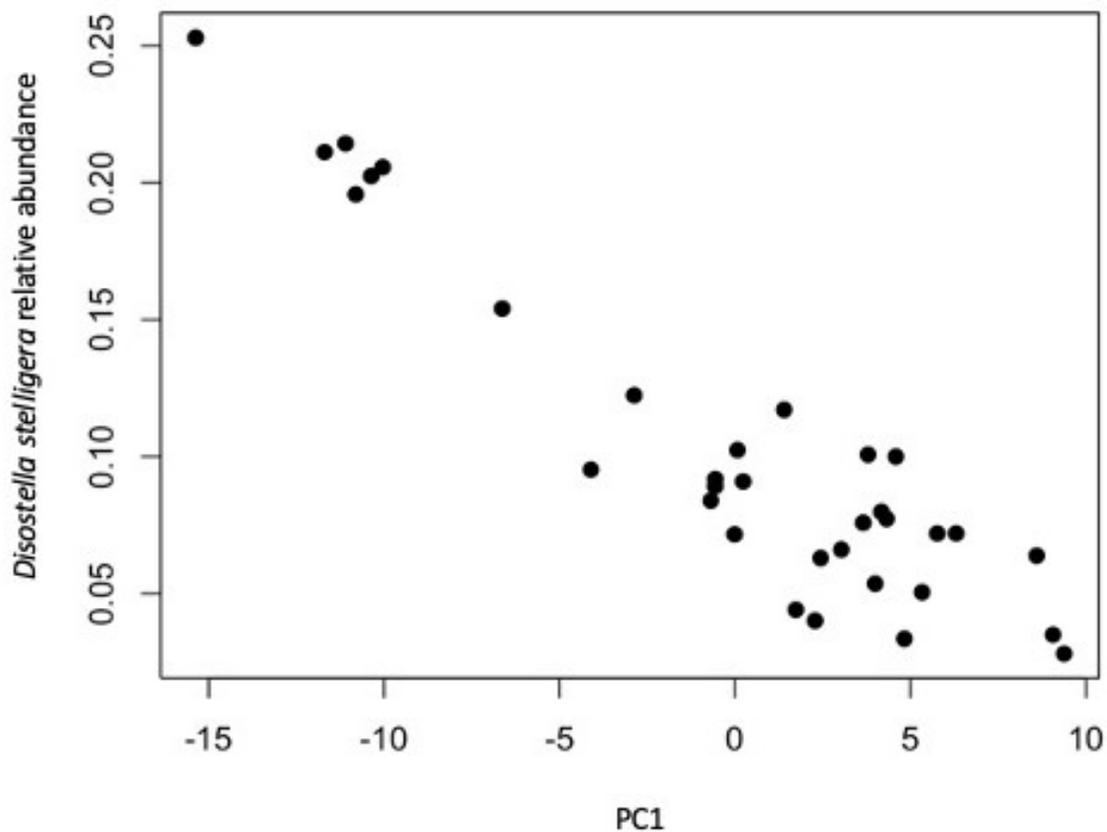


Figure 5. Pearson correlation between *Discostella stelligera* relative abundance and PC1 scores from principal component analysis. $R = -0.93$ and $p = 6.3e-15$.

Consistent climate data from Yellowknife were available from the late 1940s onwards (ECCC, 2018). Mean annual and seasonal 5-year moving average temperatures were shown to be increasing over the second half of the 20th century and into the 21st century (Fig.6). Spring temperatures saw the most notable increase when compared to summer, fall, and winter, however all seasons experienced overall increasing temperatures over the last ~70 years. Increases in overall annual and spring temperatures in Yellowknife corresponded to the same time period when relative abundances of *Discostella stelligera* sharply increased and PC1 scores decreased representing the largest variation in the overall diatom assemblage. The operational time period

of Giant Mine did not seem to affect *Discostella stelligera* relative abundances, however, it did seem to correspond to subtle dips when compared to the percent organic matter and chlorophyll-*a* levels. As seen in the strat plot (Fig.3), organic matter was highest in recent intervals where *Discostella stelligera* was also highest. These increases in organic matter occur following annual warming in the later 1990's based on the CRS dates. Chlorophyll-*a* begins to rise in the 1990's but then dips following 2000.

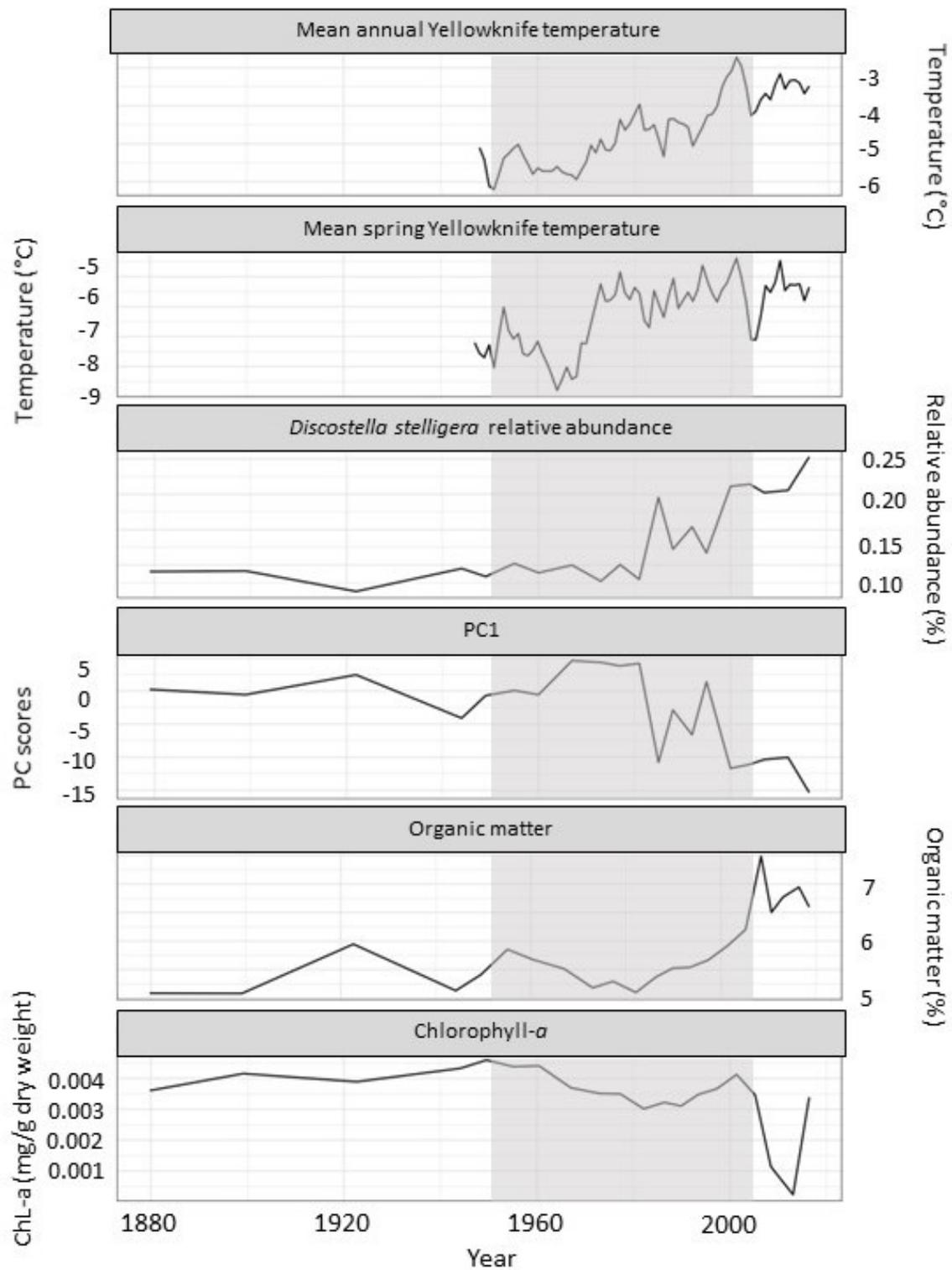


Figure 6. Comparison of the 5-year moving average annual and spring mean temperatures from Yellowknife with *Discostella stelligera* relative abundance, PC 1 scores, organic matter, and Chlorophyll-*a* overtime. The grey shaded box represents the operational period of Giant Mine.

4.2 Discussion

Until about the 1980's, Yellowknife Bay's diatom assemblage underwent relatively modest change for the most part of the 19th and 20th centuries. In more recent years, however, more pronounced changes have occurred as this core recorded a shift towards a different diatom assemblage that now includes higher relative abundances of planktonic *Discotella stelligera*, which were present only in small amounts (~5% relative abundance) in previous sediment intervals. With this expansion of planktonic diatoms, this study also recorded slight decreases in tychoplanktonic *Aulacoseira subarctica* and benthic *Fragillaria* taxa. These findings are consistent with results from other recent diatom studies in the region and across the northern hemisphere; all of which have predominantly observed distinct diatom assemblage shifts from benthic to planktonic species (Rühland et al., 2008; Rühland & Smol, 2005; Rühland & Smol, 1998; Sivarajah et al., 2020; Smol et al., 2005; Sorvari et al., 2002).

High-latitude lakes in the Arctic and subarctic generally have ecosystems with low species diversity, low productivity, simple food webs, and short open water seasons with their water column and biological life closely connected with the atmosphere (Sorvari et al., 2002). These factors mean that arctic and subarctic waters and their biota are vulnerable to changes in climate (Douglas et al., 1994; Sorvari et al., 2002). Climate change is now rapidly impacting northern lakes causing shorter periods of lake-ice cover, longer growing season, and changes in thermal stratification patterns; all of these changes are known to affect many limnological factors important to planktonic organisms, including nutrients, oxygen, light, and pH (Rühland & Smol, 2005). Although the exact mechanism behind this widespread shift to planktonic diatoms in northern lakes is still unclear, many studies argue that it is likely related to these limnological changes driven by climate change (Rühland & Smol, 2005; Sorvari et al., 2002). Shorter growing

seasons and cooler conditions typically favour benthic and tychoplanktonic diatoms as they are able to grow in the limited open-water zones along the shallow margins of the lakes whereas the strengthening of thermal stratification may greatly favour the expansion of planktonic diatoms in deeper subarctic lakes (Rühland & Smol, 2005).

Previous studies have shown that smaller northern lakes also display a more abrupt decrease in benthic species whereas larger isothermal lakes have been experiencing more subtle shifts (Rühland & Smol, 2005; Sorvari et al., 2002). This relates to the findings of our study as benthic and tychoplanktonic species did not completely disappear with the sudden increase of *Discotella stelligera*, but rather began a subtle decline starting in the late 20th century (Fig.3). Considering the large size of Great Slave Lake, it makes sense that there would be a more subtle decline of previously dominant diatoms, even if major limnological changes are occurring.

Increases in *Discotella stelligera* since the 1980s in Yellowknife Bay sediments suggest that the thermal properties and mixing conditions of Yellowknife Bay have been changing in recent years due to shorter duration of seasonal ice-cover and longer periods of thermal stratification. This sharp increase in *Discotella stelligera* also corresponds to warmer overall annual temperatures and warmer spring seasonal temperatures which would be key drivers for increasing the duration of thermal stratification in Yellowknife Bay. Warmer springs in particular could greatly lengthen the ice-free season which would then contribute to a longer thermally stratified summer period. Higher temperatures in summer and more stable stratification in lakes around the World have also been predicted to lead to increased harmful algal blooms (Free et al., 2021), which could further contribute to the reduction in deep water oxygen levels in Yellowknife Bay and promote arsenic diffusion from sediments into overlying surface waters. Conversely, in reducing settings, arsenic can be sequestered by sulphides (Miller et al., 2021).

The Yellowknife Bay core showed relatively steady levels of organic matter until the later half of the 20th century, when gradual and steady increase began (Fig.2). Chlorophyll-*a* is also often studied as a proxy of phytoplankton biomass (Free et al., 2021). Interestingly, chlorophyll-*a* levels remained fairly consistent throughout the core, until a decline was observed in the uppermost intervals of the layer (Fig.2). There was also a slight decline around the same time period in organic matter; it's possible that this sudden and sharp decline in chlorophyll-*a* could have had to do with 2004 being a colder year overall (-6.5 °C) compared to the mean annual temperatures of other years since the 1990's. Overall, organic matter and chlorophyll-*a* levels remained generally low overtime (maximum recorded level of 0.004 mg/g), similar to the findings of Stoermer et al. (1990), which is what one would expect to see in such a large oligotrophic lake in the subarctic. In comparison, Nivan lake near Yellowknife, which experienced sewage input from the city, had chlorophyll-*a* levels of 0.11 mg/g within its sediments (Stewart et al., 2018).

Regarding legacy arsenic contamination within Yellowknife Bay sediments, our results indicate that environmental conditions are changing in a way which could further promote the release of arsenic contamination from lake sediments into overlying water and/or sequester arsenic into sulphides. Lake stratification has a decisive impact on the redistribution of dissolved substances, such as nutrients and oxygen, and longer/stronger thermal stratification will inhibit the circulation of these substances from the epilimnion and atmosphere to the hypolimnion (Boehrer & Schultze, 2008). If longer and stronger periods of thermal stratification are occurring in Yellowknife Bay, as our diatom indicators suggest, then this would contribute to the reduction of dissolved oxygen levels in the deep waters of the bay near the sediment-water interface. Organic matter overtime has gradually increased as well, which may result in more

decomposition in the hypolimnion which ultimately could also contribute to decreasing oxygen levels in deep water however overall organic matter levels still remain relatively low. The chemical behaviour of arsenic is sensitive to the presence of oxygen and in the absence of oxygen, mineral-bound arsenic in sediment can dissolve and diffuse upwards to overlying water (Andrade et al., 2010; Martin & Pedersen, 2002; McDonald et al., 2015; Palmer et al., 2020) or become bound to sulphides and this can be an effective means of sequestering arsenic into sediments (Miller et al., 2021). As Andrade et al. (2010) suggests, a seasonal monitoring program for both the water column and sediments may be warranted due to the possible increase of arsenic diffusion moving into the future as a result of climate change impacts on the limnology of Yellowknife Bay.

Chapter 5.0 Conclusion

This thesis investigated the impacts of multiple stressors on the water quality in Yellowknife Bay, Northwest Territories. Yellowknife has undergone climate warming over the last few decades however there is very little to no limnological monitoring data within Yellowknife Bay in Great Slave Lake. The results of this study are evidence that Yellowknife Bay is being impacted by climate change and is undergoing limnological changes based on a shift towards *Discotella stelligera* becoming the most dominant diatom species within the Yellowknife Bay diatom assemblage. Yellowknife Bay sediments contain high amounts of legacy arsenic contamination as a result of gold mining activities which took place in the region throughout the second half of the 20th century. Giant mine alone contributed to a massive amount of the arsenic contamination that is still found across the landscape and in lakes to this day. The primary objective of this thesis was to assess if climate change has been impacting Yellowknife Bay in a way that could control the behaviour of legacy arsenic contamination within the bay. This was done by studying changes in the diatom assemblage and productivity of the bay over the last century in order to infer historical environmental conditions.

This investigation was completed through the analysis of a sediment core that was analyzed in Chapter 3.0 to determine background conditions and assess long-term environmental change in Yellowknife Bay. Historically, the older sections of the core were mainly dominated by the tychoplanktonic *Aulacoseira subarctica*. The diatom assemblage remained relatively unchanged overtime until the late 20th century, when a large increase of the planktonic *Discotella stelligera* was observed, overtaking the *Aulacoseira subarctica* as the dominant species in the assemblages, and remaining the most abundant species until present day. *Aulacoseira subarctica* abundances still remained relatively high in the assemblage over this time compared to all other

diatom species found, however when compared to the older intervals within the core, its relative abundance decreased and was lowest in the most recent intervals. This sharp increase in *Discostella stelligera* in diatom assemblages recorded in Yellowknife Bay has also been observed in various other studies within the Canadian subarctic and the northern hemisphere. Organic matter was also seen to gradually increase around similar periods when compared to *Discostella stelligera* increases and annual temperature increases. Organic matter still remained fairly low as would be expected from a deep, oligotrophic lake in the subarctic. This was the first study of this kind in Yellowknife Bay within Great Slave Lake, which is overall a much bigger and deeper lake than previous study lakes analyzing diatoms in the subarctic around Yellowknife. The results from this study are representative of a sheltered bay within Great Slave lake, however, given how large of a lake it is, different areas within the lake will respond differently to climate change. Results may differ from Yellowknife Bay compared to the deep open water areas of the main lake body, the shallower waters of Great Slave Lake's North Arm, or from the +600m basin in the east arm of the lake. Similar paleolimnological studies in different regions of Great Slave Lake would provide further understanding about how climate change is impacting large, deep subarctic lakes.

The mechanism for this shift in diatom assemblage in northern lakes has been of interest to many researchers as this is most likely an indication of widespread, global change that is impacting even the most remote high-latitude lakes that have never directly been exposed to specific anthropogenic stresses. Because of this, it is widely accepted that this specific diatom shift in many northern hemisphere lakes is the result of increased duration and strength of thermal stratification caused by climate change through warmer air temperatures causing longer ice-free periods on lakes, warming up the surface waters more over the course of the year and

therefore making them stratified. The lengthening duration and strength of stratification in deep subarctic lakes can have negative impacts on these natural ecosystems, such as disturbing fish habitat for cold water species such as lake trout, but it can also have negative impacts on the health of impacted aquatic ecosystems where arsenic within lake sediments is high, such as in Yellowknife Bay, as arsenic is a redox sensitive element and can diffuse from sediments into lake water in low oxygen situations.

Yellowknife Bay is a fascinating example of a water-body that has undergone multiple stressors which have cumulative impacts on the ecosystem. This study has explored the link between the global issue of climate change and its connection and influence on anthropogenic disturbances in lakes, such as legacy arsenic contamination from mining operations. This study builds on the knowledge of previous research in the area, and provides evidence that changes are occurring within Yellowknife Bay. This information is important for stakeholders and decision makers who are involved in the environmental management of this important waterbody for residents of N'dilo, Detah, and Yellowknife. As mentioned earlier, further paleolimnological studies within Great Slave Lake and other deep subarctic lake would further increase our understanding of climate change impacts on deep lakes in the subarctic. Future limnological studies in Yellowknife Bay could also measure and monitor lake properties, such as dissolved oxygen and temperature, along with arsenic levels in water and lake sediments at a high temporal resolution in order to further understand and observe climate change impacts on the bay (ie. what is the strength and duration of stratification in the bay and how does it affect oxygen levels in the hypolimnion?) and to better understand arsenic mobility within a deep subarctic lake undergoing changes.

Chapter 6.0 References

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Appendices

Table A-1 Relative abundances of diatom species from the Yellowknife Bay core.

Depth	<i>Aulacoseira subarctica</i>	<i>Aulacoseira humilis</i>	<i>Aulaoseira nygaardii</i>
0.5	0.155452436	0.044083527	0.004640371
1.5	0.172897196	0.01635514	0.007009346
2.5	0.161445783	0.024096386	0
3.5	0.152073733	0	0
4.5	0.148058252	0.055825243	0
5.5	0.243559719	0	0
6.5	0.132701422	0.021327014	0
7.5	0.164705882	0.021176471	0.009411765
8.5	0.140811456	0	0
9.5	0.232394366	0.023474178	0.014084507
10.5	0.270983213	0.033573141	0
11.5	0.229508197	0.042154567	0.004683841
12.5	0.295348837	0	0
13.5	0.164319249	0.009389671	0.018779343
14.5	0.207142857	0.016666667	0
15.5	0.160839161	0.013986014	0.006993007
16.5	0.085714286	0	0
17.5	0.124105012	0.040572792	0.00477327
18.5	0.172494172	0.002331002	0
19.5	0.116009281	0.006960557	0.004640371
20.5	0.152173913	0	0
21.5	0.103773585	0.007075472	0
22.5	0.167464115	0	0
23.5	0.177156177	0.027972028	0
24.5	0.181603774	0	0
25.5	0.153110048	0.019138756	0.004784689
26.5	0.201421801	0.026066351	0
27.5	0.209134615	0.048076923	0
28.5	0.258992806	0	0
29.5	0.244755245	0.025641026	0.006993007
30.5	0.281323877	0.028368794	0
31.5	0.242990654	0.056074766	0.002336449
32.5	0.229698376	0	0

Depth	<i>Achnanthydium neomicrocephalum</i>	<i>Asterionella formosa</i>	<i>Amphora copulata</i>
0.5	0.030162413	0.062645012	0.004640371
1.5	0.018691589	0.037383178	0
2.5	0.028915663	0.053012048	0.009638554
3.5	0.036866359	0.050691244	0
4.5	0.041262136	0.019417476	0.004854369
5.5	0.030444965	0.051522248	0
6.5	0.035545024	0.016587678	0
7.5	0.047058824	0.032941176	0.004705882
8.5	0.071599045	0.035799523	0
9.5	0.039906103	0.030516432	0.014084507
10.5	0.028776978	0.014388489	0
11.5	0.025761124	0.051522248	0.009367681
12.5	0.058139535	0.088372093	0
13.5	0.030516432	0.08685446	0
14.5	0.030952381	0.078571429	0.004761905
15.5	0.03962704	0.065268065	0.002331002
16.5	0.05	0.047619048	0.002380952
17.5	0.033412888	0.040572792	0.009546539
18.5	0.032634033	0.037296037	0.013986014
19.5	0.044083527	0.118329466	0.004640371
20.5	0.04589372	0.043478261	0
21.5	0.04009434	0.04245283	0
22.5	0.033492823	0.062200957	0.019138756
23.5	0.020979021	0.06993007	0
24.5	0.066037736	0.066037736	0
25.5	0.035885167	0.04784689	0.004784689
26.5	0.021327014	0.037914692	0
27.5	0.055288462	0.045673077	0.014423077
28.5	0.057553957	0.007194245	0
29.5	0.055944056	0.013986014	0
30.5	0.042553191	0.026004728	0
31.5	0.060747664	0.01635514	0.004672897
32.5	0.060324826	0.018561485	0

Depth	<i>Amphora pediculus</i>	<i>Achnanthes lanceolata</i>	<i>Achnanthes curtissima</i>
0.5	0.006960557	0.051044084	0.025522042
1.5	0	0.046728972	0.030373832
2.5	0.01686747	0.045783133	0.026506024
3.5	0	0.052995392	0.023041475
4.5	0.009708738	0.033980583	0.026699029
5.5	0	0.06323185	0.037470726
6.5	0.014218009	0.052132701	0.047393365
7.5	0.011764706	0.105882353	0.023529412
8.5	0.019093079	0.07875895	0.033412888
9.5	0.011737089	0.079812207	0.016431925
10.5	0.004796163	0.076738609	0.047961631
11.5	0.004683841	0.044496487	0.018735363
12.5	0	0.053488372	0.03255814
13.5	0	0.056338028	0.021126761
14.5	0.00952381	0.057142857	0.023809524
15.5	0.013986014	0.048951049	0.025641026
16.5	0	0.069047619	0.030952381
17.5	0.023866348	0.069212411	0.045346062
18.5	0.013986014	0.053613054	0.032634033
19.5	0.002320186	0.071925754	0.025522042
20.5	0	0.137681159	0.031400966
21.5	0	0.122641509	0.037735849
22.5	0	0.081339713	0.038277512
23.5	0.016317016	0.083916084	0.025641026
24.5	0	0.049528302	0.028301887
25.5	0.014354067	0.066985646	0.033492823
26.5	0.007109005	0.068720379	0.056872038
27.5	0.009615385	0.072115385	0.016826923
28.5	0	0.064748201	0.028776978
29.5	0.016317016	0.074592075	0.023310023
30.5	0.004728132	0.054373522	0.040189125
31.5	0.014018692	0.065420561	0.03271028
32.5	0	0.03712297	0.134570766

Depth	<i>Achnanthes carissima</i>	<i>Achnanthese minutissima</i>	<i>Cymbella arctica</i>
0.5	0.013921114	0.032482599	0
1.5	0.014018692	0.081775701	0
2.5	0.024096386	0.065060241	0.002409639
3.5	0.011520737	0.099078341	0.002304147
4.5	0.024271845	0.050970874	0.004854369
5.5	0.009367681	0.081967213	0.016393443
6.5	0.021327014	0.071090047	0.002369668
7.5	0.032941176	0.089411765	0.002352941
8.5	0.016706444	0.066825776	0.007159905
9.5	0.016431925	0.107981221	0.004694836
10.5	0.023980815	0.095923261	0.004796163
11.5	0.016393443	0.088992974	0.004683841
12.5	0.009302326	0.074418605	0.018604651
13.5	0.023474178	0.08685446	0.004694836
14.5	0.014285714	0.066666667	0
15.5	0.025641026	0.074592075	0
16.5	0.016666667	0.138095238	0.002380952
17.5	0.028639618	0.100238663	0.007159905
18.5	0.009324009	0.121212121	0.006993007
19.5	0.018561485	0.141531323	0
20.5	0.007246377	0.086956522	0.002415459
21.5	0.028301887	0.103773585	0.002358491
22.5	0.0215311	0.088516746	0.009569378
23.5	0.020979021	0.076923077	0
24.5	0.021226415	0.075471698	0.004716981
25.5	0.009569378	0.088516746	0.002392344
26.5	0.028436019	0.092417062	0.002369668
27.5	0.024038462	0.079326923	0
28.5	0.014388489	0.107913669	0.007194245
29.5	0.016317016	0.093240093	0
30.5	0.01891253	0.096926714	0
31.5	0.023364486	0.060747664	0.004672897
32.5	0.018561485	0.102088167	0

Depth	<i>Cymbella minuta</i>	<i>Discostella stelligera</i>	<i>Cyclotella areolata</i>
0.5	0.016241299	0.252900232	0.009280742
1.5	0.025700935	0.205607477	0.002336449
2.5	0.021686747	0.202409639	0
3.5	0.036866359	0.214285714	0.002304147
4.5	0.012135922	0.211165049	0
5.5	0.018735363	0.117096019	0
6.5	0.059241706	0.154028436	0
7.5	0.042352941	0.122352941	0.009411765
8.5	0.021479714	0.195704057	0.002386635
9.5	0.035211268	0.079812207	0.007042254
10.5	0.019184652	0.100719424	0
11.5	0.021077283	0.077283372	0
12.5	0.011627907	0.1	0
13.5	0.016431925	0.089201878	0.002347418
14.5	0.019047619	0.102380952	0
15.5	0.025641026	0.083916084	0.009324009
16.5	0.019047619	0.095238095	0.007142857
17.5	0.026252983	0.071599045	0.00477327
18.5	0.018648019	0.062937063	0
19.5	0.018561485	0.044083527	0.004640371
20.5	0.04589372	0.09178744	0
21.5	0.023584906	0.04009434	0.009433962
22.5	0.023923445	0.090909091	0
23.5	0.027972028	0.053613054	0.011655012
24.5	0.025943396	0.066037736	0
25.5	0.016746411	0.033492823	0.019138756
26.5	0.047393365	0.075829384	0
27.5	0.021634615	0.050480769	0.021634615
28.5	0.035971223	0.071942446	0
29.5	0.041958042	0.034965035	0.006993007
30.5	0.021276596	0.063829787	0
31.5	0.01635514	0.028037383	0.014018692
32.5	0.044083527	0.071925754	0

Depth	<i>Cyclotella bodanica</i>	<i>Cocconeis neothumensis</i>	<i>Diploneis elliptica</i>
0.5	0	0.013921114	0
1.5	0	0.03271028	0.004672897
2.5	0	0.024096386	0.002409639
3.5	0	0.032258065	0.004608295
4.5	0	0.031553398	0.007281553
5.5	0	0.056206089	0.007025761
6.5	0	0.028436019	0.002369668
7.5	0.002352941	0.032941176	0.002352941
8.5	0	0.050119332	0
9.5	0	0.028169014	0.002347418
10.5	0	0.019184652	0.004796163
11.5	0.004683841	0.028103044	0.00234192
12.5	0	0.034883721	0.009302326
13.5	0	0.023474178	0.009389671
14.5	0	0.035714286	0.002380952
15.5	0	0.020979021	0.002331002
16.5	0	0.045238095	0
17.5	0	0.028639618	0
18.5	0	0.034965035	0
19.5	0	0.009280742	0.002320186
20.5	0	0.053140097	0
21.5	0	0.011792453	0.004716981
22.5	0	0.033492823	0.004784689
23.5	0	0.027972028	0.006993007
24.5	0	0.063679245	0
25.5	0	0.045454545	0
26.5	0	0.033175355	0
27.5	0	0.03125	0.012019231
28.5	0	0.021582734	0
29.5	0	0.048951049	0.006993007
30.5	0	0.052009456	0
31.5	0	0.044392523	0.002336449
32.5	0	0.039443155	0

Depth	<i>Diploneis parma</i>	<i>Eunotia parallela</i>	<i>Fragillaria construens</i>
0.5	0.011600928	0.002320186	0.002320186
1.5	0.018691589	0.007009346	0.004672897
2.5	0.012048193	0.01686747	0.021686747
3.5	0.004608295	0	0.006912442
4.5	0.007281553	0.016990291	0.004854369
5.5	0.004683841	0.007025761	0.018735363
6.5	0.004739336	0.004739336	0.002369668
7.5	0.004705882	0.004705882	0.002352941
8.5	0.00477327	0.011933174	0.00477327
9.5	0.011737089	0.002347418	0.007042254
10.5	0.002398082	0.011990408	0.002398082
11.5	0.009367681	0.011709602	0.00234192
12.5	0	0.011627907	0.009302326
13.5	0.011737089	0.014084507	0.007042254
14.5	0.007142857	0.007142857	0.004761905
15.5	0.006993007	0.016317016	0.006993007
16.5	0.00952381	0	0.030952381
17.5	0.007159905	0	0.00477327
18.5	0.004662005	0.004662005	0.002331002
19.5	0	0.006960557	0.002320186
20.5	0.009661836	0.007246377	0.014492754
21.5	0.023584906	0.007075472	0.011792453
22.5	0.011961722	0.004784689	0.007177033
23.5	0.006993007	0.013986014	0.004662005
24.5	0	0.011792453	0.009433962
25.5	0.016746411	0.007177033	0
26.5	0.011848341	0.002369668	0.004739336
27.5	0.014423077	0.002403846	0.004807692
28.5	0	0.007194245	0.014388489
29.5	0.009324009	0.002331002	0.004662005
30.5	0.004728132	0.011820331	0.014184397
31.5	0.01635514	0	0.004672897
32.5	0	0.002320186	0.020881671

Depth	<i>Fragillaria capucina</i>	<i>Fragilaria parasitica</i>	<i>Fragilaria elliptica</i>
0.5	0.034802784	0.011600928	0
1.5	0.037383178	0.011682243	0.002336449
2.5	0.021686747	0.014457831	0
3.5	0.011520737	0.006912442	0
4.5	0.033980583	0.004854369	0
5.5	0.018735363	0.011709602	0
6.5	0.033175355	0.016587678	0
7.5	0.04	0.002352941	0
8.5	0.009546539	0	0
9.5	0.049295775	0.004694836	0
10.5	0.019184652	0.007194245	0
11.5	0.06088993	0.014051522	0
12.5	0.034883721	0.006976744	0
13.5	0.098591549	0.004694836	0
14.5	0.076190476	0.004761905	0
15.5	0.097902098	0.002331002	0
16.5	0.014285714	0.014285714	0
17.5	0.090692124	0.011933174	0
18.5	0.055944056	0.018648019	0
19.5	0.090487239	0.011600928	0
20.5	0.014492754	0.007246377	0
21.5	0.073113208	0.007075472	0
22.5	0.009569378	0.007177033	0
23.5	0.032634033	0.006993007	0
24.5	0.007075472	0.014150943	0
25.5	0.026315789	0	0
26.5	0.035545024	0.014218009	0.002369668
27.5	0.043269231	0.007211538	0
28.5	0.009592326	0	0
29.5	0.044289044	0.004662005	0
30.5	0.023640662	0.007092199	0
31.5	0.039719626	0	0
32.5	0.016241299	0.013921114	0

Depth	<i>Fragilaria brevistriata</i>	<i>Fragillaria ulna</i>	<i>Gomphonema acuminatum</i>
0.5	0.006960557	0.067285383	0.002320186
1.5	0.018691589	0.074766355	0
2.5	0.024096386	0.053012048	0.009638554
3.5	0.023041475	0.041474654	0
4.5	0.019417476	0.065533981	0.007281553
5.5	0.004683841	0.046838407	0.018735363
6.5	0	0.052132701	0.007109005
7.5	0.011764706	0.070588235	0.002352941
8.5	0.026252983	0.088305489	0
9.5	0.014084507	0.044600939	0
10.5	0.023980815	0.079136691	0.002398082
11.5	0.018735363	0.037470726	0.014051522
12.5	0.018604651	0.037209302	0.006976744
13.5	0.011737089	0.103286385	0.002347418
14.5	0.021428571	0.076190476	0.007142857
15.5	0.016317016	0.132867133	0
16.5	0.016666667	0.119047619	0.00952381
17.5	0.023866348	0.059665871	0.002386635
18.5	0.025641026	0.13986014	0.004662005
19.5	0.013921114	0.120649652	0.004640371
20.5	0.028985507	0.062801932	0.002415459
21.5	0.025943396	0.091981132	0.004716981
22.5	0.026315789	0.066985646	0.004784689
23.5	0.006993007	0.055944056	0.009324009
24.5	0.021226415	0.063679245	0.011792453
25.5	0.026315789	0.105263158	0.007177033
26.5	0.016587678	0.049763033	0.004739336
27.5	0.012019231	0.064903846	0.007211538
28.5	0.052757794	0.059952038	0.026378897
29.5	0.004662005	0.062937063	0.004662005
30.5	0.009456265	0.01891253	0.009456265
31.5	0.01635514	0.058411215	0.002336449
32.5	0	0.016241299	0.002320186

Depth	<i>Navicula radiosa</i>	<i>Navicula submularis</i>	<i>Navicula laevis</i>
0.5	0	0	0
1.5	0	0	0
2.5	0.002409639	0	0
3.5	0	0	0
4.5	0.004854369	0	0
5.5	0	0	0
6.5	0.002369668	0	0
7.5	0	0	0
8.5	0	0	0
9.5	0.002347418	0	0
10.5	0	0	0
11.5	0	0	0
12.5	0	0	0
13.5	0	0	0
14.5	0	0	0
15.5	0.002331002	0	0
16.5	0	0	0
17.5	0.002386635	0	0
18.5	0	0	0
19.5	0.004640371	0	0
20.5	0	0	0
21.5	0.002358491	0	0
22.5	0	0	0
23.5	0.006993007	0	0
24.5	0	0	0
25.5	0	0	0
26.5	0	0	0
27.5	0	0	0
28.5	0	0	0.002398082
29.5	0	0	0
30.5	0	0	0
31.5	0.002336449	0	0
32.5	0	0	0

Depth	<i>Navicula minuscula</i>	<i>Navicula lanceolata</i>	<i>Nitzchia radiosa</i>
0.5	0	0.011600928	0
1.5	0	0.014018692	0
2.5	0	0.007228916	0
3.5	0	0.029953917	0
4.5	0	0.016990291	0
5.5	0	0.021077283	0
6.5	0	0.030805687	0
7.5	0	0.018823529	0.002352941
8.5	0	0.016706444	0
9.5	0	0.016431925	0.002347418
10.5	0.002398082	0.011990408	0
11.5	0.00234192	0.011709602	0.004683841
12.5	0	0.020930233	0
13.5	0	0.007042254	0
14.5	0	0.021428571	0
15.5	0	0.009324009	0
16.5	0	0.016666667	0
17.5	0	0.021479714	0
18.5	0.004662005	0.023310023	0
19.5	0	0.006960557	0.002320186
20.5	0	0.016908213	0
21.5	0	0.009433962	0
22.5	0	0.028708134	0
23.5	0	0.018648019	0
24.5	0	0.011792453	0.002358491
25.5	0	0.0215311	0
26.5	0	0.002369668	0
27.5	0	0.014423077	0.002403846
28.5	0	0.021582734	0
29.5	0	0.013986014	0
30.5	0.007092199	0.009456265	0
31.5	0	0.028037383	0
32.5	0	0.011600928	0

Depth	<i>Nitzchia palea</i>	<i>Pinnularia cardinaliculus</i>	<i>Pleurosigma elongatum</i>
0.5	0.016241299	0	0.006960557
1.5	0.028037383	0.004672897	0.011682243
2.5	0.019277108	0.004819277	0.002409639
3.5	0.016129032	0.013824885	0.016129032
4.5	0.016990291	0.009708738	0.007281553
5.5	0.009367681	0.00234192	0.004683841
6.5	0.030805687	0.007109005	0.016587678
7.5	0.018823529	0	0.018823529
8.5	0.014319809	0.007159905	0.002386635
9.5	0.03286385	0.007042254	0.009389671
10.5	0.019184652	0.016786571	0.002398082
11.5	0.028103044	0.004683841	0.009367681
12.5	0.027906977	0	0
13.5	0.030516432	0.002347418	0.004694836
14.5	0.04047619	0.004761905	0.004761905
15.5	0.032634033	0.006993007	0.009324009
16.5	0.052380952	0.028571429	0.004761905
17.5	0.040572792	0.002386635	0.007159905
18.5	0.025641026	0.002331002	0.006993007
19.5	0.03712297	0	0.004640371
20.5	0.033816425	0.014492754	0.002415459
21.5	0.033018868	0	0.007075472
22.5	0.0215311	0.002392344	0.004784689
23.5	0.027972028	0.002331002	0.016317016
24.5	0.030660377	0.004716981	0.004716981
25.5	0.023923445	0	0.004784689
26.5	0.014218009	0	0.014218009
27.5	0.03125	0.004807692	0.009615385
28.5	0.023980815	0.007194245	0.002398082
29.5	0.027972028	0	0.011655012
30.5	0.016548463	0.002364066	0.002364066
31.5	0.018691589	0	0.002336449
32.5	0.025522042	0.002320186	0.018561485

Depth	<i>Tabellaria flocculosa</i>	<i>Strauroneis phoenicenteron</i>	<i>Stephanodiscus niagarae</i>
0.5	0.071925754	0	0.002320186
1.5	0.049065421	0.002336449	0.009345794
2.5	0.065060241	0	0.007228916
3.5	0.082949309	0	0.01843318
4.5	0.077669903	0	0.02184466
5.5	0.070257611	0	0.023419204
6.5	0.106635071	0	0.023696682
7.5	0.030588235	0	0.009411765
8.5	0.033412888	0	0.028639618
9.5	0.03286385	0	0.009389671
10.5	0.035971223	0	0.009592326
11.5	0.084309133	0	0.007025761
12.5	0.01627907	0	0.011627907
13.5	0.042253521	0	0.009389671
14.5	0.026190476	0	0.014285714
15.5	0.027972028	0.002331002	0.004662005
16.5	0.047619048	0	0.021428571
17.5	0.050119332	0.002386635	0.00477327
18.5	0.051282051	0	0.013986014
19.5	0.053364269	0	0.002320186
20.5	0.04589372	0	0.036231884
21.5	0.101415094	0	0.011792453
22.5	0.098086124	0	0.028708134
23.5	0.116550117	0	0.020979021
24.5	0.108490566	0	0.037735849
25.5	0.14354067	0	0.014354067
26.5	0.111374408	0	0.014218009
27.5	0.057692308	0	0.002403846
28.5	0.064748201	0.002398082	0.023980815
29.5	0.083916084	0	0.006993007
30.5	0.104018913	0	0.028368794
31.5	0.109813084	0	0.004672897
32.5	0.081206497	0.002320186	0.016241299

Depth	Unkown
0.5	0.00232019
1.5	0.00934579
2.5	0.00963855
3.5	0.00921659
4.5	0.00242718
5.5	0.00468384
6.5	0.00236967
7.5	0.00470588
8.5	0.01193317
9.5	0.00938967
10.5	0.00719425
11.5	0.00468384
12.5	0.01162791
13.5	0.00704225
14.5	0.01428571
15.5	0.00466201
16.5	0.00476191
17.5	0.00954654
18.5	0.002331
19.5	0.00464037
20.5	0.00483092
21.5	0.01179245
22.5	0.00239234
23.5	0.00466201
24.5	0.01179245
25.5	0.00717703
26.5	0.00236967
27.5	0.00961539
28.5	0.00479616
29.5	0.00699301
30.5	0
31.5	0.00700935
32.5	0.01392111