Biogeochemistry, Limnology, and Ecology of Arctic Lakes

by

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Supervisory Committee

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Abstract

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Accelerated warming of high latitude systems of the northern hemisphere is expected to cause significant changes to the hydro-ecology of Arctic lakes. To record comprehensive and meaningful baseline hydrological, limnological, and ecological conditions to which future change can be compared, all available environmental information generated on Noell Lake, NWT was compiled and synthesized. Data included: physical and geographical characteristics (bathymetric and drainage basin attributes); general regional climatology; water quality (nutrients, major anions/cations, dissolved oxygen, dissolved organic carbon); biological composition (fish community, macrophyte, phytoplankton, epiphyton and epipelon surveys) and seasonal patterns in primary productivity (as measured by chlorophyll-*a* (Chl-*a*)).

A field-monitoring study was conducted from September 2010 to July 2013 assessing the application, reliability, and quality control/quality assurance of a newly developed automated buoy-based Arctic Lake Monitoring System (ALMS). The ALMS continuously measured a range of lake limnological and water quality parameters under both open-water and under-ice conditions. Overall, the ALMS provided a usable, uninterrupted record of changes in measured environmental, hydrological, and limnological parameters in both the epilimnion and hypolimnion. Noell Lake was determined to be spatially homogeneous with respect to the limnological measurements taken and, thus, the data recorded by the instrument arrays were determined to be representative of the lake as a whole.

In addition to the measurements made by environmental sensors mounted on the buoy and mooring components, an augmentary array of *in-situ* sampling campaigns and controlled experiments were conducted to produce a continuous and comprehensive description of daily and seasonal changes to the hydrological and limnological conditions of Noell Lake. The continuous data series confirmed that Noell Lake is dimictic, with mixing events occurring in August and June, and hypoxic oxygen conditions occurring in March. Nutrient limitation experiments revealed that autotrophic productivity in Noell Lake was nitrogen-limited.

Compiling data from existing literature involved >700 northern, high-latitude lakes; patterns in temporal and latitudinal changes in Arctic lake primary productivity (as measured by open-water, epilimnion Chl-*a*) and geochemistry were assessed. The key hypothesis tested was whether Arctic lakes are showing increased primary productivity (i.e., “greening”), through time and by latitude, similar to that documented for Arctic terrestrial systems. In general, significant decreases in lake Chl-*a* was observed in Arctic and sub-Arctic lakes over a ≈50 year time span. Separation of lakes by latitudinal bands revealed that trends in the lower Arctic region (60.00-69.99 Degrees North) showed a significant decreasing time trend, while high Arctic lakes displayed no trends. Corresponding temporal trends of total phosphorous (TP), total nitrogen (TN), and dissolved organic carbon (DOC) differed depending on the latitude of the lakes.

Re-evaluation of the original northern-lake productivity models developed by Flanagan et al. (2003) through the use of the new, independent datasets (>700 lakes) as well as the addition of other environmental variables (DOC, dissolved inorganic carbon, lake depth, conductivity, and ice-cover) showed that the original models were valid and the most parsimonious in predicting variation in algal biomass in northern latitude lakes. Only measures of dissolved nutrients (TP, TN) and latitude are required to predict autotrophic water column productivity.

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CHAPTER 1: INTRODUCTION

Comprising a substantial portion of the northern hemisphere, the Arctic is distinguished by several distinctive climatic, geological, and biophysical characteristics (AMAP, 1998; Wrona et al., 2006a; Jeffries et al., 2012; Larsen et al., 2014). These distinguishing characteristics include extreme seasonality, large disparities in temperature extremes, significant intra- and inter-annual variability in precipitation and temperature, large seasonal differences in summer and winter daylight, and steep latitudinal gradients in incident solar and ultra-violet radiation levels (Wrona et al., 2006a; Prowse et al., 2006a). The Arctic is underlain by vast areas of permafrost (Wrona et al., 2006a), defined as geological material that remains at 0 ^{o}C or below for at least two consecutive years (Anisimov and Nelson, 1996). Permafrost covers approximately 25% of the earth's surface and plays an important role in local hydrological cycles by limiting exchanges between surface and ground water (Prowse, 1990). In addition to permafrost - seasonal snow cover, glaciers, ice caps, ice sheets, river-, lake-, sea ice all constitute the cryosphere of the Arctic environment (Wrona et al., 2006a; Prowse et al., 2009).

Convincing evidence of increasing ambient air temperatures has been reported planet-wide and Arctic regions have been identified as being particularly susceptible to the impacts of climate warming (IPCC, 2013). The rate of warming in the Arctic has been more than double the global average during the past several decades (ACIA, 2005; Trenberth et al., 2007; AMAP, 2011; IPCC, 2013). Commencing in the 1800s, persistent warming has left the Arctic warmer than at any point in the preceding 2000 years (Kaufman *et al*., 2009; AMAP, 2011). Recorded Arctic surface air temperatures since 2004 are warmer than at any point previously in the historical instrumental record (AMAP, 2011). More specifically, North America's western Arctic has undergone some of the most significant warming on earth (Serreze et al., 2000). Corroboration by global circulation models suggests that the Arctic will continue to undergo the most severe warming (Flato et al., 2013).

The Arctic cryosphere is considered to be a particularly sensitive to the effects of a changing climate (e.g. Anisimov and Nelson, 1996; Holland and Bitz, 2003; Prowse et al., 2009) and documented changes to the various cryospheric components have already manifested themselves in several different ways: (1) longer open water seasons in rivers and lakes due to later ice formation and earlier break-up (AMAP, 2011); (2) decreases in the magnitude of snow coverage, especially in the spring and summer (Serreze et al., 2000; Dye, 2002; Hinzman et al., 2005); (3) an intensification of the hydrological cycle contributing to greater levels of precipitation; (4) decreases in the extent and overall disappearance of low-lying glaciers and ice caps (Serreze et al., 2000; Hinzman et al., 2005); (5) reductions in sea-ice thickness and extent (Vinnikov et al., 1999; Rothrock et al., 2003; Serreze et al., 2003), (6) degradation of permafrost (AMAP, 2011); and (7) changes to the distribution, abundance, and properties of Arctic lakes (Walsh et al., 2005; Smith et al., 2005; Prowse et al., 2006a).

1.1 Arctic Lakes

The low-lying landscapes of Arctic coastal and interior plains host various lentic freshwater ecosystems spanning a range of environmental settings (Vincent et al., 2012). Wetlands, ponds, and lakes differing in size, depth, morphology, geology, food web structure, energy, nutrient input, and abundance provide a variety of seasonal and ephemeral aquatic environments for a wide range of biological organisms (Flanagan et al., 2003; Prowse et al., 2006a; Prowse et al., 2006b). Furthermore, these aquatic ecosystems contain immense cultural, economic, and ecological significance (Vincent et al., 2012). The circumpolar Arctic has been described as “the world’s largest wetland”, with freshwater Arctic lakes and ponds comprising upwards of 90% of the total surface area in certain Arctic regions (Raatikainen and Kuusisto, 1990; Pienitz et al., 2008). The Yukon Delta has approximately 200,000 lakes situated within its 80,000 km^2 boundary (Maciolek, 1989) and the Mackenzie Delta encompasses roughly 45,000 lakes despite an area of 13,000 km^2 (Emmerton et al., 2007). These freshwater ecosystems facilitate the sustainment of a significant portion of Arctic biodiversity relative to the surrounding drier landscapes (MacDonald et al., 2009) and have, thus, been referred to as “tundra oases” (Rautio et al., 2011). More importantly, the sheer abundance of northern freshwater ecosystems enables them to affect global biogeochemical dynamics (Walter et al., 2006).

The individual physical and chemical characteristics of Arctic lakes are dependent on a multitude of factors: surface sediments, underlying bedrock geochemistry, and to a lesser extent, atmospheric mineral deposition determine the geochemical conditions of the lake (Hutchinson, 1957; Wetzel, 1983); physical aspects such as surface area, depth, distance from outflow, slope, and water retention factors influence the inflow and outflow of water (Hamilton et al., 2000). The primary inputs of water into Arctic lakes stem from local catchments (Hartman and Carlson, 1973; Woo et al., 1981; Woo and Xia, 1995). The contributing processes include snow and/or ice accumulation and melt, hillslope runoff (Woo et al., 1981), and lateral overflow from wetlands and streams (Marsh and Hey, 1989).

The onset and pace of the freshet depends on several climatic processes: the rate of temperature increase in late spring/early summer; wind; the inflow of basin meltwater; and terrestrial heat exchanges (Prowse et al., 2006b). The main mechanisms of water loss include evaporation and seepage (Kane and Slaughter, 1973; Woo, 2000). Arctic lakes experience significant annual fluctuations in sunlight hours, air temperature, and consequently, water temperatures (Rautio et al., 2003; Laurion et al., 2010). The patterns of biota, food web structure, and productivity of Arctic lakes exhibit considerable regional and local variability; the diversity and abundance of these parameters are affected by broad environmental conditions characterizing a specific region, as well as the local-scale physical characteristics unique to each lake (Prowse et al., 2006b).

Arctic lakes generally experience lower levels of primary productivity than more southerly, temperate lakes (e.g. Shortreed and Stockner, 1986; Flanagan et al., 2003; Prowse et al., 2006b). Several factors influence the composition, pigment structure, and biomass of primary production in Arctic lakes: (a) lake-ice and snow covering lakes for the majority of the year (eight months) (Hobbie et al., 1999a), (b) comparatively shorter growing seasons resulting in less time for biological activity to take place (Flanagan et al., 2003), (c) low nutrient availability, (d) cold temperatures, (e) freeze-up and desiccation-induced trauma during the extensive winter, (f) high photosynthetically active radiation (PAR) and ultraviolet radiation throughout the short summer, and (g) predation from grazing zooplankton (Rautio et al., 2011). The compounding effect of these interacting abiotic and biotic factors results in rates of biological activity well below those of subarctic and temperate lakes (Hobbie et al., 1999b).

There exists scientific uncertainty surrounding whether nitrogen, phosphorous, or combinations of both nutrients limit productivity in Arctic lakes (Gregory-Eaves et al., 2000; Levine and Whalen, 2001). While some studies conclude that phosphorous limits productivity in Arctic lakes (e.g., Schindler et al., 1974; Shortreed and Stockner, 1986) other studies suggest that nitrogen is limiting (e.g., Alexander et al., 1989**;** Lim et al., 2001). The inability of soil moisture and groundwater to permeate through ice-rich permafrost results in the presence or absence of this cryospheric component robustly influencing Arctic terrestrial hydrological processes (White et al., 2007). As a result of hydrological processes being confined to a thin, relatively impermeable, and nutrient-poor active layer (Kokelj et al., 2009a), the thickness of both the active layer and underlying permafrost have a strong influence on runoff, groundwater, and lake water characteristics (Kokelj et al., 2009a).

The pedology of the surrounding catchment is one of the most influential properties affecting dissolved organic carbon (DOC) in Arctic lakes (Rautio et al., 2011). Through the leaching of organic matter and aquatic plant material found within the catchment, both dissolved autochthonous and allochthonous carbon are introduced to the lake system. The nature of dissolved organic matter can be modified by ultraviolet radiation through processes such as photolysis (Lean, 1998). Through the conversion of high molecular-weight organic matter to lower-weight structures, organic carbon becomes more biologically available to lake organisms (Tranvick, 1998). Conversely, organic carbon can be lost through chemical reactions between coloured dissolved organic matter and cations and metals, resulting in the chemical products being absorbed by lake sediments (Thomas, 1997).

1.2 Relevance of Arctic Lakes to Climate Change Studies

Arctic lakes are sensitive to environmental changes, with research suggesting that climate warming has prompted distinct changes to the surface area, water levels, and ecological components of various northern lakes over the past few decades (e.g., Rühland and Smol, 2005; Smith et al., 2005; Smol et al., 2005; Riordan et al., 2006; Smol and Douglas, 2007; Plug et al., 2008; Labrecque et al., 2009; Vincent, 2009; Williamson et al., 2009). Similarly, paelolimnological investigations have revealed that Arctic lakes have responded robustly to changes in climate during the past several hundred years

(Douglas and Smol, 1999; Rühland et al., 2003; Rühland and Smol, 2005; Smol et al., 2005).

Given that polar regions will continue to experience the most significant warming (Kattsov et al., 2005; Prowse et al., 2006a), Arctic lakes, in particular, afford a unique ability to reveal climate-induced changes, as well as illuminate how other aquatic ecosystems, both Arctic and temperate, are likely to respond in the future (Lim and Douglas, 2003). Lakes and ponds comprise a vast and well-distributed linkage of ecosystems that reflect the on-going responses of terrestrial aquatic ecosystems to climate change (Williamson et al., 2009). As a result, it is evident that Arctic lakes represent crucial reference ecosystems to which past, recent, and future global environmental change can be compared (Lim and Douglas, 2003).

Expected consequences of climate change in lake ecosystems have been discussed in numerous publications dating back several decades (e.g., Rouse et al., 1997; Schindler and Smol, 2006; ACIA, 2005; Prowse et al., 2006a; White et al., 2007; Williamson et al., 2009; Vincent, 2009; Vincent et al., 2012), and three broad classes of change, as well as their expected effect on primary production, have been highlighted below: (1) biogeochemical (e.g., nutrient cycling, dissolved organic matter, and oxygen dynamics); (2) physical (e.g., water transparency, water temperature, water level, duration and thickness of ice cover, and thermal stratification); and (3) biological (e.g., productivity, species invasions/interactions) (Williamson et al., 2009).

1.2.1 Biogeochemical Impacts of Climate Change on Arctic Freshwater Ecosystems

As air, water, and subsurface temperatures increase across the circumpolar Arctic, the biogeochemistry of lake ecosystems can be affected directly by increased chemical and biochemical reaction rates, as well as indirectly through a multitude of hydrological and landscape processes (Vincent et al., 2012). Biogeochemical characteristics of northern lake ecosystems are intimately linked with the dynamic conditions of the underlying permafrost (White et al., 2007). Permafrost thaw and degradation is expected to affect Arctic lake productivity primarily through processes associated with the deepening of the active later. As the active layer of the area surrounding a lake deepens, previously non-existent infiltration pathways develop. As a result, accelerated geochemical weathering of subsurface materials that were formerly isolated from infiltrating water is likely to result in increased inputs of nutrients (Hobbie et al., 1999a; Breton, et al., 2009) and organic carbon into Arctic lakes, thereby increasing microbial produced carbon dioxide and methane (Walter et al., 2006; Mazéas et al., 2009; Laurion et al., 2010). Rates of geochemical weathering may be intensified further by projected increases in precipitation expected to affect the terrestrial regions of North America, Europe, and Asia (Kattsov et al., 2005). An increase in the supply of nitrogen, phosphorus or carbon to freshwaters can result in their eutrophication, increasing biological productivity and potentially affecting the entire aquatic food web.

Similar to nutrient levels, inputs of DOC are likely to increase with deepening active layers and increased runoff (Wrona et al., 2006b). Expected changes in tundra vegetation are also likely to cause changes to DOC dynamics in Arctic catchments (White et al., 2007). DOC affects lentic productivity through a number of direct and indirect processes related to the penetration of radiation, turbidity, and carbon processing (Wrona et al., 2006b). The relationship between DOC and productivity is, therefore, complicated by the multitude of factors as well as species-specific responses that climate-change-induced alterations to DOC regimes can impose on Arctic aquatic organisms (Wrona et al., 2006b). While reductions in light penetration and availability can inhibit biological productivity in lakes, the contrasting effect of reductions in the levels of harmful UV-B radiation can also offset the arresting effects (Vincent and Hobbie, 2000).

1.2.2 Physical Impacts of Climate Change on Arctic Freshwater Ecosystems

Increased evapotranspirative loses, facilitated by warmer summer air temperatures and longer open-water seasons, may impact local water balances of lakes across the circumpolar Arctic with important implications for the overall abundances of lakes in northern latitudes (Schindler and Smol, 2006; Smol and Douglas, 2007). Climatic changes have been associated with concomitant changes in lake surface area (Hinzman et al., 2001); however, responses in lake water balances are complicated by a multitude of confounding impacts relating to annual and decadal variations in weather and permafrost degradation (White et al., 2007). Additionally, whether lake-formation or -drying occurs is largely dependent on local physiographic characteristics, the most important of which, is the presence of continuous versus discontinuous permafrost. Lakes in areas of continuous permafrost are expected to grow in size and abundance, while lakes in areas of discontinuous or degraded permafrost may shrink and disappear entirely (White et al., 2007).

Freshwater ice that covers Arctic lakes for upwards of 8 months of the year exerts one of the strongest influences on phytoplankton growth and lake productivity (AMAP, 2011). As a result, changes to the timing, duration, extent, and overall phenology of lake ice will have prominent impacts on the limnology of northern lake ecosystems (Vincent et al., 2008; Mueller et al., 2009; Prowse et al., 2011). Reductions in ice-cover duration and extent are likely to affect productivity in Arctic lakes through several mechanisms: (a) warmer and longer ice-free seasons will extend the period in which biological activity can take place, facilitating a likely increase in productivity (Douglas and Smol, 1995); (b) decreased ice cover thickness is likely to augment under-ice oxygen and algal production by increasing the amount of solar radiation penetrating through the ice (Prowse and Stephenson; 1986); (c) longer ice-free seasons will lengthen the stratified season of certain lakes contributing to increased mixing depths; (d) warmer water temperatures will increase reaction rates and the metabolic activity of organisms (Wrona et al., 2006b); and (e) changes to lake ice albedo due to increases in wetted snow transforming into white ice could alter under-ice productivity (Yao et al., 2014). It is important to note, however, that the extent and duration of these changes will vary locally with differences in physical characteristics and local climate (Rouse et al., 1997).

While prolonged intervals of the open-water season may reduce the constraining influence of light limitation, any increases in the annual rate of primary production in Arctic lakes may be offset by the detrimental effects of intensified ultraviolet radiation on phytoplankton communities (Gareis et al., 2010). Modelling undertaken by Vincent et al. (2007) indicates that the loss of protective lake-ice and snow coverage can result in greater increases in UV-exposure than even the depletion of ozone in the stratosphere. Although both phytoplankton cells and communities can utilize a range of photoprotection and repair strategies, changes to Arctic lake algal community structure are likely (Smol, 1988; Vincent et al., 2012).

Warming of overall water column temperatures is likely in many northern lakes as both increased incident sunlight and shorter under-ice seasons contribute to an overall radiative heating of high latitude lakes (Vincent et al., 2012). As a result of Arctic lakes being comprised of water often at 3.98 ^{o}C or less, even minute changes to lake thermal structure can have profound effects on the mixing and stratification regimes of these limnological systems (Vincent et al., 2012). For example, lakes that are cold monomictic (winter stratification but not summer) may shift to dimictic (mixing and stratification during both seasons), and thermokarst ponds may endure shorter spring- and fall-mixing periods (Laurion et al., 2010). Potential changes to the mixing and stratification regimes of lakes can have important implications for the thermodynamics of the surface layer, phytoplankton and zooplankton growth rates, and the exhaustion of oxygen in deep water (Sorvari et al., 2002; Vincent et al., 2012). Consequently, impacts to higher trophic organisms are probable due to increased preservation of contaminants within Arctic ecosystems (Chételat and Amyot, 2009).

1.2.3 Biological Impacts of Climate Change to Arctic Freshwater Ecosystems

Broadly speaking, the compounding effects of reductions in ice cover extent and duration, warmer ambient air and water temperatures, and enhanced nutrient and DOC supplies derived from more biogeochemically active catchments have the potential to enhance lentic productivity in Arctic lakes (Bonilla et al., 2005; Prowse et al., 2006a; Bonilla et al., 2009; Antoniades et al., 2011). Analyses of the diatom communities found in lake sediment cores from the last century indicate significant shifts in diatom community assemblages, likely in response to changes in climate (Smol et al., 2005; Rühland et al., 2005). Furthermore, these paleolimnoligcal investigations indicate that any contemporary changes in diatom community structure are likely to manifest themselves differently across and even within disparate regions of the circumpolar Arctic (Vincent et al., 2012).

The biota, structure, function, and overall diversity of Arctic lake ecosystems are likely to be affected by climate change (e.g., Wrona et al. 2006). However, projecting how organisms living in Arctic lake systems will respond to climate change is complicated by the multitude of anticipated species- and system-specific responses limiting the certainty with which broad generalizations can be made. Physiological responses to other environmental stressors (i.e., mineral and gas exploration) simultaneously affecting Arctic aquatic ecosystems may interact with each other, and be difficult to distinguish from climate-related stressors (Williamson et al., 2009).

1.3 Knowledge Gaps

Despite recent increases in the number of limnological investigations undertaken across the circumpolar Arctic, including - Alaska (e.g., LaPerriere et al., 2003); the western Canadian Arctic (e.g., Ogbebo et al., 2009); the eastern Canadian Arctic (e.g., Westover et al., 2009); Finland (e.g., Luoto, 2009); Greenland (e.g., Cremer et al., 2005); Norway (e.g., Løvik and Kjellberg, 2003); Russia (e.g., Moiseenko et al., 2009); and Sweden (e.g., Brunberg et al., 2002) – scientific, ecological investigations of temperate lakes overwhelmingly outnumber those of Arctic lakes (MacDonald et al., 2012). Furthermore, there have historically been few long-term biological monitoring programs situated in northern latitudes (e.g., Douglas and Smol, 1993, 1994, 1995; Michelutti et al., 2003; ACIA, 2004). As a result, there still an urgent need for improved spatial and, more importantly, temporal coverage of Arctic lake ecosystems.

Groups and organizations responsible for the management and stewardship of these areas typically do not have the financial resources available to undertake the uninterrupted, recurrent sampling regimes necessary to produce meaningful and instructive long-term datasets (MacDonald et al., 2012). Research attempting to investigate climate-driven changes in northern lake ecosystems is complicated by a legacy of relatively instantaneous research initiatives predicated on defining the state of a lake ecosystem given principally transitory measurements (Bailey et al., 2004). Furthermore, these momentary observations have traditionally been summer lake-water samples analyzed to reveal conventional physical and chemical parameters, indicative of only the days or hours immediately before sampling (Bailey et al., 2004). Not only are these “snapshot” samples limited in their temporal scope, but they also fail to reflect and integrate other relevant biological, physical, and chemical conditions of the lake crucial to defining its state.

Uninterrupted and long-term monitoring programs of Arctic freshwater ecosystems have not been undertaken due to the financially and logistically prohibitive nature of northern research. Continuous, high-frequency sampling is challenging to undertake and sustain in Arctic lakes, particularly in those under government regulation, since they are generally far away from towns or established centres of human activity (MacDonald et al., 2012). Moreover, northern research is technically challenging, subjected to extreme weather conditions, and study lakes are commonly only accessible by snowmobile and/or helicopter, imposing additional financial constraints on any research initiative (Hille, 2010).

Although there has been an increasing effort to document the physical, biological, and chemical limnological characteristics of water bodies across the Canadian Arctic (i.e. Kokelj et al., 2005; Mesquita, 2008; Thompson, 2009; Kokelj et al., 2009a,b; Hille, 2010), the aforementioned financial and logistical difficulties associated with undertaking field work in the North have limited these limnological investigations to sampling regimes unable to integrate the necessary temporal, chemical, physical, and biological parameters to accurately define the current state of these systems. Identifying and forecasting the effects of climate change in northern lakes, especially in light of additional anthropogenic and environmental stressors acting concomitantly on these ecosystems, are crucial to the stewardship and management of these aquatic resources.

1.4 Purpose of Study

To date, attempts to integrate and characterize the data generated by decades of Arctic limnological investigations have been limited in both temporal and spatial coverage. Additionally, despite the multitude of publications affirming that Arctic lake ecosystems are important indicators of a changing global climate, the cumulative responses of Arctic limnology remain relatively unclear. As a result, the main objectives of this thesis are:

1. *Conduct a comprehensive limnological overview of Noell Lake, NWT by compiling available data from a multitude of limnological, hydrological,*

environmental, geographical, and ecological research and monitoring initiatives.

(Chapter 2)

2. *Assess the reliability and data validity (quality assurance and quality control) of a newly developed sensor-based, Arctic Lake Monitoring System (ALMS) and complementary non-automated buoy and subsurface mooring systems in recording and relaying continuous limnological measurements during open-water and under-ice seasons.*

   (Chapter 3)

3. *Develop an updated and improved description of the general limnology and patterns of spatial and temporal variability of water quality parameters of Noell Lake. An array of different sensors, data-sources, and experiments were utilized to develop a comprehensive understanding into the annual limnological cycle of Noell Lake.*

   (Chapter 4)

4. *Using data from existing literature, investigate whether observed patterns in temporal and latitudinal changes in Arctic lake primary productivity and geochemistry and be related to concomitant changes in Arctic climate (i.e., temperature-, precipitation-regimes) and cryospheric conditions (i.e., ice-cover duration). The key hypothesis being tested is whether Arctic lakes are showing increased primary productivity "greening", through time and by latitude, similar to that documented for Arctic terrestrial systems.*

   (Chapter 5)

5. *Expand on the original northern-lake productivity models developed by Flanagan et al. (2003) and determine whether the addition of other limnological variables (dissolved organic carbon, dissolved inorganic carbon, lake depth, conductivity, and ice-cover) improve the prediction of*

autotrophic epilimnion productivity in high latitude lakes of the northern hemisphere.
(Chapter 6)

Chapter 7 contains general conclusions and recommendations on future related research required to address identified knowledge gaps.

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CHAPTER 2: THE CLIMATOLOGY, HYDROLOGY, LIMNOLOGY, AND ECOLOGY OF NOELL LAKE, NORTHWEST TERRITORIES, CANADA

Abstract

All relevant historical climatic, hydrological, limnological, and ecological information pertaining to Noell Lake was compiled and synthesized. Relevant material, comprising both published and unpublished scientific sources, a multitude of significant geomorphological, geological, ecological, geochemical, and environmental information pertaining to Noell Lake was assembled. More specifically, information pertaining to phytoplankton, macrophyte, epipphyton, epipelon and fish communities were included, as well as results from historical water quality sampling, analyses of water column organic carbon and chlorophyll-*a*, open-water depth profiles of dissolved oxygen and chlorophyll-*a*, drainage basin characteristics, recent and historical forest fire histories, lake watershed characteristics, and information on wildlife. The establishment of reliable baseline conditions for the current ecology and limnology of Noell Lake is crucial to rigorous environmental assessments, comprehensive monitoring initiatives, and well-informed stewardship of the region in the future.

Keywords: Noell Lake, NWT; Historical limnological compilation; Baseline conditions

2.1 Introduction

Noell Lake (68^{o}31’37’’ N, 133^{o}30’48’’ W) is an Arctic lake, located in the uplands region, east of the Mackenzie River Delta in the western Canadian Arctic and approximately 15 km northeast of the town of Inuvik, NWT (de Rham & Carter, 2009) (Figure 2-1). The waterbody is situated in a lake-rich area and underlain primarily by expanses of continuous permafrost (Pienitz et al., 1997a). Thaw lake-formation cycles commencing approximately 11.6 to 8.5 ka BP likely caused the formation of thousands of lakes in the region (Dallimore et al., 2000). Despite its close proximity to the Mackenzie River, Noell Lake is geomorphically and hydrologically separated from this large river system (Ramlal et al., 1991). Regionally, Noell Lake is an important freshwater lake supporting established populations of Arctic grayling, lake trout, as well as several other fish species (Rescan Environmental Services Ltd., 1998). Given the proximity of Noell Lake to the town of Inuvik, it is a popular and important fishing location and has been identified as having particular significance to the broader community of Inuvik, due largely to these aquatic resources (Rescan Environmental Services Ltd., 1998).

Over the past 35 years, a limited number of limnological and ecological investigations have been conducted on Noell Lake. In 1982 and 1983, the Government of Canada Department of Fisheries and Oceans (DFO) undertook the first investigations of the general limnology and ecology of the lake; measurements included fish creel censuses, experimental gillnetting, tagging, biological sampling, water quality analyses, as well as depth profiles of dissolved oxygen and chlorophyll-*a* during the open-water season (Read and Roberge, 1986). Subsequently between 1985 and 1987, DFO undertook additional investigations on Noell Lake, as well as other lakes and channels in the region. Biweekly sampling of water chemistry, organic carbon, and chlorophyll-*a* was undertaken during the open-water season of 1985 (Anema et al., 1990a,b) and as part of the Northern Oil and Gas Action Program (NOGAP), DFO collected further information on macrophytes, phytoplankton, epiphyton, and epipelon during the open-water seasons of 1985 and 1986, (Ramlal et al., 1991). Noell Lake was also a part of northern lake-transect investigations by Pienitz et al. (1997a,b), who gathered water chemistry and other associated limnological data for 59 lakes along a south-north transect in the Yukon and Northwest Territories. Ogbebo et al. (2009a,b) gathered information on limnological variables and nutrient-limitation for 30 lakes (including Noell Lake) situated along the proposed Mackenzie Gas Project pipeline route in the Northwest Territories. Several recent northern limnological, ecological, and hydrological studies have used Noell Lake as a study site: Lantz and Kokelj (2008), Kokelj et al. (2009), and Thompson (2009) investigated ecological and geochemical effects of permafrost-thaw and retrogressive slump activity in lakes of the Mackenzie Delta region.

Although generating hydro-ecological baseline information, a major limitation of these previous studies is that measurements were often singular and sampled exclusively in the open-water season. To date, no studies have been conducted on any lake in the western Canadian Arctic that provide a continuous time series of measurements, report seasonal and inter-annual variations in water quality and related limnological parameters, and/or include measurements recorded during under-ice conditions. The objective of this chapter was to provide a synthesis of all existing geographical, limnological, water quality, and ecological on Noell Lake and its surrounding catchment. Collectively, this information serves as an important baseline of environmental conditions on Noell Lake for use in future environmental impact assessments and studies investigating the effects of regional climate variability and change as well as anthropogenic development on Noell Lake and the surrounding catchment region.

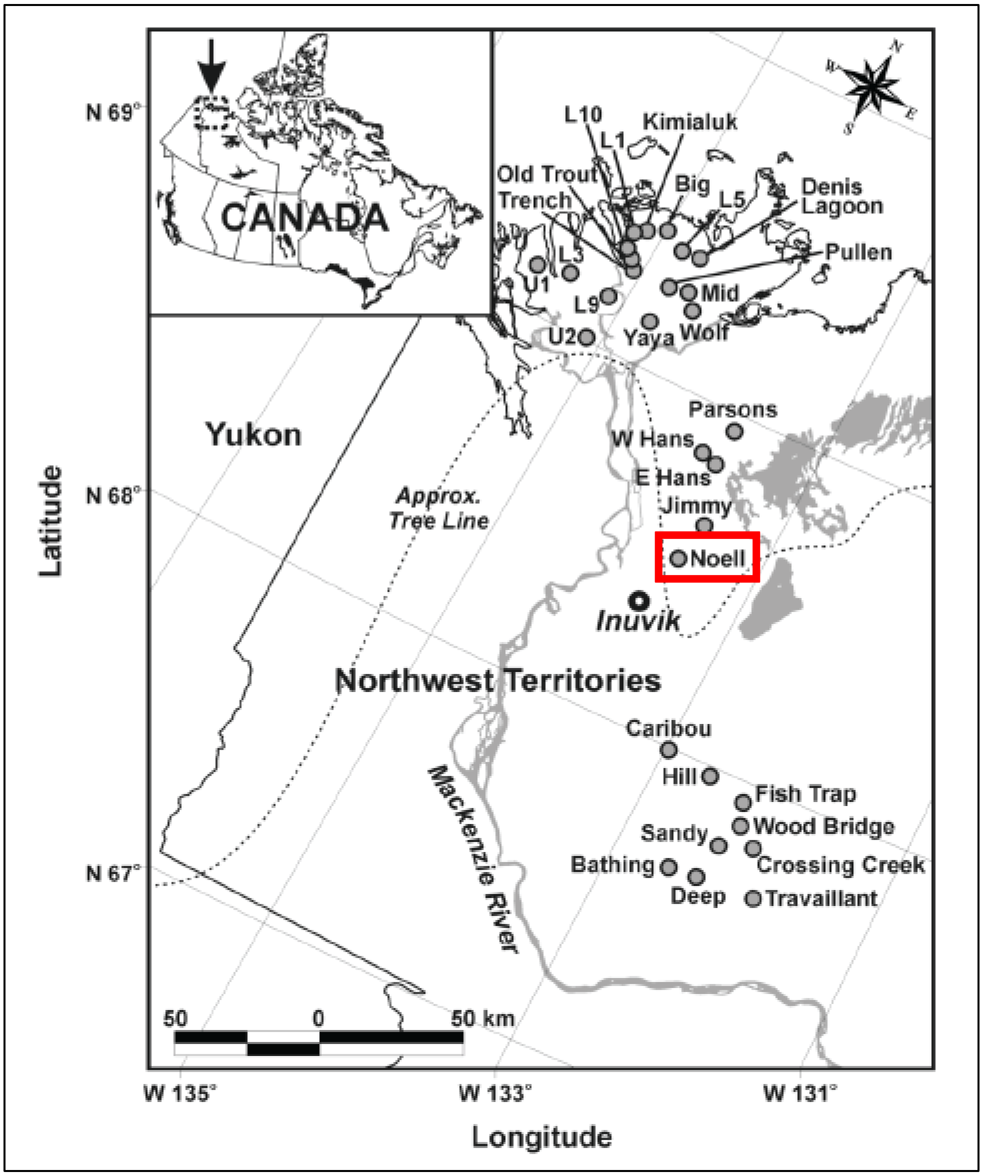


Figure 2-1: Location of Noell Lake, as well as various other lakes (grey dots), in the lower Mackenzie River basin, NWT.
SOURCE: Ogbebo et al. 2009a

2.2 General Geographical Features

Noell Lake is a medium-sized, oval-shaped, upland tundra lake. The south-, east-, and west-facing shorelines are gently curved while the northern shoreline is angular, funneling into Noell Lake’s singular outflow: Jimmy Creek. Jimmy Creek flows northward from Noell Lake into Jimmy Lake and ultimately empties into the Husky Lakes Basin (Read and Roberge, 1986). Noell Lake is surrounded by elevated hills on all sides, forming a ringed-dike around the circumference of the lake. Various rills, streams, and gullies flow down from the top of the ringed-dike into the lake below. Lake surface elevation is 89.93 m ASL (de Rham and Carter, 2009; Ogbebo et al., 2009a) and Noell Lake does not undergo the regular flooding that Ogbebo et al. (2009a) suggest adjacent lakes below 5 m ASL experience. There are several cabins constructed along the shoreline of Noell Lake. There are no islands in the lake and there had been minimal industrial development in the catchment area until the construction of the aforementioned all-season road began in the spring of 2013. Important biogeochemical, hydrogeological, and ecological changes to the limnological and ground water components of Noell Lake are likely to accompany the construction and use of the all-season road (Meriano and et al., 2009).

Table 2-1: Physical characteristics of Noell Lake. Climate data is from the Inuvik Weather Station (1971 – 2000 Climate Normals), acquired from Environment Canada's *Weather Office* website.

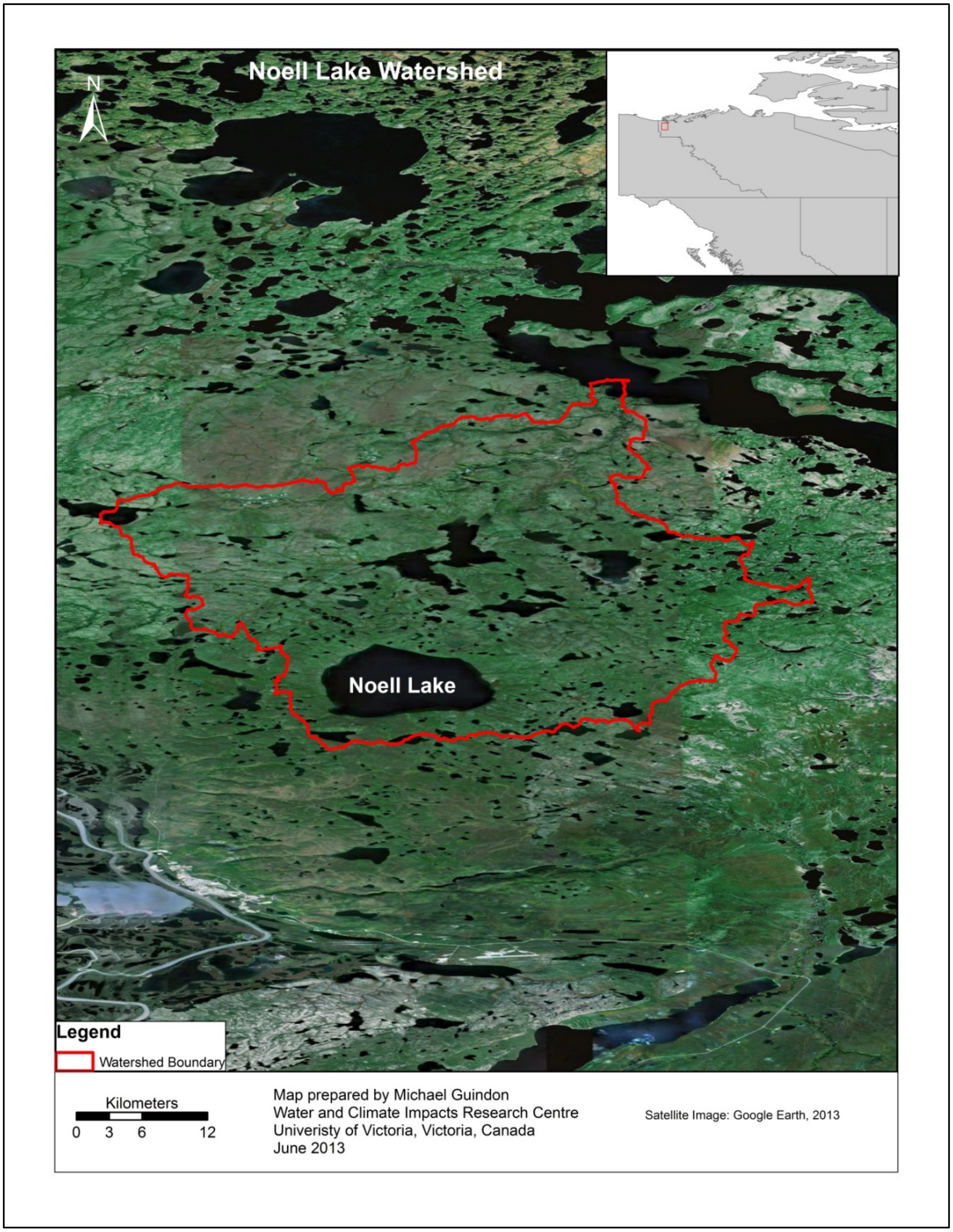
Latitude^{a}	68^{o}31' N
Longitude^{a}	133^{o}30' W
Elevation (m)^{a}	89.93
Surface Area (km^{2})^{a,b,c}	30.00
Maximum Depth (m)^{a}	18.30
Mean Depth (m)^{c}	2.80
Mean July Air Temperature (^{o}C)^{d}	14.00
Mean January Air Temperature (^{o}C)^{d}	-27.00
Mean Annual Precipitation (mm)^{d}	250.00
Outflows^{b}	Jimmy Creek
Open Water Period	Early June – Early November

SOURCES: [a]de Rham and Carter 2009a; [b]Read and Roberge 1986; [c]Ogbebo et al. 2009a; [d]Weather Office

2.2.1 Watershed Characteristics

In June 2013, four previously non-existent maps pertaining to various watershed characteristics of Noell Lake were completed. The delineation of Noell Lake's watershed boundaries was mapped using both projection (Figure 2-2) and satellite techniques (Figure 2-3); additionally, projections of the bedrock geology underlying the Noell Lake watershed (Figure 2-4) as well as vegetation characterizing the region (Figure 2-5) were produced.

Figure 2-2: Watershed boundary of Noell Lake, NWT.



Map prepared by Michael Guindon
Water and Climate Impacts Research Centre
Univeristy of Victoria, Victoria, Canada
June 2013

Satellite Image: Google Earth, 2013

Figure 2-3: Satellite image of the watershed boundary surrounding Noell Lake, NWT.

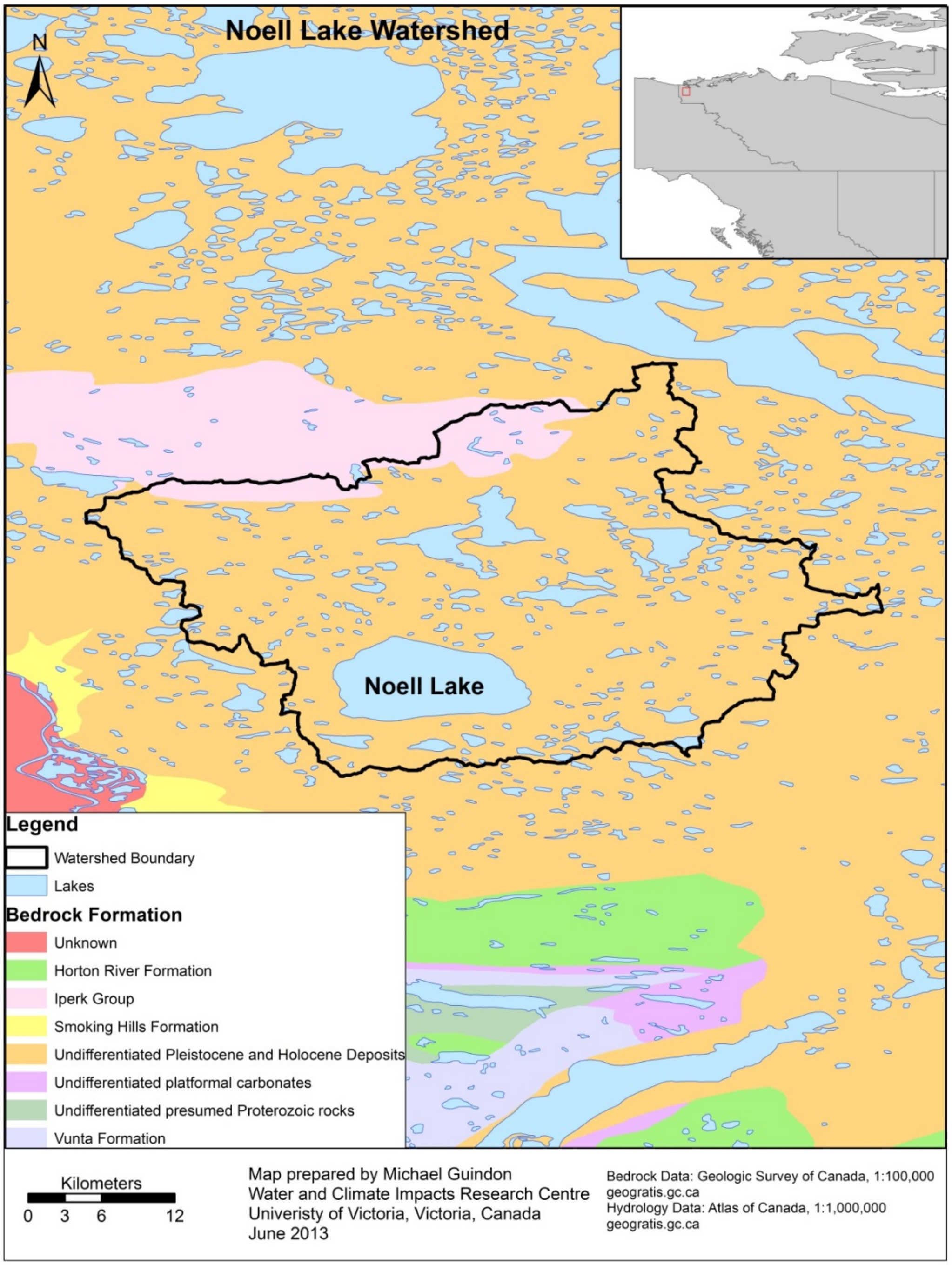


Figure 2-4: Bedrock geology of the Noell Lake, NWT watershed.

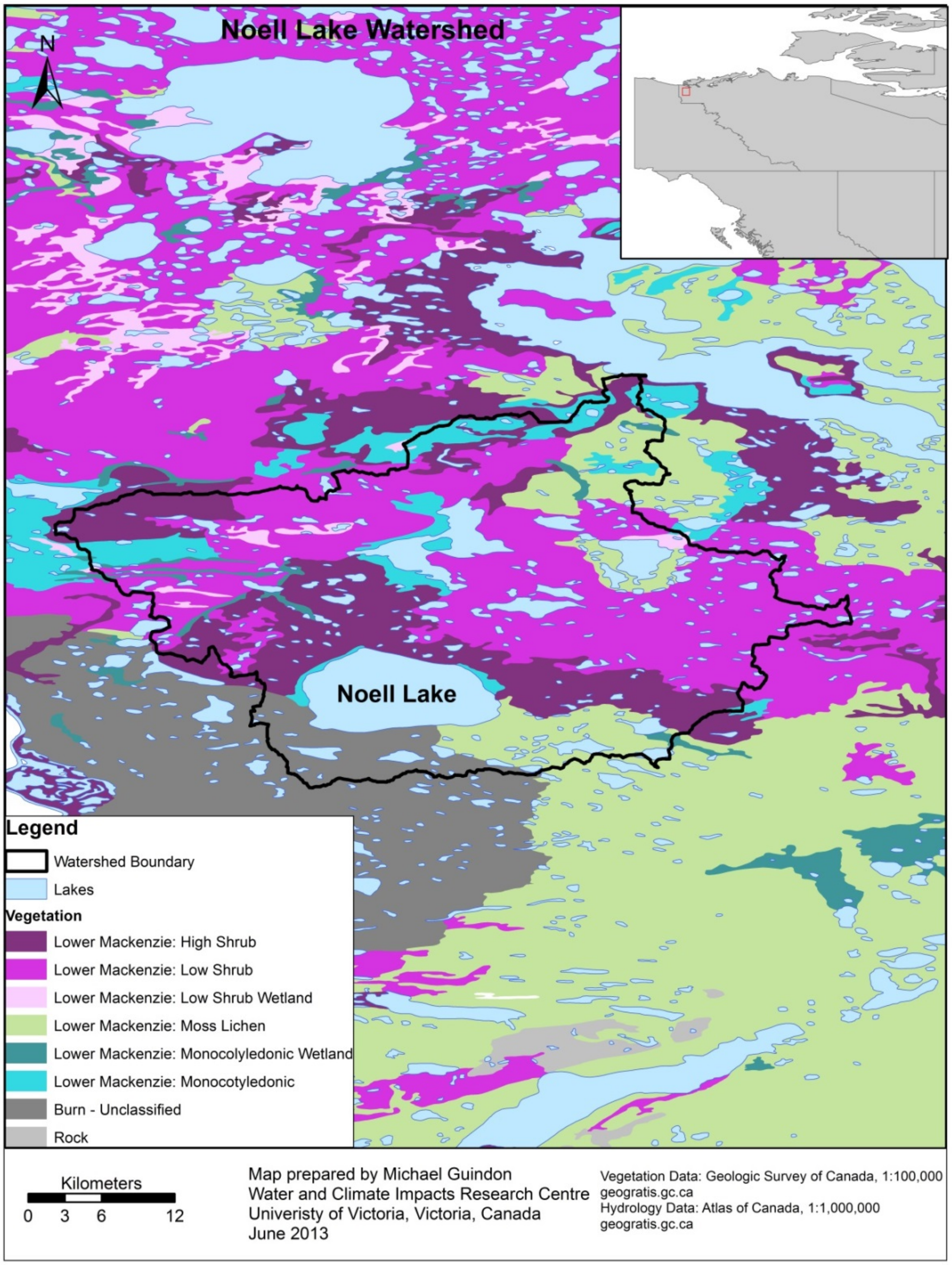


Figure 2-5: Vegetation of the Noell Lake, NWT watershed.

2.2.2 Bathymetry and Shoreline Features

Several operations related to the bathymetry profiling of Noell Lake were undertaken during 2008 and 2009 and are summarised by de Rham and Carter (2009). A multitude of methodologies and data sources were used to produce novel bathymetric maps of Noell Lake including helicopter-based bathymetry surveys in September of 2008, shoreline and landscape mapping, lake surface areas calculation, additional mapping of lake bathymetry using Surfer software, and the development of east-west and north-south transects of Noell Lake’s bathymetry.

The lake has a maximum length of 6.8 km, a maximum width of 5.5 km (Read and Roberge, 1986), and a total lake area of Noell Lake was determined to be 30 km^2 (de Rham and Carter, 2009). The maximum lake depth has been determined to be 18.3 m (de Rham and Carter, 2009). Noell Lake can be dissected into 4 main quadrants: (1) northeast, (2) northwest, (3) southeast, and (4) southwest. Prominent lake features include 3 deep holes located in the northeast and southwest quadrants of the lake (Figure 2-6). The northeast quadrant contains 2 of the 3 holes; the more westerly hole is approximately 15 m deep and the hole to the east of the quadrant is approximately 18 m deep. The hole in the southwest quadrant is the deepest part of Noell Lake, extending to depths of 16 m. The westerly hole in southwest quadrant is partially visible in the North-South bathymetric profile of Noell Lake (Figure 2-7).

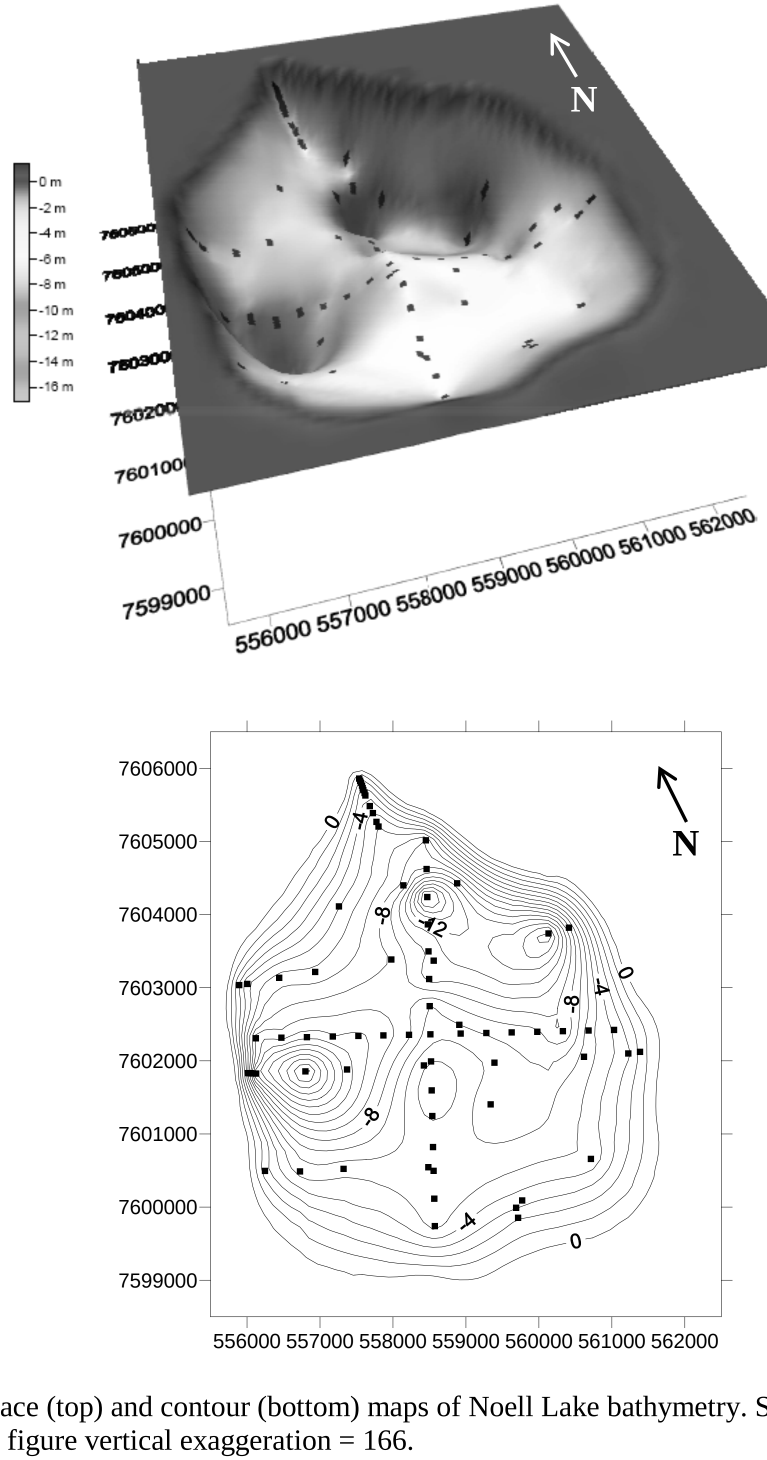


Figure 2-6: Surface (top) and contour (bottom) maps of Noell Lake bathymetry. Survey points are included; top figure vertical exaggeration = 166.
SOURCE: de Rham and Carter 2009

The northwest and southeast quadrants are comparatively shallower than the northeast and southwest quadrants; the lake bottom slopes gently from the shoreline towards the centre of the lake. Maximum depth in the northwest quadrant is approximately 8 m, and only 6 m in the southeast quadrant. In general, the lake bottom of Noell Lake slopes gradually from the shoreline inwards with the exception of western and northeastern shores, where considerably steeper slopes exist.

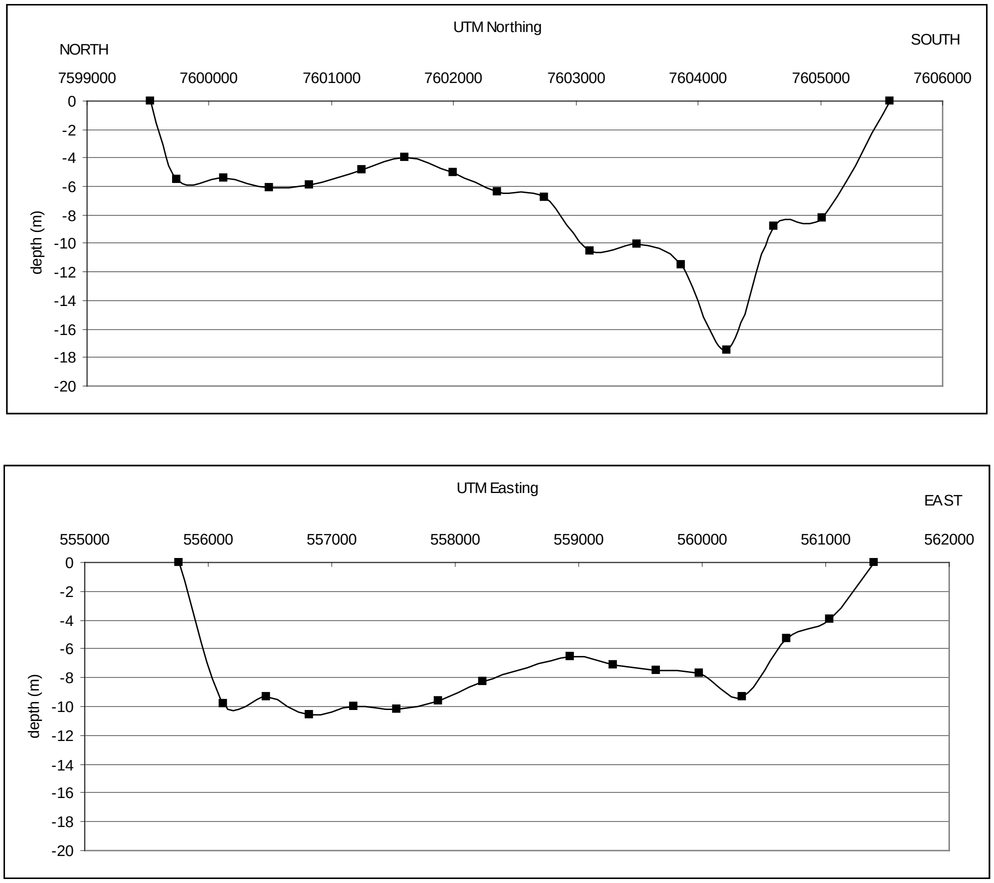


Figure 2-7: North-South (top) and East-West (bottom) bathymetric profiles of Noell Lake, NWT; vertical exaggeration = 199 times.
SOURCE: de Rham & Carter 2009

2.3 Drainage Basin Characteristics

Noell Lake is situated within the southern end of the Husky Lakes Drainage Basin, which is located to the east of the Mackenzie Delta and is bordered by the Mackenzie Delta Drainage Basin to the south, and the Kugmallit Bay Drainage Basin to the north (Figure 2-8). The total land area draining into Husky Lakes, excluding the surface area of lakes, is 7462 km^2 (Gushue et al., 1996). Discharge within the basin is highly seasonal and the Husky Lakes Basin eventually discharges directly into the Beaufort Sea (Gushue et al., 1996).

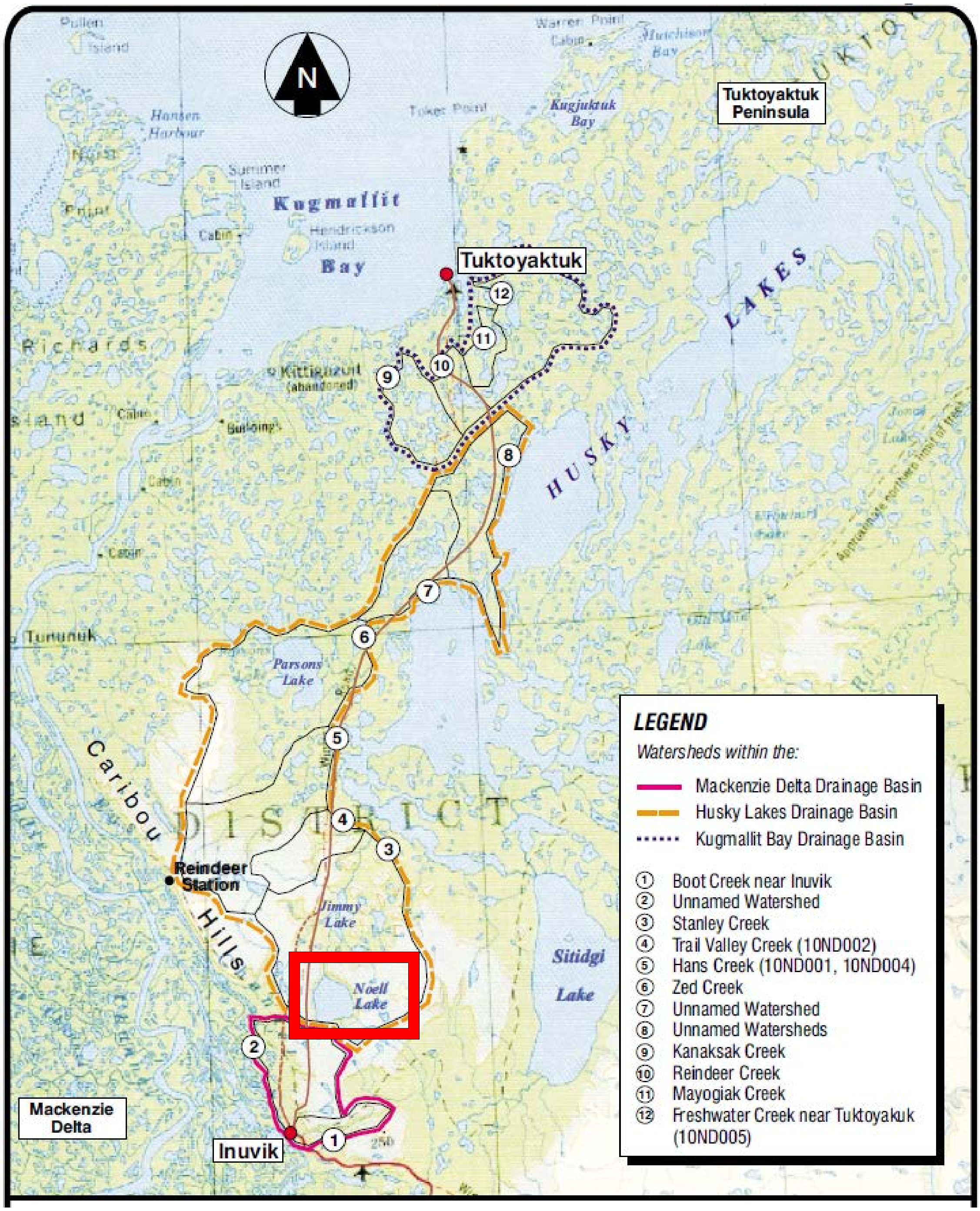


Figure 2-8: Drainage basins and watersheds within the Inuvik-Tuktoyaktuk, NWT region.
SOURCE: Rescan Environmental Services Ltd. (1998)

Noell Lake is located just north of the treeline, within the relatively barren landscape of the Arctic-tundra ecological zone (Ogbebo et al., 2009a). Ogbebo et al. (2009b) describe the area surrounding Noell Lake as a “dry, cold, treeless desert”. The various boreal and tundra flora, grasses, dwarf shrubs, herbs, lichens, mosses, and sedges that characterize the vegetation of the region are well-adapted to the short growing seasons that characterize the area (Ogbebo et al., 2009a,b). Although comparatively less abundant, Labrador Tea and assorted berries also comprise the vegetative community of the Noell Lake region (Quinton and Marsh, 1999).

Table 2-2: Characteristics of Husky Lakes Drainage Basin.

Area - excluding lakes (km^2)[i]	7, 462
Soil[a]	Organic; mineral
Bedrock Geology[b]	Carbonate and shale bedrock;
Surficial Materials[b]	Glacial till; ice-contact deposits; glacial outwash
Terrain[c,d]	Rolling tundra; continuous permafrost; localized taliks; Noell Lake is surrounded on all sides by hills, forming a ringed-dyke around the lake’s circumference
Topographical Features[e,f]	Ice-wedge polygons; pingos; retrogressive thaw slumps
Ecoregion[h]	Arctic-tundra ecological zone
Dominant Vegetation[g]	Variety of boreal and tundra vegetation; dwarf shrubs; mosses, lichens; Labrador Tea; berries
Mean Annual Discharge[i, j] (m^3/s)	0.22

NOTES: [j]Station Name = Trail Valley Creek Near Inuvik (10ND002) 1980-1990
SOURCES: [a]Hille 2010; [b]Kokelj et al. 2005; [c]Read and Roberge 1986; [d]Burn 2002; [e]Mackay 1963; [f]Rampton 1988; [g]Quinton and Marsh 1999; [h]Ogbebo et al. 2009; [i]Gushue et al. 1996

The edges of each ecological stage are heavily influenced by historical fire events (Ritchie, 1984; Timoney et al., 1992; Landhausser and Wein, 1993; Payette et al., 2001; Lantz et al., 2010). In 1968, a wildfire burned from the area around Inuvik, NWT north toward Noell Lake (Landhausser and Wein, 1993; Mackay, 1995). The intensity of the fire incinerated the existing vegetation and organic matter of the area facilitating a deepening of the active later and surface permafrost thaw (Mackay, 1995). Burned areas are characterized by dense alder and willow growth and are distinguishable from the unburned tundra (Landhausser and Wein, 1993).

In the summer of 2012, lightning is believed to be the cause of another wildfire that burned approximately 4,000 ha of forest and vegetation around the Noell Lake area

(Busch, 2012). The blaze left a mosaic of burned and unburned areas revealing a wide spectrum of burn intensities around the Noell Lake region (Figure 2-9). Increased fluxes of dissolved organic carbon, dissolved nutrients, and ionic constituents were likely consequences of the wildfire (Carigan et al., 2011).

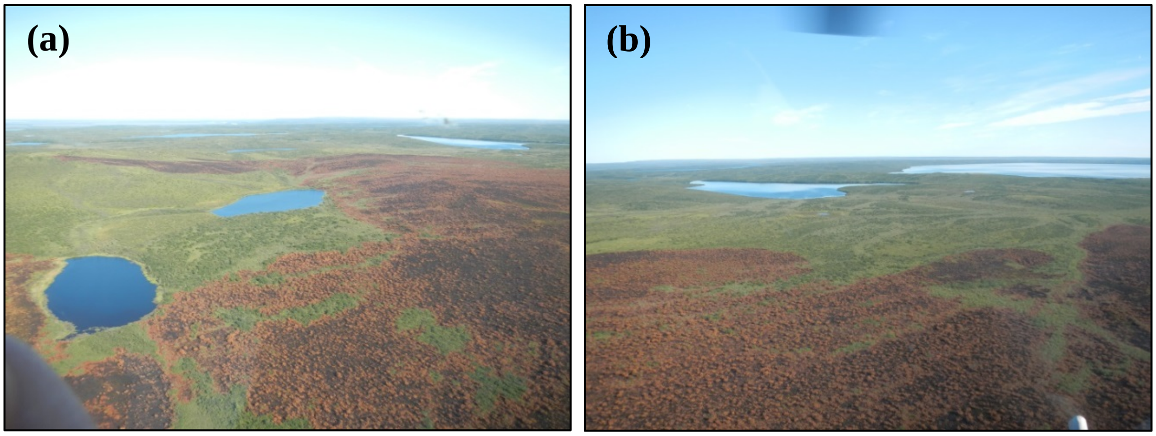


Figure 2-9: (a) Evidence of wildfire burns in the Noell Lake region, and (b) Noell Lake itself (Noell Lake is visible on the top-right of the figure). Pictures were taken on July 11th, 2012.

Surficial materials of the region are comprised of glacial till, ice-contact deposits, and glacial outwash eroded from the carbonate and shale bedrock found within the Mackenzie Basin (Kokelj et al., 2005). Continuous permafrost lies beneath the expansive tundra landscape with localized taliks forming beneath lakes deep enough to resist complete freezing during winter (Burn, 2002). Frequent topographical features of the landscape include ice-wedge polygons, pingos, and retrogressive thaw slumps (Mackay, 1963; Rampton, 1988). The ground surface is composed of primarily mineral earth hummocks ranging in both height and diameter from 0.4 to 1.0 m (Quinton and Marsh, 1998). Upper and lower peat layers composed of organic and mineral soils freeze during the 8 month winter period (Hille, 2010). Although vertical infiltration is limited while the soil profile is frozen, with the return of above 0 ^{o}C temperatures in May, meltwater is able to infiltrate into the newly thawed active layer (Hille, 2010).

2.4 General Climatology

Characterized by relatively brief summers offsetting long winters, Noell Lake exhibits extreme variations in daylight hours over the course of the year. While the winters are cold, dark, and lengthy, the summers offer continuous sunlight and significantly warmer temperatures (Ogbebo et al., 2009a). The closest Environment Canada weather station to Noell Lake is located at the Inuvik, NWT airport, approximately 15 km away. Polar night is a period of darkness lasting more than 24 hours. The first day Inuvik does not experience a sunrise typically occurs on December 7th and persists until January 6th. This span of 31 days without incident sunlight has important implications and constraining influences on primary productivity during the winter period of Noell Lake.

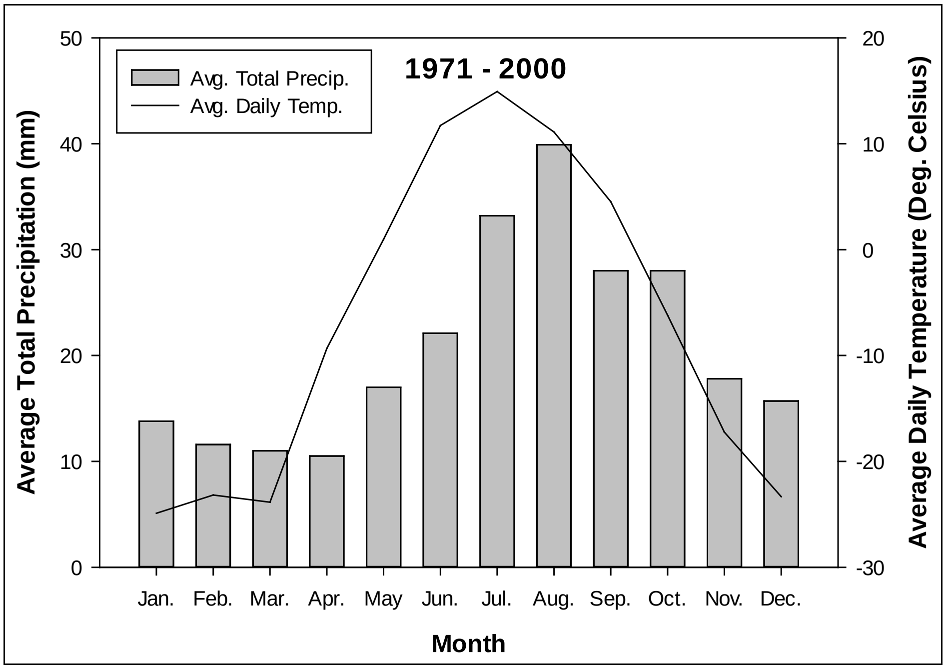


Figure 2-10: Total precipitation (mm) and average daily temperature (°C) from 1971-2000 at the Inuvik Airport.

For the most recent 5 year period (2007–2012), annual daily average air temperature was -7.1°C, the annual daily average maximum air temperature was -3.4°C, and the annual daily average minimum air temperature was -10.8°C. As a result of the significant changes in daylight hours over the course of the year, Inuvik experiences a wide spectrum of temperatures: average January air temperature was -24.9°C while average July air temperature was 14.9°C; the extreme maximum temperature in 2012 was 31.3 °C (June 24th) while the extreme minimum was -39.8°C (January 31st). Average annual precipitation was 201.6 mm, with the greatest proportion falling as rain during the warmer months (Figure 2-10). Average annual wind speed is 9.7 km/hour and prevails from the east for the majority of the year with the exception of May, June, and July when the wind direction shifts to northeast (Environment Canada, 2012).

Air temperatures near Inuvik have been increasing since the 1958-1970 time period, particularly during the colder months (e.g., November, December, January, and February) (Figure 2-11 and Table 2-3). The annual daily average maximum, minimum, and mean temperature of each successive time period has increased in this time (Figure 2-14). Furthermore, the variability of temperatures has also been affected. For example, in 2012 Inuvik experienced the coldest mean daily March temperature on record since 1970 followed by the warmest June and July (Environment Canada, 2012). Important fluctuations have also been observed in the timing, magnitude, variability and type of precipitation in a given year (Figure 2-12) (Table 2-4).

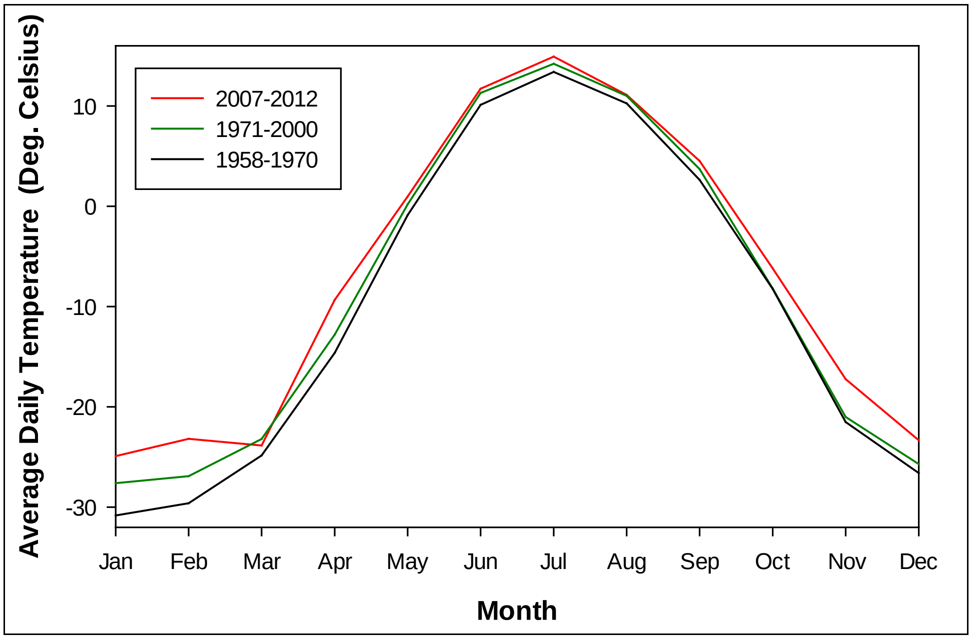


Figure 2-11: Means of daily average temperatures (^{o}C) for Inuvik, NWT over the periods 1958-1970, 1971-2000, and 2007-2012.

Table 2-3: Monthly mean of daily maximum, minimum, and mean temperatures for Inuvik, NWT from different time periods (1958-1970, 1971-2000, 2001-2006, and 2007-2012). Data were acquired from the Inuvik Airport Automated Weather Observation System.

	Daily Maximum (°C)				**Daily Minimum (°C)**				**Daily Mean (°C)**			
Month	**1958-1970**	**1971-2000**	**2001-2006**	**2007-2012**	**1958-1970**	**1971-2000**	**2001-2006**	**2007-2012**	**1958-1970**	**1971-2000**	**2001-2006**	**2007-2012**
Jan.	-25.7	-23.2	-21.6	-21.8	-35.9	-31.9	-27.8	-28.0	-30.8	-27.6	-24.7	-24.9
Feb.	-24.2	-22.0	-19.9	-19.7	-35.0	-31.7	-27.4	-26.5	-29.6	-26.9	-23.7	-23.2
Mar.	-18.4	-17.5	-18.7	-19.7	-31.2	-28.8	-27.0	-27.9	-24.8	-23.2	-22.8	-23.8
Apr.	-7.9	-7.1	-6.6	-4.8	-21.2	-18.4	-16.4	-13.7	-14.6	-12.8	-11.5	-9.3
May	4.2	5.0	3.7	3.4	-5.9	-4.7	-4.7	-3.9	-0.9	0.2	-0.5	1.0
Jun.	16.2	17.3	17.0	17.3	4.0	5.3	4.8	7.0	10.1	11.3	10.9	11.7
Jul.	19.2	19.8	18.1	19.8	7.5	8.5	8.1	10.0	13.4	14.2	13.1	14.9
Aug.	15.5	16.1	15.4	15.2	5.0	5.9	5.9	6.9	10.3	11.0	10.7	11.1
Sep.	6.8	7.8	8.3	7.8	-1.6	-0.4	1.4	1.3	2.6	3.7	4.8	4.5
Oct.	-4.6	-4.8	-2.5	-3.8	-11.8	-11.6	-7.7	-8.5	-8.2	-8.2	-5.1	-6.2
Nov.	-16.9	-16.8	-15.3	-14.6	-26.3	-25.1	-21.2	-19.9	-21.5	-21.0	-18.3	-17.2
Dec.	-21.5	-21.3	-17.8	-20.4	-31.6	-30.1	-23.7	-26.2	-26.6	-25.7	-20.8	-23.3
ANNUAL MEAN	**-4.8**	**-3.9**	**-3.3**	**-3.4**	**-15.3**	**-13.6**	**-11.3**	**-10.8**	**-10.1**	**-8.8**	**-7.3**	**-7.1**

SOURCES: Environment Canada - National Climate Data and Information Archive (http://climate.weatheroffice.gc.ca/Welcome_e.html)

Table 2-4: Monthly mean rainfall, snowfall, and total precipitation for Inuvik, NWT from different time periods (1958-1970, 1971-2000, 2001-2006, and 2007-2012). Data were acquired from the Inuvik Airport Automated Weather Observation System.

	Rainfall (mm)				**Snowfall (cm)**				**Total Precipitation (mm)**			
Month	**1958-1970**	**1971-2000**	**2001-2006**	**2007-2012**	**1958-1970**	**1971-2000**	**2001-2006**	**2007-2012**	**1958-1970**	**1971-2000**	**2001-2006**	**2007-2012**
Jan.	0.2	0.1	0	--	21.5	17.4	26.5	--	20.2	13.8	18.2	9.5
Feb.	0	0	0	--	13.9	15.0	30.8	--	12.4	11.6	18.3	5.4
Mar.	0	0	0	--	17.5	14.6	30.0	--	15.2	11.0	16.6	8.3
Apr.	0	0	0	--	18.5	13.5	24.6	--	17.2	10.5	17.8	7.6
May	5.4	6.1	9.4	--	14.5	13.1	7.8	--	18.3	17.0	15.0	8.4
Jun.	17.3	20.2	24.2	--	2.4	1.9	0	--	21.2	22.1	24.2	18.9
Jul.	37.5	32.9	33.0	--	0.2	0.3	0	--	37.7	33.2	33.0	37.5
Aug.	38.5	37.5	41.4	--	4.7	2.4	0.04	--	46.3	39.9	41.4	42.4
Sep.	11.5	18.7	18.8	--	10.5	10.7	13.7	--	20.7	28.0	28.1	29.8
Oct.	2.3	1.3	2.8	--	35.2	34.9	28.9	--	34.3	28.0	23.2	19.2
Nov.	0.1	0	0.8	--	19.2	23.7	39.0	--	15.2	17.8	26.3	8.4
Dec.	0	0	0	--	22.5	20.4	38.1	--	19.1	15.7	25.3	6.0
ANNUAL SUM	**112.9**	**117.0**	**[a]130.4**	**--**	**180.5**	**167.9**	**[a]239.5**	**--**	**277.9**	**248.4**	**[a]287.5**	**[b]201.6**

NOTES: [a]2001 data was unavailable; [b]Value displayed is based on incomplete data

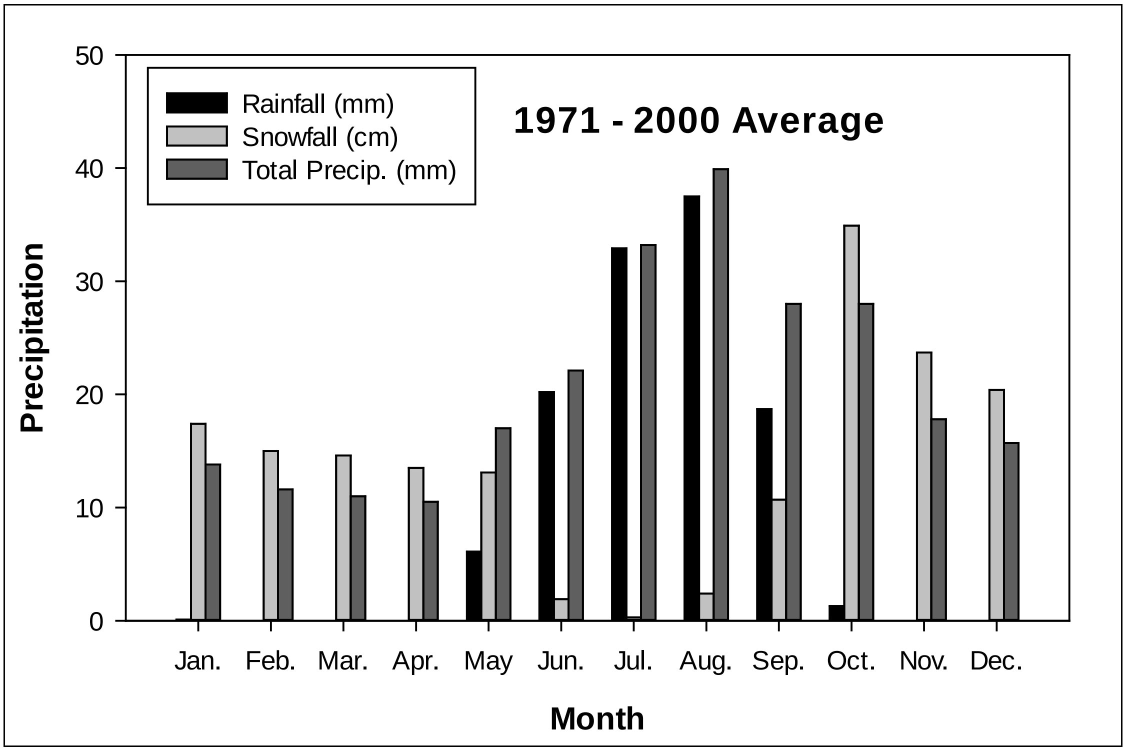


Figure 2-12. Mean monthly rainfall, snowfall, and total precipitation for Noell Lake, NWT.

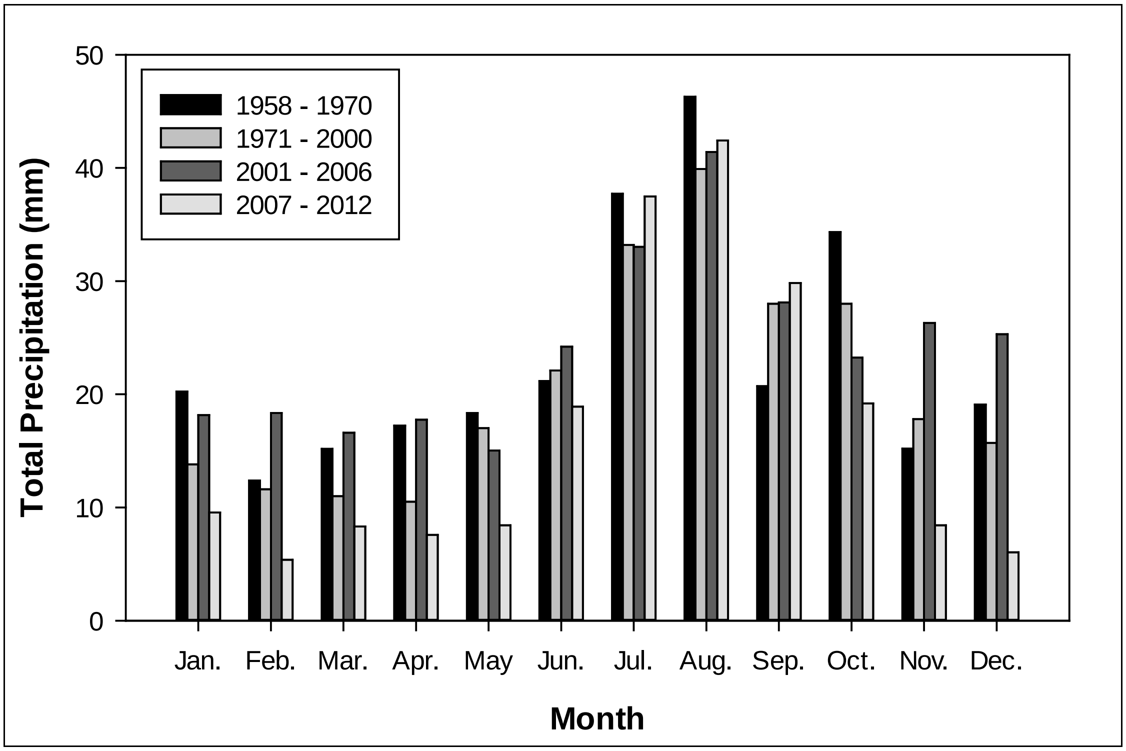


Figure 2-13: Mean total precipitation of Noell Lake, NWT for different time periods 1958-1970, 1971-2000, 2001-2006, and 2007-2012.

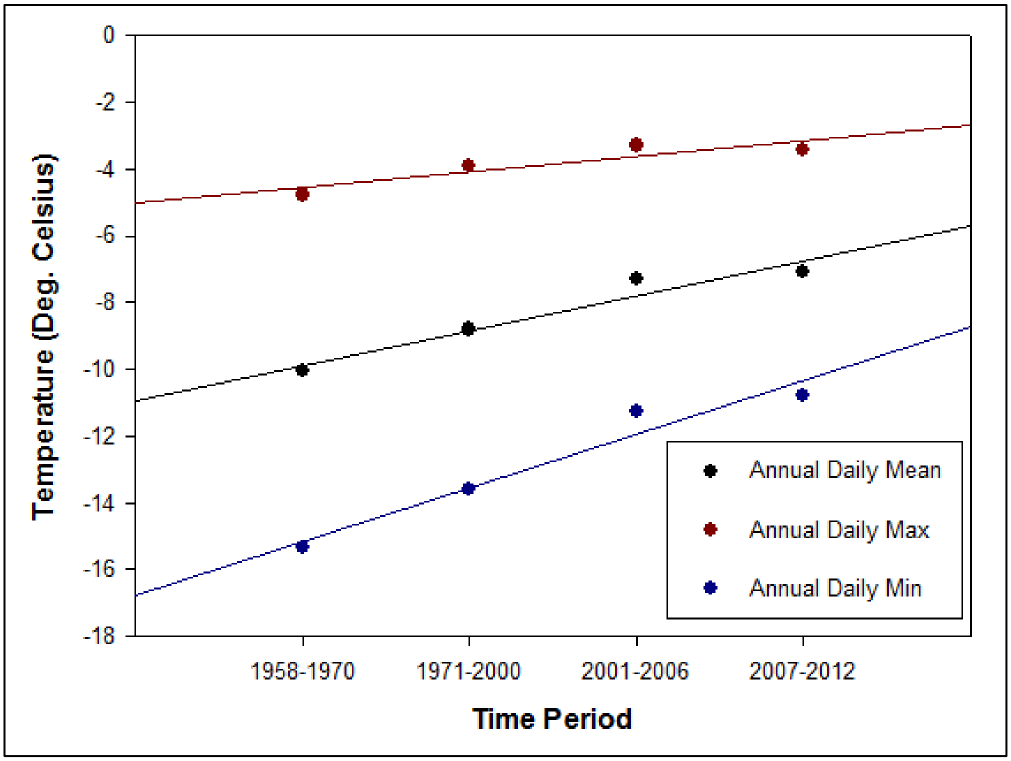


Figure 2-14: Mean annual daily average, maximum, and minimum temperatures of Inuvik, NWT over the time periods 1958-1970, 1971-2000, 2001-2006, and 2007-2012.

2.5 Water Quality

The hydrological regimes of lakes near Inuvik, such as Noell Lake, are influenced most strongly by the minimal inputs of precipitation, as well as the expansive presence of permafrost underlying most of the region. Generally speaking, the majority of Arctic lakes have low ionic concentrations since both over-land and subsurface runoff are quickly transported through an active layer with a low nutrient content (Lim and Douglas, 2003). The presence of permafrost impedes the infiltration of runoff and snowmelt to lower soil horizons and, thus, a significant portion of the hydrological processes are confined to the landscape surface (Kokelj et al., 2005; 2009). This magnifies the degree to which geomorphological parameters (i.e., disturbance, vegetation, parent materials) regulate the geochemistry of Arctic lakes (Pienitz et al., 1997a; Rühland et al., 2003; Kokelj et al., 2005).

Runoff from snowmelt is the central hydrological event affecting Noell Lake and induces considerable seasonal variations in lake water chemistry. Arctic lakes can exhibit substantial fluctuations in the concentrations of major cations and anions (e.g., Welch and

Legault, 1986; Cornwell, 1992) due to offsetting processes of dilution and concentration. In the springtime, as snowmelt contributes runoff into lakes, the overall lake concentrations of ions are diluted; conversely, ion concentration can occur as a result of exclusion by evaporation in the summer, and exclusion by ice-formation and/or permafrost processes in the winter (Hobbie, 1984; Kling et al., 1992). As a result, time of year is an important factor to consider when describing or analyzing the water chemistry data of a given lake.

The water quality of Noell Lake was studied by the Department of Fisheries and Oceans in 1982 (Read and Roberge, 1986) and 1986 (Anema et al., 1990b), and by Ogbebo et al. (2009a) in 2007 as part of a greater limnological transect-investigation in the lower Mackenzie River Basin. Chemical and physical water quality data of Noell Lake are included in Table 2-5. Nutrient, chlorophyll *a*, and Secchi depth data of Noell Lake are included in Table 2-6.

Table 2-5: Inter-annual average concentrations for major ions and related water quality variables for Noell Lake. Average concentrations are in mg/L except for pH in pH units, water temperature in (^{o}C), and specific conductivity in (µs/cm).

	August Mean 1982[b]	**Summer Mean 1986**[c]	**July 2007**[a]
pH	--	7.6	8.2
Water Temperature	13.5	10.2	15.7
Dissolved Oxygen	9.0	--	9.8
Specific Conductivity	--	71.1	67.3
Mg	1.2	2.0	--
Na	1.9	2.4	2.5
K	0.6	1.0	--
Cl	2.4	2.1	1.7
Ca	4.7	7.0	7.1
SO_4	9.5	10.9	11.0
SiO_2	--	--	0.4
Fe	--	0.04	--
Mn	--	0.01	--

SOURCES: [a]Ogbebo et al., 2009; [b]Read and Roberge, 1986; [c]Anema et al., 1990b

Table 2-6: Inter-annual average concentrations of nutrient, chlorophyll-*a*, and Secchi depth data for Noell Lake. Average concentrations are in µg/L unless designated otherwise.

	August 1982[b]	**Summer Mean 1986[c]**	**July 2007[a]**
Secchi Depth (m)	--	--	2.8
Total Phosphorous	--	--	7.8
Dissolved Phosphorous	70.0	--	1.8
Total Nitrogen	--	--	201.5
Total Dissolved Nitrogen	1000.0	--	192.8
NH_3	--	--	6.3
$NO_3 + NO_2$	--	--	5.0
Chlorophyll a	0.3	0.8	1.6
Particulate Organic Carbon	--	--	103.0
Dissolved Organic Carbon (mg/L)	--	--	4.9
Dissolved Inorganic Carbon (mg/L)	--	--	4.3

SOURCES: [a]Ogbebo et al. 2009; [b]Read and Roberge 1986; [c]Anema et al. 1990b

Using particulate organic carbon (POC) to chlorophyll-*a* ratios, Ogbebo et al. (2009b) determined that Noell Lake relies mainly on autochthonous carbon as its source of carbon. Current research indicates that small northern lakes experience significant watershed impacts and, thus, a greater portion of carbon is derived from allochthonous sources (Hamilton et al., 2001). Since Noell Lake is a comparatively large lake for this region, its reliance on carbon sources emanating from outside its immediate watershed is, therefore, anticipated.

During the open-water season of 1982, Read and Roberge (1986) used digital oxygen meters (YSI Model 58) to generate depth-profiles of temperature and dissolved oxygen over multiple days (Figure 2-15). It is evident that at the time sampling, Noell Lake was well-mixed throughout the entirety of the measured depth-profile.

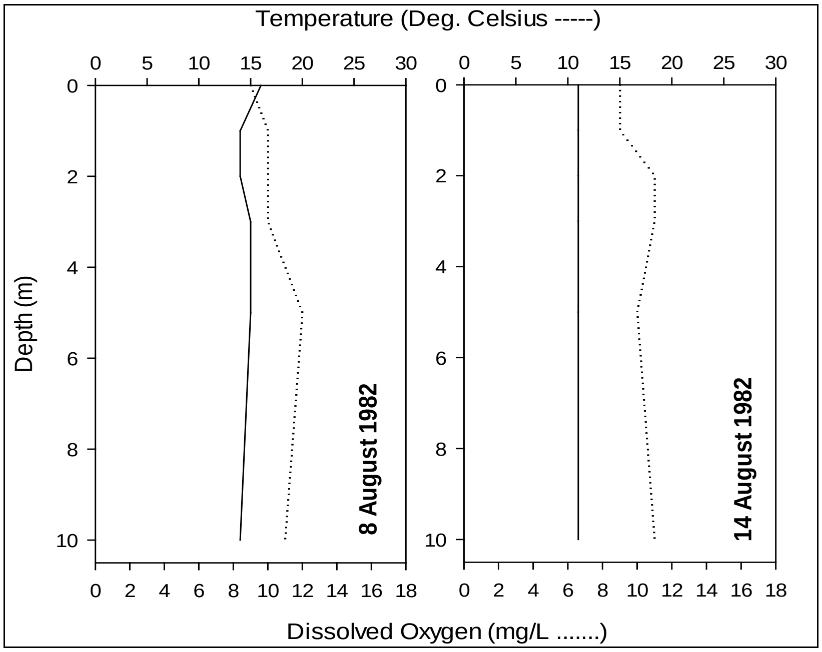


Figure 2-15: Open-water temperature and dissolved oxygen profiles of Noell Lake, NWT 1982.
SOURCE: Read and Roberge 1986

2.6 Biological Characteristics

The ecosystem of Noell Lake is characterized by primary producers (e.g. phytoplankton, microbial mats), secondary producers (e.g. zooplankton), and fish such as lake trout, northern pike, and Arctic grayling. Due to the relatively diverse abundance of fish species in Noell Lake, various subsistence, recreational, and commercial fishing activities are supported.

2.6.1 Plants

The composition of the phytoplankton floral community in Arctic lakes is similar to that of more temperate lakes with green algae, chrysophytes, diatoms, and cyanobacteria comprising approximately 80% of the species composition (Sheath, 1986). Chrysophytes (Charvet et al., 2011) and cryptophytes (Sheath, 1986) contribute the largest percentage of overall biomass out of the nanophytoplankton community and flagellates, including Chrysophyta, Cryptophyta, and Dinophyta.

Table 2-7: Summary of phytoplankton biomass and cell numbers in Noell Lake during open-water periods of 1985 and 1986.

		Biomass (mg/m^3)		% Composition based on Biomass						
Date	**Station No.**[a]	**Phyto**	**Proto**	**Cyano**	**Chloro**	**Euglen**	**Chryso**	**Diatom**	**Crypto**	**Peridi**
02/07/86	30	190.0	44.0	1.6	5.6	0	54.2	8.0	8.2	22.5
07/07/86	38	213.0	19.0	0.5	5.1	0	52.5	13.5	6.2	22.2
14/07/86	45	184.0	21.0	1.8	6.2	0	50.5	15.8	12.3	13.5
29/07/86	58	121.0	11.0	0.8	1.4	0	56.1	10.8	4.4	26.5
25/08/86	89	104.0	113.0	14.4	17.3	0	37.2	9.2	5.6	16.3
04/09/86	105	147.0	62.0	1.0	8.8	0	48.8	6.6	5.2	29.7

		[b]Cell No./m^3		% Composition based on Cell Number						
Date	**Station No.**[a]	**Phyto**	**Proto**	**Cyano**	**Chloro**	**Euglen**	**Chryso**	**Diatom**	**Crypto**	**Peridi**
02/07/86	30	3.05E+06	8170.0	17.2	2.0	0	71.5	3.0	6.0	0.3
07/07/86	38	8.60E+06	3610.0	64.9	1.2	0	27.5	3.3	3.0	0.1
14/07/86	45	6.33E+06	3610.0	58.4	3.4	0	30.9	2.9	3.9	0.5
29/07/86	58	2.56E+06	2470.0	11.5	2.8	0	78.5	2.6	4.5	0.1
25/08/86	89	5.02E+06	6840.0	68.6	3.3	0	24.8	1.2	2.0	0.1
04/09/86	105	7.08+06	3040.0	65.3	2.3	0	29.5	1.5	1.4	0

SOURCE: Ramlal et al. 1991

NOTES: [a]Station numbers refer to stations used in Fee et al. 1988; [b]E+06 means x 1,000,000

The phytoplankton of Noell Lake was studied by the Department of Fisheries and Oceans (DFO) throughout the open water periods of 1985 and 1986 (Ramlal et al., 1991) (Table 2-7). With the exception of water samples taken in late August where protozoa biomass exceeded that of phytoplankton, both the biomass and cell numbers indicate that phytoplankton are more common than protozoa in the water of Noell Lake. Broadly speaking, Noell Lake's algal community was dominated by golden algae (*Chrysophyceae*) throughout the duration of the open water period with respect to both biomass and cell number (Table 2-7). The blue-green algae (*Cyanophyta*) were the most dominant algal form with respect to cell number; however, they were comparatively less common when considered in the context of the overall algal biomass. This is due primarily to the small size cell of the blue-green algae in comparison to the other algae groups. While the diatoms (*Bacillariophyta*) and flagellate protists (*Dinophyceae* – Peridi in Table 2-7) represented a relatively small fraction of total algal composition based on cell number, they had a comparatively higher percent composition with respect to biomass, especially the flagellate protists. The green algae (*Chlorophyta*) and brownishgreen (*Cryptophyceae*) algae exhibited a similar pattern although the difference between the two was less significant. The euglenoid flagellates (*Euglenaphyta*) are generally found in eutrophic environments and, thus, were undetected in the oligotrophic waters of Noell Lake.

Water samples for chlorophyll-*a* analysis were taken from Noell Lake by the Western Region branch of DFO in 1982 (Read and Roberge, 1986) and 1986 by the Central and Arctic Region branch (Ramlal et al., 1991) (Figure 2-16). Studies undertaken by Ogbebo et al. (2009b) demonstrated that phytoplankton communities of lakes situated east of the Mackenzie River (such as Noell Lake) do not have access to sufficient resources of nitrogen and phosphorous relative to their needs.

Open-Water:

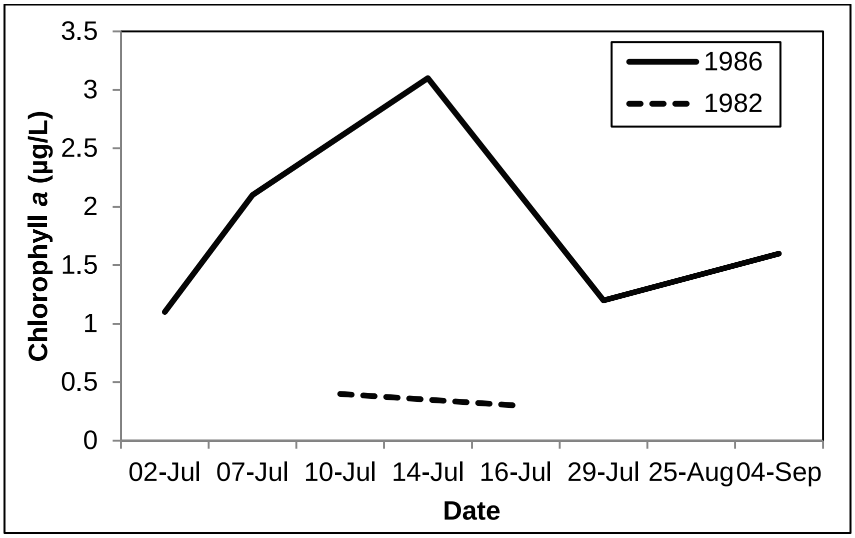


Figure 2-16: *In-situ* measurements of chlorophyll-*a* during the 1982 and 1986 open-water seasons of Noell Lake, NWT.

2.6.2 Invertebrates

Variations in Arctic zooplankton species composition are a function of dissimilar glacial histories, dispersal rates and abilities, and environmental conditions (Rautio et al., 2011). Pleistocene glaciations have had a considerable influence in shaping Arctic freshwater zooplankton communities (Weider and Hobaek, 2000). Broadly speaking, zooplankton species diversity in Arctic lakes such as Noell Lake is relatively poor compared to temperate lakes. Short growing seasons, high exposure to ultraviolet radiation in summer, and freezing stresses in winter present extreme living conditions in which zooplankton must survive. Studies by McCart et al. (1974) determined that the benthos of Noell Lake outflow included ephemeropterans, plecopterans, dipterans (e.g., Tipulidae, Chironomidae, Empidiae), nematodes, oligochaetes, arachnids, and other assorted invertebrates.

2.6.3 Fish

Nine species of fish have been reported in Noell Lake: burbot, least cicso, lake trout, lake whitefish, slimy sculpin, ninespine stickleback, Arctic grayling, northern pike, and round whitefish (Read and Roberge, 1986; Unpublished Collection Report Form 2009, 2010). In 1982, DFO undertook a descriptive campaign involving creel censuses, experimental gillnetting, tagging, and biological sampling in Noell Lake (Read and Roberge, 1986). Lake trout was the main species caught with northern pike being the only other species caught at that time (Table 2-9).

Tables 2-8: Estimated angler numbers, effort, and harvest from Noell Lake. Estimates are based on creel survey data collected on 4 days (August 7th, 8th, 16th, and 17th) in 1982.

	Lake Trout	**Northern Pike**	**Total**
Number of Anglers	--	--	10.00
Angler-hours	--	--	28.50
Total Number of Fish Harvested	10.00	1.00	11.00
Catch/Angler	2.00	0.20	2.20
Catch/Angler-Hour	0.35	0.040	0.40

SOURCES: Read and Roberge 1986

Additional DFO sampling was undertaken in 2009 and 2010 as part of a broader scientific study (Unpublished Collection Report Form 2009, 2010). In 2009, the most common species caught during this campaign was lake trout with least cisco also being abundant. The 2010 campaign yielded comparatively less fish with least cisco being the most common catch (Table 2-9).

Table 2-9: Species caught in Noell Lake using angling, gill nets, seine nets, and ponar grabs. 2009 sampling took place from June 13th-17th and on September 25th, 2010 sampling took place in August and September.

Species Caught	[a]Total No. Caught in 2009	[b]Total No. Caught in 2010
Burbot	2	--
Least Cisco	17	4
Lake Trout	30	2
Lake Whitefish	6	2
Ninespine Stickleback	3	1
Northern Pike	2	2
Slimy Sculpin	2	--
Round Whitefish	--	2

SOURCES: Unpublished Collection Report Form [a]2009, [b]2010

In 1993, the community of Inuvik prepared a list of ten fish species that it considered to be of greatest significance to traditional fishing in the area (Inuvik, 1993). Of the ten species listed, six can be found in Noell Lake: northern pike, lake whitefish, least cisco, lake trout, Arctic grayling, and burbot. As a result, Noell Lake serves as an important element in the overall management and conservation of these fish species with respect to traditional fishing in the region.

2.6.4 Wildlife

Comprehensive studies of the wildlife traversing the Noell Lake catchment region have been undertaken. In a 1982 environmental impact statement, Dome Petroleum Ltd. (1982) reported 33 species of terrestrial mammals living in the Mackenzie Delta and Arctic Coastal Plain including several species of shrew (e.g., Masked, Dusky, Arctic), hare (e.g., Showshoe, Arctic), squirrel (e.g., Arctic ground, American red), lemming (e.g. Brown, Collared), vole (e.g. Singing, Meadow, Tundra, Northern red-backed), fox (e.g., Arctic, Red), and bear (e.g., American black bear, Grizzly); as well as the Collared Pika, Muskrat, Coyote, Wolf, American Marten, Ermine, Least weasel, American Mink, Wolverine, River otter, lynx, moose, caribou, Dall’s sheep, and muskoxen.

The Mackenzie Delta and Tuktoyatuk Peninsula provide both permanent and seasonal habitat for over 100 species of birds. Significant species include geese (e.g., Canada, brant, snow, white-fronted), tundra swans, loons (e.g., common, yellow-billed, Pacific, red-throated), ducks (e.g., scoter, pintail, scaup, mallard, wigeon), eagles (e.g., bald, golden), rough-legged hawks, peregrine falcons, gyrfalcons, sandhill cranes, willow ptarmigans, and snowy owls (Rescan Environmental Services Ltd., 1998).

2.7 Ice Phenology, Vertical Temperature Distributions, and Mixing Characteristics

The shallow depths of small Arctic lakes and ponds prevent the majority of these small water bodies from undergoing any sustained stratification (Rautio et al., 2011); however, larger lakes such as Great Bear Lake are monomictic, with stratification occurring in summer (Johnson, 1975). The degree to which Noell Lake stratifies is unknown, and a vast majority of Arctic lakes have no available data on inter- and intra-annual variability of vertical temperature distributions (Kirillin et al., 2011). Seasonal changes to the vertical temperature profile and mixing regime of a lake has important consequences to assessing water quality, local climatology, response to global warming, and the recreational potential of lakes (Kirillin et al., 2011).

The open-water period of Noell Lake typically lasts from mid- to late-June until mid- to early-October. As the colder, winter season commences, net energy losses from the lake body to the atmosphere facilitate the development of ice-cover along the peripheries of the lake (AMAP, 2011). Given tranquil meteorological conditions, stable ice formation will progress concentrically across the entire lake surface given the relatively round, oval shape of Noell Lake. Pending further net energy losses from the water column, ice cover will progress vertically downward towards the lake bottom (AMAP, 2011). Ice forming under the surface of the lake is relatively transparent and commonly referred to as ‘blue’, ‘black’, and/or ‘congelation’ ice (AMAP, 2011).

2.8 Summary

This represents the first comprehensive compilation of pertinent climatic, hydrological, limnological, and ecological information pertaining to a western Canadian Arctic tundra lake. Through the integration of relevant material comprising both published and unpublished scientific sources, a multitude of significant geomorphological, geological, geochemical, and environmental information pertaining to Noell Lake was assembled. More specifically, information pertaining to phytoplankton, macrophyte, epipphyton, epipelon and fish communities were included, as well as results from historical water quality sampling, analyses of water column organic carbon and chlorophyll-*a*, open-water depth profiles of dissolved oxygen and chlorophyll-*a*, drainage basin characteristics, recent and historical forest fire histories, lake watershed characteristics, and information on wildlife.

More generally, Noell Lake is a typical, medium-sized oval-shaped Arctic lake believed to be dimictic, with a lake-ice regime that generally experiences freezeup in mid- to late-October, and undergoes breakup in mid- to late-June. Compared to other non-flooded, Arctic tundra lakes measured by Ogbebo et al. (2009a), Noell Lake was reported to contain some of the lowest ionic concentrations (e.g., conductivity, Na, Cl, SO_4, and DIC). Ogbebo et al. (2009a) determined that parameters encompassing the ionic concentrations of water (i.e. conductivity, Na, Cl, Ca, SO_4, and DIC) were correlated with each other, and negatively correlated with both elevation and latitude. Since Noell Lake was both the farthest south and the second highest in elevation out of the 9 lakes investigated, it possibly explains why ionic concentrations were low; since Noell Lake is the farthest south, it is the farthest away from the ionic inputs of the Arctic ocean (Evans and Grainger, 1980). The composition of the phytoplankton floral community in Noell Lake is similar to that of more temperate lakes with green algae, chrysophytes, diatoms, and cyanobacteria comprising approximately 80% of the species composition (Sheath, 1986). Chrysophytes (Charvet et al., 2011) and cryptophytes (Sheath, 1986) contribute the largest percentage of overall biomass out of the nanophytoplankton community and flagellates, including Chrysophyta, Cryptophyta, and Dinophyta.

Importantly, the construction of an all-season road directly through the catchment of Noell Lake, in addition to the concomitant effects of demonstrated climate warming in the area, are likely to impose significant consequences on the water quality, stratification-regime, and food-web structure of Noell Lake in the future. Comprehensive monitoring of significant ecological, physical, and hydrological parameters of Noell Lake can divulge how similar waterbodies may respond to similar climate stressors and anthropogenic disturbance elsewhere. The establishment of reliable baseline conditions for the current ecology and limnology of Noell Lake is, therefore, crucial to rigorous environmental assessments, comprehensive monitoring initiatives, and well-informed stewardship of the region in the future.

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CHAPTER 3: VALIDATION OF AUTOMATED AND NON-AUTOMATED BUOY AND SUBSURFACE MOORING COMPONENTS

Abstract

Increasing awareness surrounding the usefulness of Arctic lakes and ponds in reflecting climate change has initiated demands for the development and implementation of research initiatives designed to track changes to the ecological integrity of these systems. There is an urgent need for improved spatial and temporal coverage of Arctic lake ecosystems. Research attempting to investigate climate-driven changes in northern lake ecosystems is complicated by a legacy of relatively instantaneous research initiatives predicated on defining the state of a given lake ecosystem given principally transitory lake-water samples analyzed to reveal traditional and physical parameters indicative of only the days or hours immediately before sampling. From September 2010 until July 2013, the validity of a series of both automated and non-automated buoy and subsurface mooring systems in relaying continuous and real time measurements was tested. In evaluating the overall efficacy and utility of using buoy and mooring components to record uninterrupted meaningful physical, chemical, and biological parameters in an Arctic lake over consecutive open-water and under-ice seasons, three main criteria were considered: (1) Functionality; (2) Spatial Homogeneity; and (3) Measurement Validity. Overall, the various buoy and mooring components successfully recorded uninterrupted limnological parameters in an upland, Arctic tundra lake in both the epilimnion and hypolimnion.

Keywords: Automated buoy; Subsurface mooring; Continuous monitoring; Validation; Spatial heterogeneity

3.1 Introduction

Lakes and ponds comprise a vast and well-distributed linkage of ecosystems that reflect the on-going responses of terrestrial aquatic ecosystems to climate change (Williamson et al., 2009). More importantly, since polar regions are expected to continue to experiencing significant warming (Kattsov et al., 2005; Prowse et al., 2006), Arctic lakes possess an even greater capacity to reflect changes due to climate warming, as well as illuminate how aquatic ecosystems are likely to respond in the future. Arctic lakes and ponds can serve as crucial reference ecosystems to which recent and future global environmental change can be compared (Lim and Douglas, 2003). Changes affecting the hydrological parameters of this region (e.g., source and timing of water inputs, evaporation, and ice-cover extent and duration) are both instigating and interacting simultaneously with changes in biogeochemical cycling and habitat availability **(**Prowse et al., 2006; Mesquita et al., 2010; AMAP, 2011; Prowse et al., 2011a,b; Scholes et al., 2013). Broadly speaking, the compounding effects of reductions in ice cover extent and duration, warmer ambient air and water temperatures, and enhanced nutrient and dissolved organic carbon (DOC) supplies derived from more biogeochemically active catchments have the potential to enhance lentic primary productivity in Arctic lakes (Bonilla et al., 2005; Prowse et al., 2006; Bonilla et al., 2009; Antoniades et al., 2011).

Recently, increasing emphasis on the importance of understanding how Arctic lakes and ponds are affected by climate change have catalyzed attempts at developing and implementing research initiatives designed to track changes to the ecological integrity of these systems (e.g., Lim and Douglas, 2003; Rautio et al., 2011; MacDonald et al., 2012). Developing the capacity to effectively track changes in this environmentally sensitive area is urgently needed (Schindler and Smol, 2006). However, it is important that monitoring initiatives be designed to accommodate and capture the unique conditions that characterize Arctic lake ecosystems, as well as differentiate climate-induced changes from the confounding effects of other anthropogenic forces (e.g., acidification and eutrophication) (Williamson et al., 2009).

Despite recent increases in the number of limnological investigations undertaken across the circumpolar Arctic - Alaska (e.g., LaPerriere et al., 2003); western Canadian Arctic (e.g., Ogbebo et al., 2009); eastern Canadian Arctic (e.g., Westover et al., 2009);

Finland (e.g., Luoto, 2009); Greenland (e.g., Cremer et al., 2005); Norway (e.g., Løvik and Kjellberg, 2003); Russia (e.g., Moiseenko et al., 2009); Sweden (e.g., Brunberg et al., 2002) – scientific, ecological investigations of temperate lakes overwhelmingly outnumber those of Arctic lakes (MacDonald et al., 2012). Furthermore, there have historically been few long-term integrated hydrological, geochemical, and biological monitoring programs situated in northern latitudes (e.g., Douglas and Smol, 1993, 1994, 1995; Michelutti et al., 2003; ACIA, 2004). As a result, there is an urgent need for improved spatial and, more importantly, temporal coverage of Arctic lake ecosystems.

Uninterrupted and long-term monitoring programs of Arctic freshwater ecosystems have not been undertaken due to the financially and logistically prohibitive nature of northern research. Continuous, high-frequency sampling is challenging to undertake and sustain in Arctic lakes, particularly in those under government regulation, since they are generally far away from towns or established centres of human activity (MacDonald et al., 2012). Moreover, northern research is technically challenging, subjected to extreme weather conditions, and study lakes are commonly only accessible by snowmobile and/or helicopter, imposing additional financial constraints on any research initiative (Hille, 2010).

Groups and organizations responsible for the management and stewardship of these areas typically do not have the financial resources necessary to undertake the uninterrupted, recurrent sampling regimes necessary to produce meaningful and instructive long-term data sets (MacDonald et al., 2012). Research attempting to investigate climate-driven changes in northern lake ecosystems is complicated by a legacy of relatively instantaneous research initiatives predicated on defining the state of a given lake ecosystem given principally transitory observations of the lake under investigation (Bailey et al., 2004). Furthermore, these momentary observations have traditionally been lake-water samples analyzed to reveal traditional physical and chemical parameters, indicative of only the days or hours immediately before sampling (Bailey et al., 2004). Not only are these “snapshot” samples limited in their temporal scope, but they also fail to reflect and integrate the other relevant biological, physical, and chemical conditions of the lake crucial to defining its state.

The combination of these challenges has resulted in sparse baseline environmental data available for analysis and comparison (Lim and Douglas, 2003). Identifying and forecasting the effects of climate change in northern lakes, especially in light of the anthropogenic and environmental stressors acting concomitantly on these ecosystems, are crucial to the stewardship and management of these aquatic resources moving forward. Monitoring initiatives that are well-suited to the challenges of Arctic environments, encompassing a broad scope of physical, chemical, and biological parameters over a consistent period are needed to accurately define the current state of these systems.

The efficacy and utility of multi-component buoy and mooring devices to record continuous, *in-situ* integrated climatological, hydrological, chemical, and biological parameters was tested. In evaluating the overall efficacy and utility of the various instrument systems to record uninterrupted meaningful physical, chemical, and biological parameters in the epilimnion and hypolimnion of an Arctic lake over consecutive open-water and under-ice seasons, three main criteria were considered: (1) were the sensors responsible for recording hydro-ecological data functional for a significant length of time? (apparatus functionality); (2) were the measurements being recorded by the sensors representative of the entire lake? (spatial homogeneity); and (3) were the values recorded by the sensors representative of the true parameter-values of the lake? (measurement validity).

3.2 Methods

A continuous integrated monitoring program of the hydro-ecology of Noell Lake, NWT was established using a multi-component Arctic Lake Monitoring System (ALMS) that was deployed from September 2010 to July 2013. The ALMS collected continuous *in-situ* time-series during open-water and under-ice seasons, and in both the epilimnion and hypolimnion, of key climatological, hydrological, chemical, and biological parameters. The ALMS consisted of several coupled sub-components: (1) a primary automated Arctic Lake Buoy and Subsurface Mooring System (AXYS Technologies, Inc.) (Section 3.2.2); (2) a supplementary Subsurface Mooring System (Section 3.2.3); and (3) an Instrumented Subsurface Mooring (Section 3.2.3) designed at the University of Victoria and assembled in Inuvik, NWT. Through the amalgamation of recorded data from all three components, a multitude of temporally continuous environmental measurements (e.g., water temperature, conductivity, specific conductance, dissolved oxygen saturation, dissolved oxygen, pH, chlorophyll *a*, oxidation-reduction potential, blue green algae, total dissolved solids) were recorded at two separate depths (e.g., 3 metres and 9 metres below the water surface).

3.2.1 Study Lake

Limnological investigations were conducted in Noell Lake, NWT (68^{o}31’37’’ N, 133^{o}30’48’’ W), which is located 15 km NE of Inuvik, NWT and just north of the treeline (Ogbebo et al., 2009). Noell Lake is an oval-shaped upland tundra lake situated 89.9 metres above sea level in the southern extent of the Husky Lakes Basin (de Rham and Carter, 2009). Noell Lake is underlain by bedrock composed primarily of carbonate and shale (Kokelj et al., 2005). The lake has a surface area of 30 km^2 and maximum depth of 18.3 m^2 (de Rham and Carter, 2009). Noell Lake has a singular outflow, Jimmy Creek, which flows northward from Noell Lake into Jimmy Lake (Read and Roberge, 1986).

3.2.2 Functionality

3.2.2.1 Arctic Lake Monitoring System (ALMS) Buoy and Subsurface Mooring System (AXYS Technologies, Inc.)

On September 28th, 2010 an Arctic Lake Monitoring Systems (ALMS) Buoy and Subsurface Mooring System (AXYS Technologies, Inc.) were deployed in Noell Lake, NWT at (68^{o}18.378’’N; 133^{o}29.377’’W). Figure 3-1 is a schematic representation of the various components of both the ALMS Buoy and Mooring System during the open-water season of Noell Lake.

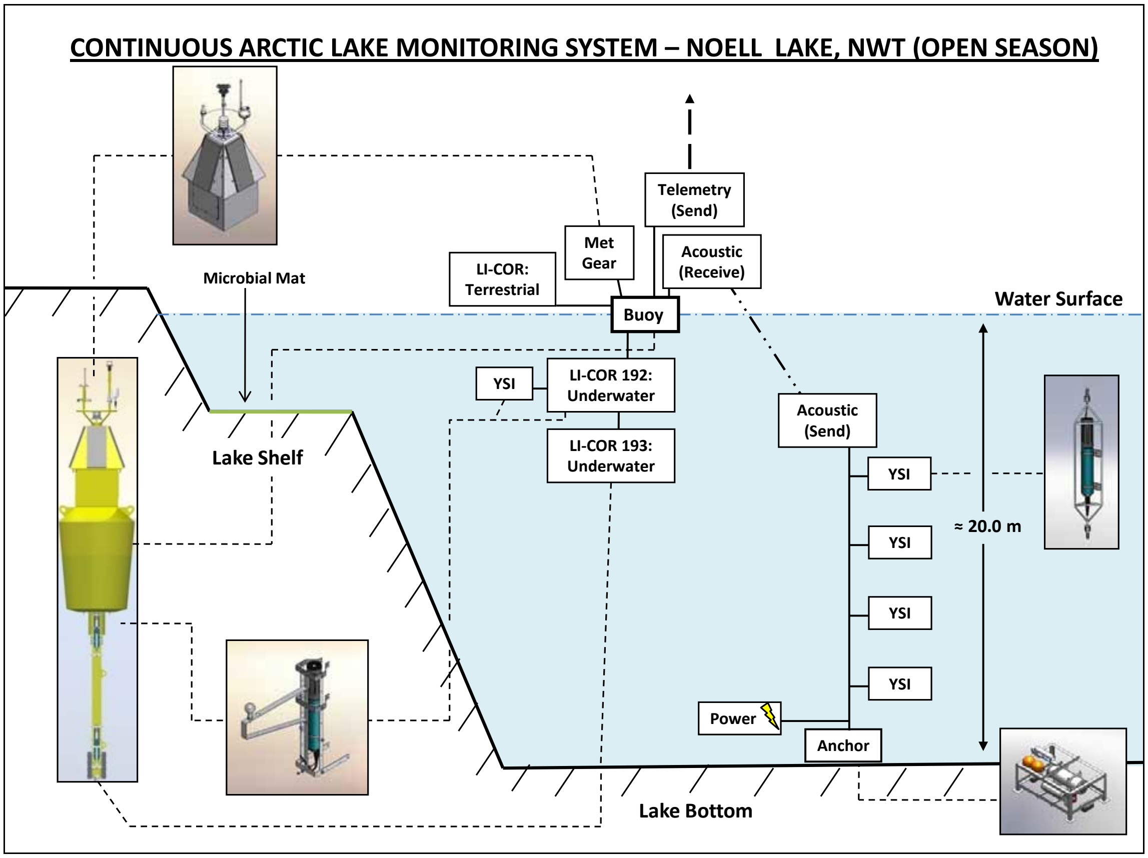

Figure 3-1: Schematic representation of the ALMS Buoy and Subsurface Mooring System deployed in Noell Lake, NWT. Met Gear: Anemometer – windspeed; temperature and relative humidity sensors; pyranometer – radiation; barometer – pressure. YSI: Yellow Springs Instruments, water quality sonde. LI-COR: light meter.

The ALMS buoy consists of three sub-components: (1) the upper housing, (2) the middle battery compartment, and (3) the bottom I-Beam and sensor assembly. Mounted to the upper housing section were various meteorological instruments including a compass (Precision Navigation Instruments, Model: TCM 2.6), windsonic anemometer (Gill Windsonic), temperature and relative humidity sensors (Rotronics, Model: MP101A), pyranometers (LI-COR, Model: LI-200SA), barometer (Sutron, Model: 5600-0120-1), and global positioning system (Skywave, Model: DMR-800L). The round, Middle Battery Compartment contains the battery (Sunlyte, Model: 12-5000X) and assists with the flotation of the buoy. The battery compartment is comprised of steel and foam and provides a dry space for the battery to be stored. The underwater I Beam of the buoy has two YSI/LI-COR assemblies (YSI6600), two light sensors (M701-S Solar LED), and an acoustic modem. Table 3-1 includes a description of the parameters measured by the various instruments installed on the buoy, as well as the approximate depths the instruments were mounted.

Table 3-1: *In-situ* instrumented components of the ALMS buoy deployed in Noell Lake in 2010. The parameters measured, as well as the initial deployment depths of the sensors, are provided.

Component:	*Meteorology*	*Radiation*	*YSI (Shallow)*	*YSI (Deep)*
Instrument Depth:	Above water surface	Above water surface	**1 m** below water surface	**3 m** below water surface
Parameters:	Lat., long., wind speed and direction, air temp., humidity, dew point, pressure	Incoming shortwave radiation	Water temp. (^{o}C), conductivity (µS/cm), total dissolved solids (µS/cm), salinity (µS/cm), pressure (Pa), pH (pH units), ORP (mV), turbidity (NTU), chlorophyll-*a* (µg/L), dissolved oxygen saturation, dissolved oxygen (mg/L), blue green algae (cells/mL)	

The subsurface mooring system was designed to work in tandem with the ALMS buoy and consists of three components: (1) the frame and mounted components, (2) the YSI multiparameter water quality sonde assemblies, and (3) the ASL ice profiler and floats. The frame contains both an acoustic modem and release. The four YSIs (YSI, Model: 6600) float in a vertical profile above the frame. The ice profiler (ASL, Model: MN-SWP15) floats above the YSI assemblies, closest to the water surface. The subsurface mooring is powered by 100 D cell batteries located in the Prevco Subsea housing. Table 3-2 provides the depth below the water surface the instruments were installed at on the Subsurface Mooring system, as well as the parameters measured.

Table 3-2: *In-situ* instrumented components of the Subsurface Mooring deployed in Noell Lake in 2010. The parameters measured, as well as the initial deployment depths of the sensors, are given in the text.

Component:	*YSI 1*	*YSI 2*	*YSI 3*	*YSI 4*	*Ice Profiler*
Instrument Depth:	**4.5 m** below water surface	**7.25 m** below water surface	**9 m** below water surface	**11.25 m** below water surface	**3.5 m** below surface
Parameters:	Water temp. (^{o}C), conductivity (µS/cm), total dissolved solids (µS/cm), salinity (µS/cm), pressure (Pa), pH (units of pH), oxidation reduction potential (mV), turbidity (NTU), chlorophyll-*a* (µg/L), dissolved oxygen saturation, dissolved oxygen (mg/L), blue green algae (cells/mL), battery voltage (mV)				Range, persistence, amplitude

The buoy and mooring system used a low-power computing system (Watchman500) to collect, process, store, and relay the limnological and meteorological data measured by the various instruments. The Watchman500 integrates the diverse sensors and subsequently formats the data before sending it to shore via the acoustic modem. A copy of the data is also stored internally in a data logger located in the upper housing section. The buoy is powered by 4 x 20 Watt solar panels and 3 banks of 5 x 17 AHr lead acid batteries.

3.2.2.2 Supplemental Subsurface Mooring (AXYS Technologies, Inc.) and Instrumented Subsurface Mooring

To extend the temporal range of the environmental and hydrological variables measured by the initial ALMS buoy and mooring components, a second subsurface mooring component was deployed through the ice of Noell Lake on November 22nd, 2012 (N 68^{o}31.486''; W 133^{o}36.497). Similar to the design of the initial mooring component, the supplementary mooring component (Figure 3-2) had four YSI multiparameter water quality sonde assemblies (YSI, Model: 6600) mounted in a vertical profile with an ASL ice-profiler (ASL, Model: MN-SWP15) located closest to the water surface. The four YSI assemblies were located at approximately the same successive depths in the water column as the YSI sondes of the initial mooring component.

An additional apparatus was assembled by attaching three YSI multiparameter water quality sonde assemblies (YSI, Model: 6600) to a cable tied between an anchor at the lake-bottom and a marine float-device at the lake-surface. This apparatus is referred to as the instrumented subsurface mooring and was deployed on July 15th, 2012 and retrieved July 3rd, 2013. The YSI assemblies were attached to knotted loops in the rope through the use of spring-loaded gate carabineers (Figure 3-2). The instrumented subsurface mooring was initially deployed and anchored at (68°18.378’’N; 133°29.377’’W), close to the initial ALMS buoy and subsurface mooring components. HOBO data loggers (Onset Computer Corp.) programed to record water temperature every hour were strapped to the cable at one-metre intervals with the use of plastic cable-ties.

Figure 3-2 is schematic representations of both the supplementary subsurface mooring (right) and the instrumented subsurface mooring (left) components during the open-water period. Refer to Table 3-3 for a description of the parameters measured by the instruments installed on the supplementary subsurface mooring, as well as their approximate depth below the water surface. Refer to Table 3-4 for a description parameters measured by the instruments installed on the instrumented subsurface mooring, as well as their approximate depth below the water surface.

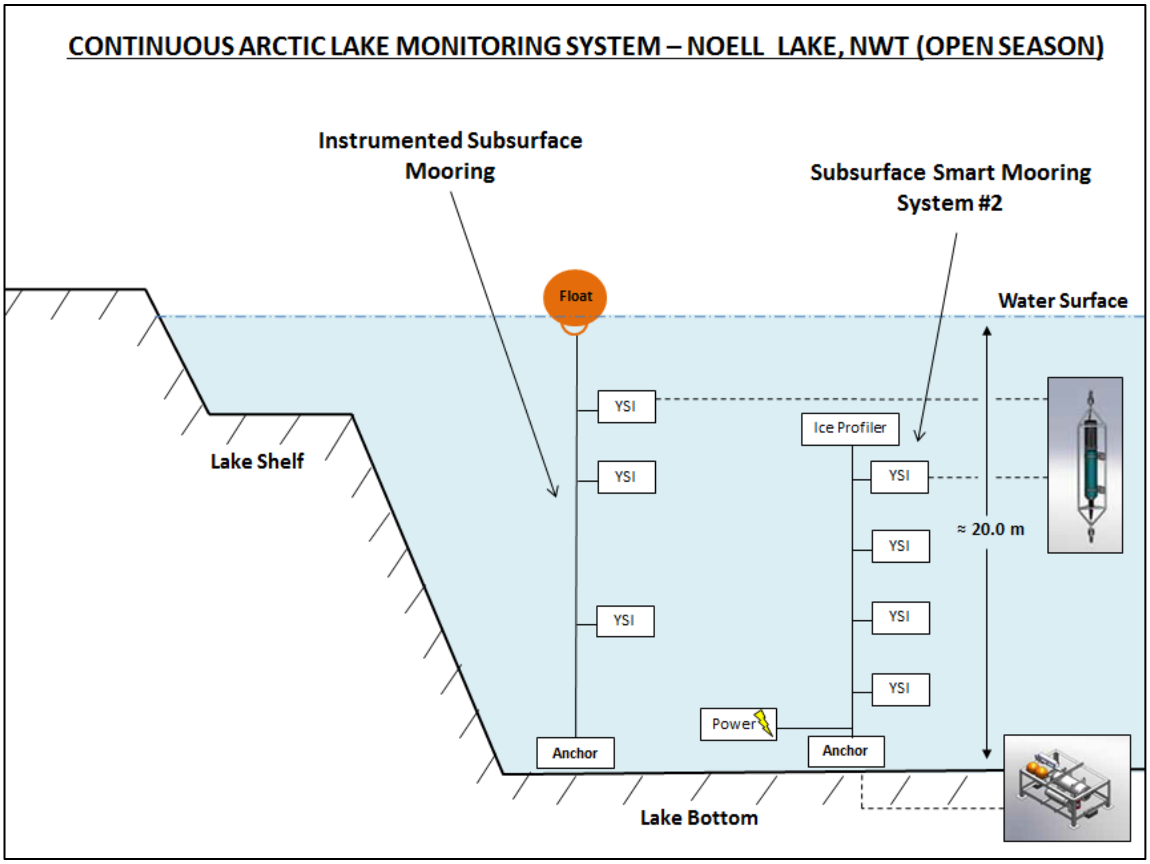

Figure 3-2: Schematic representation of the supplementary subsurface mooring (right) and instrumented subsurface mooring (left) deployed in Noell Lake, NWT. YSI: Yellow Springs Instruments, water quality sonde.

Table 3-3: *In-situ* instrumented components of the supplementary subsurface mooring deployed in Noell Lake in 2011. The parameters measured, as well as the initial deployment depths of the sensors, are provided.

	YSI 1	**YSI 2**	**YSI 3**	**YSI 4**
Instrument Depth	**4.5 m** below water surface	**7.25 m** below water surface	**9 m** below water surface	**11.25 m** below water surface
Variables	Water temperature., specific conductivity, conductivity, resistance, total dissolved solids, salinity, pressure, pH, depth, oxidative reduction potential, turbidity, chlorophyll *a*, dissolved oxygen saturation, dissolved oxygen, blue green algae, battery voltage			

Table 3-4: *In-situ* instrumented components of the instrumented subsurface mooring deployed in Noell Lake in 2012. The parameters measured, as well as the initial deployment depths of the sensors, are provided.

	YSI Top	**YSI Mid**	**YSI Bottom**
Instrument Depth	**3 m** below water surface	**4.5 m** below water surface	**10 m** below water surface
Variables	Water temp. (^{o}C), specific conductivity (µS/cm), depth (m), pH (units of pH), turbidity (NTU), dissolved oxygen saturation, dissolved oxygen (mg/L), battery voltage (mV)	Water temp. (^{o}C), specific conductivity (µS/cm), conductivity, depth (m), pH (units of pH), turbidity (NTU), dissolved oxygen saturation, dissolved oxygen (mg/L), battery voltage (mV)	Water temp. (^{o}C), specific conductivity (µS/cm), conductivity (µS/cm), depth (m), pH (units of pH), oxidation reduction potential (mV), turbidity (NTU), chlorophyll *a* (µg/L), dissolved oxygen saturation, dissolved oxygen (mg/L), blue green algae (cells/mL)

3.2.3 Spatial Homogeneity

An important element of evaluating the overall suitability of the ALMS components is assessing whether the point-measurements recorded by the *in-situ* sensors mounted on the components are representative of the entire lake. Water quality samples were collected from the site of the various buoy and mooring components, as well as six additional locations around Noell Lake, and analysed for a suite of chemical and nutrient parameters. The six additional sample locations were selected to correspond with locations used by Laurent de Rham, in partnership with the Aurora Research Institute, as part of the Noell Lake Spring 2012 Lake Ice Thickness Sampling Program (de Rham, 2012) (Table 3-5). The seventh sampling location was chosen due to its proximity to the ALMS buoy, subsurface mooring, and instrumented subsurface mooring components. Mann-Whitney Rank Sum Tests were performed to determine if the parameters of the water samples collected at the Near-Buoy location were significantly different from the parameters of the water samples collected elsewhere around Noell Lake. Near-Buoy water samples were collected at the yellow buoy symbol indicated on the map; green diamonds indicate sample locations used to compare to the buoy samples; and red diamonds indicate ice survey locations with no corresponding water sampling (Figure 3-3).

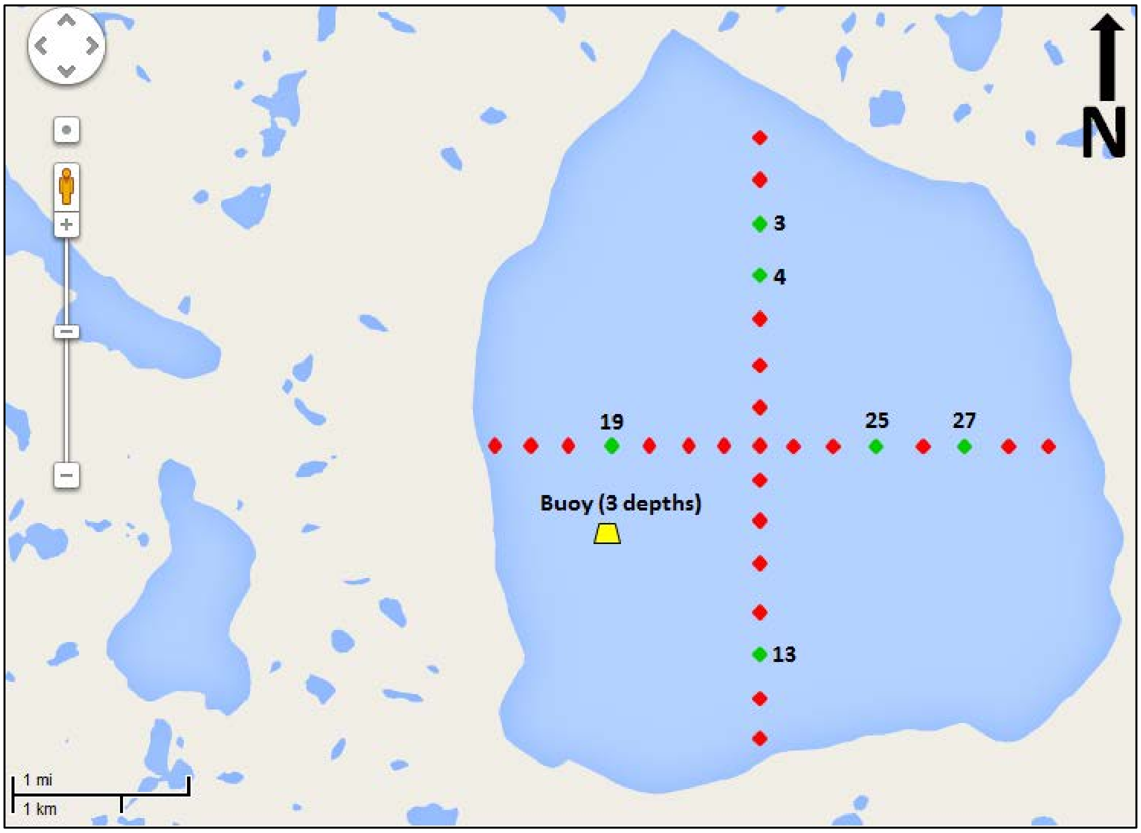

Figure 3-3: Schematic representation of the ice survey locations (red diamonds) used as water sample locations (green diamonds) in Noell Lake. Additional water samples were taken at a site near the ALMS Buoy (yellow trapezoid).

Table 3-5: GPS coordinates of sample-locations and -depths across Noell Lake, NWT. Secchi depth measurements were recorded on July 19th, 2012 between 2:00 - 3:30 pm. Sample and secchi depths are measured in metres.

Sample Location	Latitude	Longitude	Sample Depth	Secchi Depth
Site 3	68° 32’ 760’’ N	133° 34’ 050’’ W	3	4.3
Site 4	68° 32’ 559’’ N	133° 34’ 050’’ W	4.5	4.5
Site 13	68° 30’ 745’’ N	133° 34’ 050’’ W	4.5	3.5
Site 19	68° 31’ 750’’ N	133° 36’ 016’’ W	3	--
Site 25	68° 31’ 750’’ N	133° 32’ 413’’ W	3	4.7
Site 27	68° 31’ 750’’ N	133° 31’ 383’’ W	4.5	4.25
Near-Buoy	68° 31’ 498’’ N	133° 36’ 440’’ W	3	2.5
			4.5	
			8	

Noell Lake was accessed via helicopter transport, and sampling was conducted by boat. For sample locations 3, 4, 13, 19, 25, and 27, a Van Dorn Sampler was used to sample water at one of two depths (3 m and 4.5 m below the water surface); water samples from the buoy site were taken at three depths (3 m, 4.5 m, and 8 m). These sampling depths correspond to the depths of YSI sensors mounted to the ALMS buoy and subsurface mooring system. All water samples were immediately placed in coolers containing ice-packs upon retrieval. Samples were then filtered at the Aurora Research Institute in Inuvik, NWT and prepared for transport to analytical laboratories capable of performing the necessary analyses within 24 hours of removal from Noell Lake.

Nutrient analyses were performed at the National Laboratory for Environmental Testing (NLET) in Saskatoon, SK and included dissolved phosphorous (DP), ammonia (NH_3), nitrate-nitrite (NO_3NO_2), orthophosphate (OP), particulate organic carbon (POC), particular organic nitrogen (PON), total nitrogen (TN), total phosphorous (TP), total dissolved nitrogen (TDN), turbidity, and colour. Chemical analyses were performed at the NLET in Burlington, ON and included specific conductivity (Spec. Cond.), pH, alkalinity (Alk.), major anions (F, Cl, SO_4), colour, dissolved organic carbon (DOC), dissolved inorganic carbon (DIC), hardness (Hard.), major cations (Ca, Mg, Na, K), nitrogen dioxide (NO_2), silica (SiO_2), and turbidity. Standard protocols were followed in both laboratories, as delineated by Environment Canada (1992). The abbreviations used above correspond to labels that will be used in subsequent tables and figures, as well as in Appendices K-O.

Five different sampling campaigns were undertaken for chemical analysis (September 17th, 2011; May 13th, 2012; June 26th, 2012; November 19th, 2012; July 15th, 2013) and 4 for nutrient analysis (September 17th, 2011; May 13th, 2012; November 19th, 2012; July 15th, 2013). Despite several attempts to normalize the chemical and nutrient data through various transformations (e.g.., log, ln, $\log^2$, etc…), parameters continually failed the Normality Test (Shapiro-Wilk, $P < 0.05$) except for pH. As a result, non-parametric methods were used. Mann-Whitney Rank Sum tests were performed on each parameter to test for statistically significant differences between Near-Buoy and whole-lake samples.

3.2.4 Measurement Validation

To assess the validity of the measurements recorded by the various YSI sensors of the buoy and mooring components, results of chemical and nutrient NLET analyses were compared against the recorded parameter values of the various YSI sondes and tested for statistically significant differences. The parameter values recorded by the YSI sondes were tested against values derived through standardized, certified analytical techniques. Furthermore, the parameters under comparison must have been measured at the same time and date in Noell Lake to exclude the effects of changing lake conditions.

First, an investigation was performed to determine overlaps between the date water samples were collected from Noell Lake and sent to various NLET locations for analyses, and the dates any one of the various YSI sensors were recording analogous data. Due to inconsistent YSI functionality, only two lake-sampling dates corresponded with periods YSI sondes were recording comparable data: (1) September 18th, 2001 and (2) November 19th, 2012. The limnological parameters that were measured by NLET chemical analyses on water samples collected the same day YSI sondes were recording comparable data include pH and specific conductivity; parameters analyzed by NLET nutrient analyses on water samples collected the same day YSI sondes recording comparable data include turbidity; parameters analyzed by both NLET chemistry and nutrient analyses on water samples collected the same day YSI sondes were recording comparable data include turbidity.

Despite attempts at normalizing data through various transformations (e.g., log, $\log^2$), all data continued to fail Shapiro-Wilk tests of normality (P-Value = 0.05). As a result, Pairwise Multiple Comparison Procedures (Dunn’s Method) were used to test if the YSI-recorded pH, turbidity, and specific conductivity values differed significantly from the values determined through NLET analytical methods.

3.2.4.1 Air Temperature

Ambient air temperature measured by the temperature sensor (MP101A - Rotronic Instrument Corp.) mounted to the meteorological station of the ALMS buoy was plotted against ambient air temperature recorded by the Environment Canada Weather Station located in Inuvik, NWT to investigate the effectiveness and accuracy of the buoy sensor at recording ambient air temperature. Corresponding measurements of ambient air temperature were measured by both the Inuvik Weather Station and the temperature-sensor on the ALMS buoy during the period September 28th, 2010 - July 2nd, 2012. It is important to note that while the Inuvik Weather Station is located approximately 15 km to the northeast of where the ALMS buoy was deployed in Noell Lake, it represents the closest certified weather station to which buoy meteorological measurements can be compared.

3.2.4.2 Water Temperature

Water temperatures recorded by the YSI sondes at 3 metre- and 9 metre-depths were compared with water temperatures recorded by the HOBO data loggers, over the same time period, and at the same depths. The HOBO loggers were attached to the rope of the Instrumented Subsurface Mooring apparatus. Water temperatures were plotted against time, and divided between open-water and under-ice seasons, as well as between the epilimnion and hypolimnion.

3.3 Results

3.3.1 Functionality

3.3.1.1 Arctic Lake Monitoring System (ALMS) Buoy and Subsurface Mooring System (AXYS Technologies, Inc.)

The ALMS buoy and subsurface mooring system were operational from September 28th, 2010 until February 7th 2011 with a number of sensors and instruments failing to record data consistently. Table 3-6 displays the activity of the various *in-situ* sensors by month. The meteorological and radiative components of the buoy, as well as the ice-profiler of the subsurface mooring system, experienced the greatest degree of success and remained active for the majority of the deployment period. With the exception of the YSI-Deep mounted to the buoy and to a lesser extent YSI-3 of the subsurface mooring, the YSI multiparameter water quality sonde assemblies did not record data for the majority of the deployment period.

Table 3-6: Summary of instrument activity on the ALMS buoy and subsurface mooring system. Red denotes inactivity and no recorded data for the entire month. Green represents active instruments with recorded data for the entire month. Blue denotes partial instrument activity with data only being recorded for a portion of the month. Yellow indicates months in which the instruments were not available to record data.

	Buoy				**Subsurface Smart Mooring**				
Month-Year	**Meteor.**	**Radiat.**	**YSI Shallow**	**YSI Deep**	**YSI 1**	**YSI 2**	**YSI 3**	**YSI 4**	**Ice Profiler**
Sep-10	a	a	X	a	X	X	a	X	a
Oct-10	A	A	X	A	X	A	A	X	A
Nov-10	A	A	X	A	X	A	A	X	A
Dec-10	A	A	X	A	X	A	A	X	A
Jan-11	A	A	X	A	X	X	A	X	A
Feb-11	A	A	X	A	X	X	a	X	A
Mar-11	A	A	X	A	X	X	X	X	A
Apr-11	A	A	X	a	X	X	X	X	A
Jun-11	A	A	X	X	X	X	X	X	A
Jul-11	A	A	X	X	X	X	X	X	--
Aug-11	A	A	X	X	X	X	X	X	--
Sep-11	A	A	X	a	a	a	a	a	A
Oct-11	A	A	X	A	A	A	A	A	A
Nov-11	A	A	X	A	A	A	A	A	A
Dec-11	A	A	X	A	A	A	A	A	A
Jan-12	A	A	X	A	A	A	A	A	A
Feb-12	a	a	X	a	a	a	a	a	a
Mar-12	X	X	X	X	X	X	X	X	X
Apr-12	X	X	X	X	X	X	X	X	X
May-12	X	X	X	X	X	X	X	X	X
Jun-12	X	X	X	X	X	X	X	X	X

3.3.1.2 Supplemental Subsurface Mooring (AXYS Technologies, Inc.) and Instrumented Subsurface Mooring

The supplementary subsurface mooring lost all function in May of 2013 and was retrieved on July 5th, 2013. Table 3-7 displays the activity of the various sensors installed on both the instrumented subsurface mooring and the secondary subsurface mooring by month.

Table 3-7: Summary of instrument activity on the instrumented subsurface mooring and supplemental subsurface mooring components. Red denotes inactivity and no recorded data for the entire month. Green represents active instruments with recorded data for the entire month. Blue denotes partial instrument activity with data only being recorded for a portion of the month. Yellow indicates months in which instruments were not available to record data.

	Instrumented Subsurface Mooring			**Supplemental Subsurface Mooring**			
Month-Year	**YSI Top**	**YSI Mid**	**YSI Bottom**	**YSI 1**	**YSI 2**	**YSI 3**	**YSI 4**
Jul-12	a	a	a	--	--	--	--
Aug-12	A	A	A	--	--	--	--
Sep-12	A	A	A	--	--	--	--
Oct-12	A	A	A	--	--	--	--
Nov-12	A	A	A	a	a	a	a
Dec-12	A	A	X	A	A	A	A
Jan-13	A	A	X	A	A	A	A
Feb-13	A	A	X	A	X	A	A
Mar-13	A	A	X	A	X	A	a
Apr-13	A	A	X	A	X	a	X
May-13	A	A	X	a	X	X	X
Jun-13	A	A	X	X	X	X	X
Jul-13	a	a	X	X	X	X	X

3.3.2 Spatial Homogeneity

The mean, maximum, minimum, and standard deviations of all sampling locations and sampling campaigns undertaken in Noell Lake are included in Table 3-8 and Table 3-9. Raw data are included in Appendix K-O.

Table 3-8: Descriptive statistics of chemical parameters for Noell Lake, NWT. Number of samples (N), mean, maximum values (Max.), minimum values (Min.), and standard deviation (S.D.). Values are in mg/L with the exception of specific conductivity (Spec. Cond.) (µS/cm), colour (Pt-Co), and turbidity (NTU).

Parameter	N	Mean	Max.	Min.	S.D.
Spec. Cond. (µS/cm)	43	76.2	101	59.7	8.2
pH	43	7.3	7.5	7.1	0.1
Alk-$CaCO_3$	43	18.3	24.8	12.9	2.2
F	43	0.07	0.09	0.05	0.01
Cl	43	1.9	2.9	1.4	0.2
SO_4	43	12.1	16.5	8.8	1.4
Colour (Pt-Co)	43	6.9	40.7	0.2	5.7
DOC	27	6.2	14.0	5.3	1.6
DIC	34	4.6	6.7	3.0	0.8
Hrd-$CaCO_3$	43	28.0	36.6	24.0	2.7
Ca	43	7.5	9.7	6.3	0.7
Mg	43	2.3	3.0	2.0	0.2
Na	43	2.6	3.4	1.6	0.3
K	43	1.1	1.3	0.8	0.08
NO_2	22	0.002	0.004	0.001	0.0008
SiO_2	43	0.4	0.8	0.01	0.2
Turbidity (NTU)	9	3.4	26.7	0.3	8.8

Table 3-9: Descriptive statistics of nutrient parameters for Noell Lake, NWT. Number of samples (N), mean, maximum values (Max.), minimum values (Min.), and standard deviation (S.D.). Values are in mg/L, with the exception of turbidity in (NTU) and colour (Pt-Co).

Parameter	N	Mean	Max.	Min.	S.D.
DP	34	0.002	0.005	0.002	0.0007
NH_3	34	0.008	0.03	0.002	0.006
NO_3NO_3	34	0.009	0.07	0.001	0.01
OP	34	0.0004	0.002	0.0001	0.0004
POC	27	0.4	6.5	0.02	1.2
PON	27	0.05	0.8	0.005	0.1
TN	34	0.3	0.7	0.2	0.09
TP	34	0.008	0.06	0.004	0.009
TDN	34	0.2	0.4	0.2	0.03
Turbidity (NTU)	34	1.03	11.0	0.09	1.9
Colour (Pt-Co)	16	4.6	8.0	0.2	4.0

Parameter means of data derived from the analysis of water samples near the buoy location did not differ significantly from parameter means calculated from the analysis of whole-lake data. Table 3-10 and Table 3-11 contain U-Statistic and P-value information from the Whitney Rank Sum test performed on each parameter of the chemical and nutrient analyses.

Table 3-10: U-Statistics and associated P-values of Whitney Rank Sum tests performed on chemical parameters. Bolded P-values indicate significant at $P < 0.05$.

Parameter	**Spec. Cond.**	**pH**	**Alk.**	**F**	**Cl**	**SO_4**	**Colour**	**DOC**	**DIC**
U-Statistic, *P-value*	195.0, *1.000*	252.0, *0.375*	241.5, *0.244*	156.0, *0.271*	285.0, *0.989*	182.5, *0.749*	148.0, *0.218*	70.0, *0.587*	104.5, *0.566*
Parameter	**Hard.**	**Ca**	**Mg**	**Na**	**K**	**NO_2**	**SiO_2**	**Turb.**	
U-Statistic, *P-value*	296.0, *0.801*	182.5, *0.751*	194.5, *1.000*	169.5, *0.508*	183.0, *0.759*	65.5, *0.414*	162.0, *0.389*	4.5, *0.262*	

Table 3-11: U-Statistics and associated P-values of Whitney Rank Sum tests performed on nutrient parameters. Bolded P-values indicate significant at $P < 0.05$.

Parameter	**DP**	**NH_3**	**NO_3NO_2**	**OP**	**POC**	**PON**
U-Statistic, *P-value*	176.5, *0.970*	100.5, *0.464*	114.0, *0.816*	114.0, *0.831*	71.0, *0.625*	69.5, *0.569*
Parameter	**TN**	**TP**	**TDN**	**Turbidity**	**Colour**	
U-Statistic, *P-value*	108.0, *0.663*	179.5, *0.880*	91.0, *0.281*	105.0, *0.580*	18.0, *0.439*	

With respect to the chemical and nutrient parameters measured, there is evidence suggesting parameter spatial-homogeneity in Noell Lake. As a result, it is likely that measurements recorded by instruments installed on the buoy and mooring components at depth are spatially representative of Noell Lake.

3.3.3 Measurement Validation

Results from pairwise multiple comparison tests (Dunn's Method) of pH (Table 3-12 and Table 3-15), turbidity (Table 3-13 and Table 3-16), and specific conductivity (Table 3-14) are provided.

Table 3-12: Pairwise Multiple Comparison Test (Dunn's Method) results for pH. Water samples were taken from 9 locations on September 18th, 2011. Daily averages were calculated from buoy and mooring YSI data recorded on the same day. Data source, N (number of observations), median, Q-Statistic, and P-Values are displayed. P-values < 0.05 indicate a statistically significant difference from NLET analyses.

Data Source	**N**	**Median**	**Q-Statistic**	***P-Value* < 0.05**
NLET Chemical Analysis	9	7.5	--	--
Buoy - Deep YSI	23	4.2	2.75	No
Mooring - YSI 1	21	7.3	1.01	No
Mooring - YSI 2	21	7.8	3.88	**Yes**
Mooring - YSI 3	22	7.7	3.08	**Yes**
Mooring - YSI 4	22	7.5	1.20	No

Table 3-13: Pairwise Multiple Comparison Test (Dunn's Method) results for Turbidity. Water samples destined for NLET analyses were taken from 9 locations on September 18th, 2011; daily averages were calculated from buoy and mooring YSI data recorded on the same day. Data Source, N (number of observations), Median, Q-Statistic, and P-Value are displayed. P-values less than 0.05 indicate a statistically significant difference from NLET analyses.

			Compared to NLET Chemical Analysis		**Compared to NLET Nutrient Analysis**	
Data Source	**N**	**Median**	**Q-Statistic**	**P-Value < 0.05**	**Q-Statistic**	**P-Value < 0.05**
NLET Chemical Analysis	**9**	**0.3**	--	--	0.43	No
NLET Nutrient Analysis	**9**	**0.5**	0.43	No	--	--
Buoy - Deep YSI	23	11.4	6.11	**Yes**	6.63	**Yes**
Mooring - YSI 1	21	11.4	5.27	**Yes**	5.79	**Yes**
Mooring - YSI 2	21	11.2	0.59	No	1.11	No
Mooring - YSI 3	22	11.3	2.21	No	2.73	No
Mooring - YSI 4	22	11.4	3.72	**Yes**	4.24	**Yes**

Table 3-14: Pairwise Multiple Comparison Test (Dunn's Method) results for Specific Conductivity. Water samples were taken from 7 locations on November 19th, 2012; daily averages were calculated from instrumented subsurface mooring YSI data recorded on the same day. Data source, N (number of observations), median, Q-Statistic, and P-Values are displayed. P-values less than 0.05 indicate a statistically significant difference from NLET analyses.

Data Source	**N**	**Median**	**Q-Statistic**	***P-Value* < 0.05**
NLET Chemical Analysis	7	78.0	--	--
Mooring - YSI Top	24	78.0	0.38	No
Mooring - YSI Middle	24	77.0	0.65	No
Mooring – YSI Bottom	24	76.0	4.12	**Yes**

Table 3-15: Pairwise Multiple Comparison Test (Dunn's Method) results for pH. Water samples were taken from 7 locations on November 19th, 2012; daily averages were calculated from instrumented subsurface mooring YSI data recorded on the same day. Data source, N (number of observations), median, Q-Statistic, and P-Values are displayed. P-values less than 0.05 indicate a statistically significant difference from NLET analyses.

Data Source	N	Median	Q-Statistic	*P-Value* < 0.05
NLET Chemical Analysis	7	7.2	--	--
Mooring - YSI Top	24	7.3	4.01	**Yes**
Mooring - YSI Middle	24	7.3	1.57	No
Mooring – YSI Bottom	24	6.8	4.68	**Yes**

Table 3-16: Pairwise Multiple Comparison Test (Dunn's Method) results for Turbidity. Water samples were taken from 7 locations on November 19th, 2012; daily averages were calculated from instrumented subsurface mooring YSI data recorded on the same day. Data source, N (number of observations), median, Q-Statistic, and P-Values are displayed. P-values less than 0.05 indicate a statistically significant difference from NLET analyses.

Data Source	N	Median	Q-Statistic	*P-Value* < 0.05
NLET Nutrient Analysis	7	0.4	--	--
Mooring - YSI Top	24	-0.4	1.71	No
Mooring - YSI Middle	24	-0.5	3.87	**Yes**
Mooring – YSI Bottom	24	0.8	1.57	No

Overall, there is substantial variability in the accuracy with which the YSI sondes recorded limnological data. It is important to note that a multitude of parameters recorded by the YSI do not have corresponding NLET measurements with which to compare. Only pH, turbidity, and specific conductivity were tested; conductivity, dissolved oxygen, percent saturation of dissolved oxygen, chlorophyll-*a*, oxidation reduction potential, blue green algae, and total dissolved solids did not have corresponding NLET measurements to which comparisons can be made.

3.3.3.1 Air Temperature

There is strong agreement between the ambient air temperatures measured by the Inuvik Weather Station and the temperature-sensor on the ALMS buoy during the period September 28th, 2010 - July 2nd, 2012. The sensor on the buoy appears to record larger absolute values of air temperature, which could be explained by local climatic differences between the locations of the two weather stations such as the moderating effect of lake evaporation at the buoy location. Average ambient air temperatures were below-freezing for approximately 8 months of the year, with the warmer season being confined to June – October. Both sensors captured temperatures increasing above 0°C on June 3rd, 2011 and returning to below 0°C on September 24th, 2011. With the end of the darker winter season, increased sunlight contributed to temperatures steadily increasing from mid-March 2011 until August 2011. Decreasing daylight hours caused a subsequent and persistent decline in ambient air temperatures from August 2011 onwards. Furthermore, the lowest temperature recorded by each sensor occurred on the same day, January 29th, 2012, while the highest temperatures were similarly recorded simultaneously on August 5th, 2011. Considerable variation in temperature was reflected by both temperature sensors throughout the entire time period.

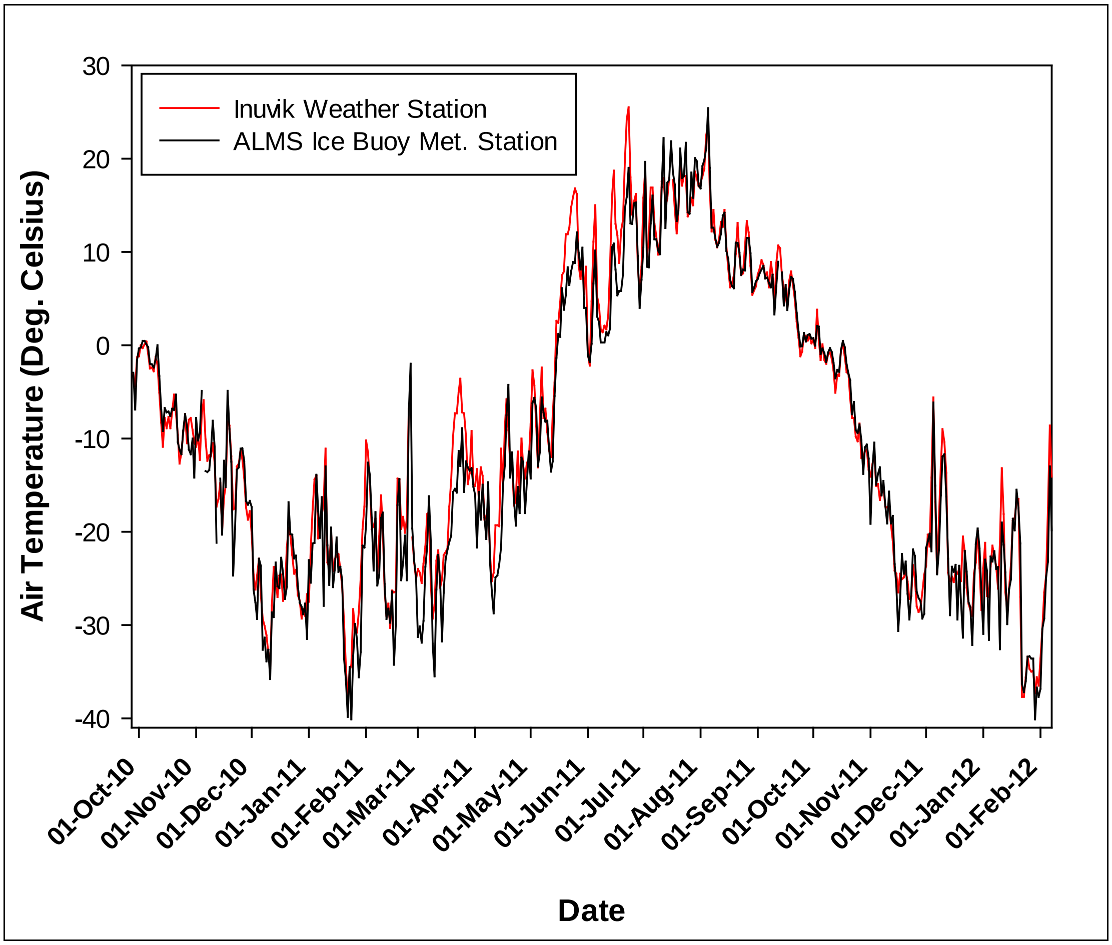

Figure 3-4: Air temperature (°C) as measured by the Environment Canada Weather Station in Inuvik (red), NWT and the temperature sensor mounted to the ALMS buoy (black).

3.3.3.2 Water Temperature

Figure 3-5 and Figure 3-6 displays water temperatures recorded by sensors on YSI sondes and HOBO data loggers, in both the hypolimnion and epilimnion, during open-water and under-ice seasons. Epilimnion and hypolimnion water temperatures were continuously recorded during the period July 16th – October 7th, 2012 constituting the open-water period of Noell Lake. During the under-ice season, epilimnion water temperatures were continuously recorded over the period October 8th, 2012 – July 2nd, 2013 and hypolimnion water temperatures were continuously recorded over the period October 8th, 2012 – April 23rd, 2013.

Open-Water:

There is strong agreement between water temperatures recorded by the YSI and HOBO devices throughout the open-water season at both depths (Figure 3-5). The epilimnion water temperatures, in particular, are nearly identical throughout the 2012 open-water season. Small divergences between recorded hypolimnion water temperatures occur at the start of the open-water season; these dissimilarities can potentially be explained by small differences in sensor depth or instrument location within Noell Lake. Refer to Chapter 4 for detailed and comprehensive explanations of limnological trends including water temperature.

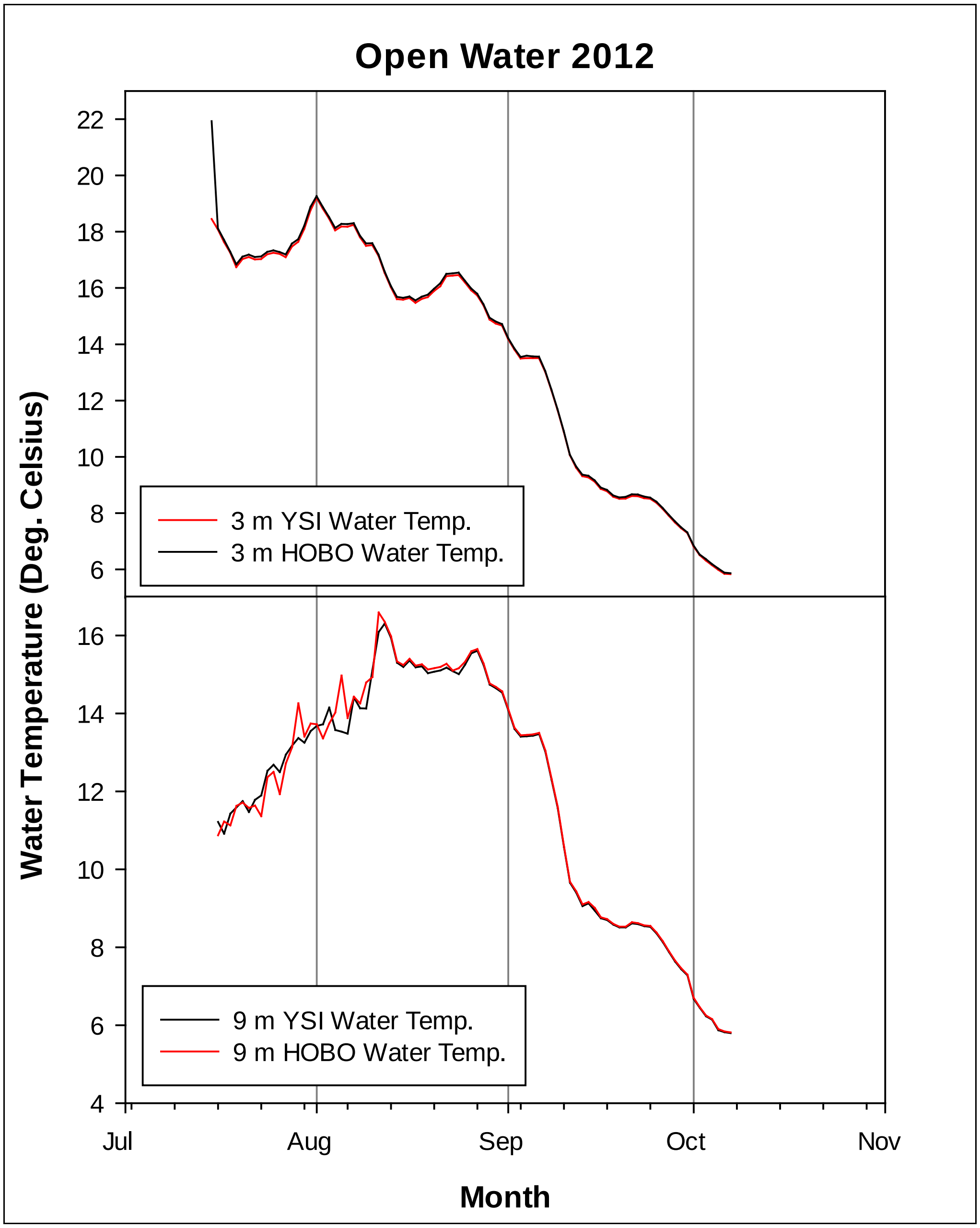

Figure 3-5: Time series of epilimnion (top) and hypolimnion (bottom) water temperatures of Noell Lake, NWT as measured by YSI- and HOBO-sensors during the open-water period July 16th – October 7th, 2012

Under-Ice:

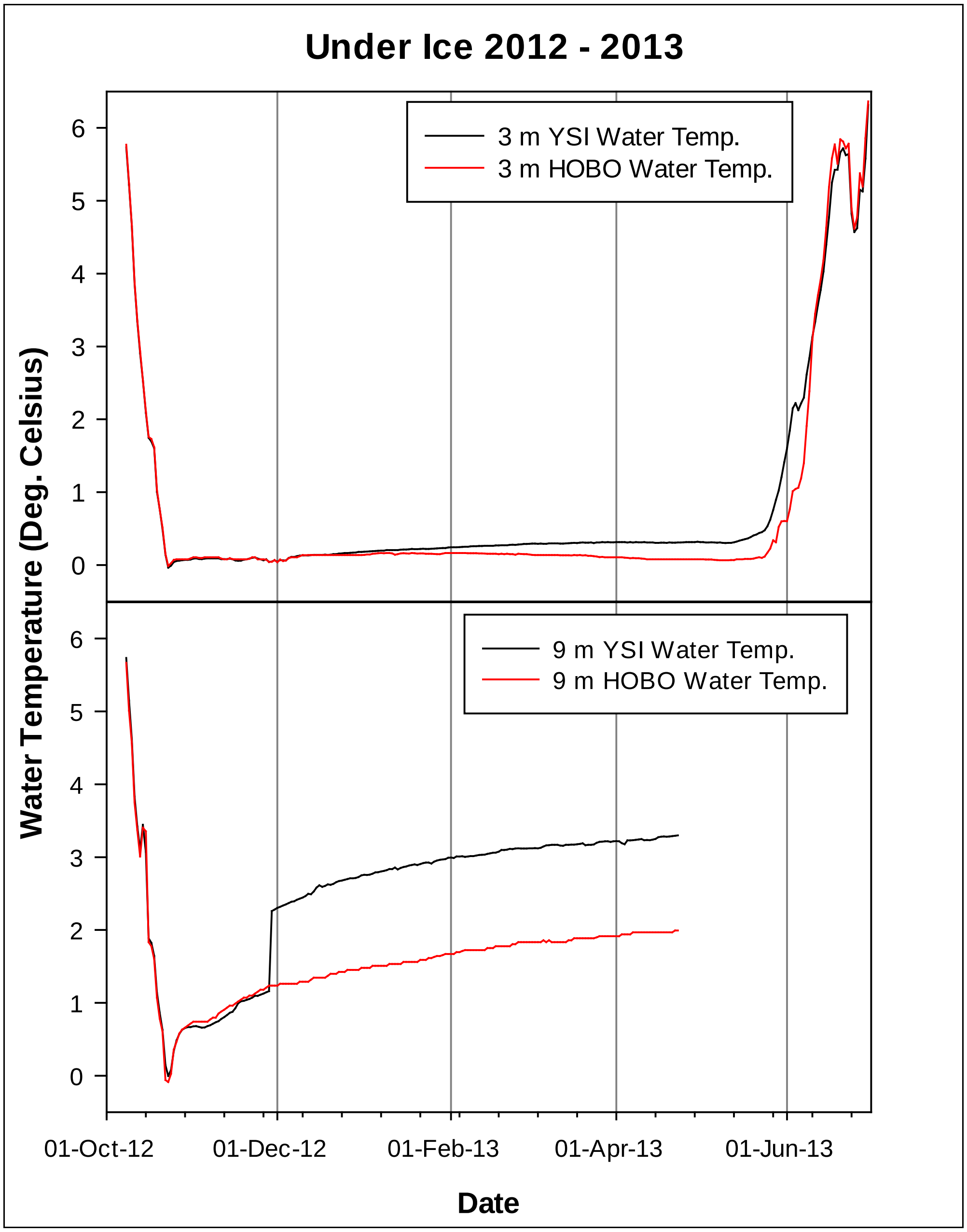

Figure 3-6: (Top) Epilimnion water temperatures (^{o}C) of Noell Lake, NWT as measured by a YSI- and HOBO-sensor plotted against time over the under-ice period October 8th, 2012 –July 2nd, 2013; (Bottom) hypolimnion water temperatures (^{o}C) of Noell Lake, NWT as measured by a YSI- and HOBO-sensor plotted against time over the under-ice period October 8th, 2012 – April 23rd, 2013.

There is agreement between YSI- and HOBO-recorded water temperatures at the beginning of both under-ice seasons at both depths; however, a slight divergence occurs in the epilimnion at the end of January but converges with hypolimnion temperature by

June (Figure 3-6). There is a substantial divergence between recorded water temperatures in the hypolimnion beginning at the end of November 2012. Despite great efforts to ensure that both sensors would remain at 9 metre depths, it is likely that the position of the YSI sensor shifted to a deeper depth than that of the HOBO sensor. As a result, the water in which the YSI was recording water temperature was deeper and, therefore, warmer since it is closer to the temperature of maximum density (3.98^{o}C). Refer to Chapter 4 for detailed and comprehensive explanations of limnological trends including under-ice water temperature.

3.4 Discussion

In evaluating the overall efficacy and utility of the various instrument systems to record uninterrupted meaningful physical, chemical, and biological parameters in the epilimnion and hypolimnion of an Arctic lake over consecutive open-water and under-ice seasons, three main criteria were considered: (1) were the sensors responsible for recording hydro-ecological data functional for a significant length of time? (functionality); (2) were the measurements being recorded by the sensors representative of the entire lake? (spatial homogeneity); and (3) were the values recorded by the sensors representative of the true parameter-values of the lake? (measurement validity).

3.4.1 Functionality

Overall, the majority of the instruments installed on the original ALMS buoy and subsurface mooring components did not adequately record continuous hydro-ecological data for a significant period of time. Notable exceptions include the meteorological and radiative instruments, as well as the ASL Ice Profiler, which recorded continuous data for an extended period of time, ranging from September 2010 until February 2012. In contrast, while YSI-Deep of the buoy only lost functionality for a brief period from mid-April 2011 until mid-September 2011, YSI-Shallow of the buoy was never functional. Of the YSIs mounted to the initial subsurface mooring, YSI-1 and YSI-4 were only functional from mid-September 2011 to mid-February 2012, YSI-2 endured two separate periods of functionality encompassing October 2010 through December 2010 as well as mid-September 2011 through mid-February 2010, and YSI-3 also endured two separate periods of functionality encompassing mid-September 2010 through mid-February 2011 and mid-September 2011 through mid-February 2012.

The YSI sondes installed on the supplemental subsurface mooring, did not significantly improve upon the functionality of the initial subsurface mooring: YSI-1 fared best, and was active from mid-November 2012 through mid-May 2013; YSI-2 was active from mid-November 2012 through January 2013; YSI-3 was active from mid-November 2012 through mid-April 2013; and YSI-4 was active from mid-November 2012 through mid-March 2013.

The YSI sondes installed on the instrumented subsurface mooring component, which was designed at the University of Victoria and assembled in Inuvik, NWT, was the most successful at continuously recording hydro-ecological variables in consecutive under-ice and open-water seasons. With the exception of the YSI-Bottom, the YSI assemblies attached to the instrumented subsurface mooring were active for a greater portion of the instrumented period than the assemblies of the supplementary subsurface mooring. The Top- and Middle-YSI assemblies of the instrumented subsurface mooring were actively recording data for the entire duration of the instrumented period while the Bottom-YSI assembly stopped recording data in November, 2012. Despite being deployed and launched simultaneously on November 22nd, 2012, the four YSI assemblies experienced differential success, with lengths of activity ranging from just over 2 months of recorded data (e.g., YSI-2) to approximately 6 months (e.g., YSI-1).

It is evident that the majority of YSI sondes were unable to maintain functionality throughout consecutive seasons. The primary cause of sensor failure was power failure. The under-ice seasons were particularly draining on the batteries of the YSI sondes, as evidenced by the majority of sensors losing function during the winter seasons. Challenges associated with lake-ice formation as well as maintaining battery-charge were the likely causes of power loss. Installing subsequent YSI sondes at analogous depths below the water surface on all components facilitated the amalgamation of hydro-ecological measurements necessary to generate a substantial, continuous time-series. As a result, despite the functionality of individual YSI sondes being problematic overall, the general temporal range of recorded hydro-ecological parameters encompassed in the investigation was successful; a multitude of physical and limnological parameters were continuously recorded during consecutive under-ice and open-water seasons, in both the epilimnion and hypolimnion.

It is evident that because of the inability of the majority of the sensors to remain active for their intended deployment periods, continuous and uninterrupted analyses of the hydro-ecology of Noell Lake was only possible through the consolidation of recorded data from different instrument platforms. Fortunately, the YSI assemblies of the various components were intentionally mounted at analogous depths below the water surface to facilitate comparison, integration, and synthesis amongst other recorded data.

3.4.1.1 Multi-Depth Time Series Development for Specific Environmental and Hydrological Parameters

To generate a continuous time series of the hydro-ecological measurements recorded by the multitude of instruments installed on the various buoy and mooring components over the three year period in Noell Lake, data from different YSI assemblies but comparable depths were amalgamated. Amalgamation facilitated the production of a more temporally comprehensive dataset. For example, the YSI-Deep of the ALMS buoy components and the YSI-Top of the instrumented subsurface mooring were both installed at 3 metres below the water surface; therefore, data recorded by the ALMS buoy's YSI-Deep - spanning from late September, 2010 until mid-May 2011 and from mid-September, 2011 until early February, 2012 – were combined with the data recorded by the YSI-Top of the instrumented subsurface mooring - which recorded data from mid-July, 2012 until mid-July, 2013. This ultimately generated a relatively uninterrupted collection of environmental parameters in the epilimnion of Noell Lake beginning in September, 2010 and ending in July of 2013.

Furthermore, the YSI-3 of both the initial subsurface mooring and supplementary subsurface mooring are both located at approximately 9 metres below the water surface, while YSI-Bottom from the instrumented subsurface mooring was located at an approximate depth of 10 metres. Accordingly, recorded YSI assembly data from these three components were integrated to produce a temporally continuous set of hydro-ecological parameters in the 9-10 metre depth range of Noell Lake; a range encompassing the hypolimnion of Noell Lake. More importantly, the development of simultaneous and temporally-continuous three-year records for a multitude of hydro-ecological parameters, in both the epilimnion and hypolimnion, is possible.

For the purposes of plotting parameter information, the calendar-year was divided into two seasons: (1) the Under-Ice season and (2) the Open-Water season. Although there are no existing data on historical ice-period timing for Noell Lake, it was selected as a location used to validate the MyLake model (Soloranta and Anderson, 2007) as part of an on-going investigation into lake-ice regimes under changing climatic conditions (de Rham, 2009). As a result, the average under-ice and open-water seasons can be estimated by the MyLake model based on the calculated average freeze up day (October 8th) and the average break up day (June 30th) in Noell Lake. Consequently, the under-ice period spans October 8th to June 30th while, inversely, the open-water period extends from July 1st to October 7th.

3.4.2 Spatial Homogeneity

Water quality parameter values quantified from water samples collected at the buoy and mooring apparatus locations did not differ significantly from the parameter values of water samples collected at the six additional sites around Noell Lake (p-value < 0.05). Of the 17 parameters quantified through NLET chemical analyses and 11 parameters quantified through NLET nutrient analyses, no parameter values determined from water samples collected near the buoy and mooring components were significantly different from the parameter values of water samples collected around the rest of Noell Lake. As a result, parameter values recorded by instruments installed on the buoy and mooring components were spatially representative of Noell Lake.

3.4.3 Measurement Validation

There was strong agreement between air temperatures and water temperatures recorded by the buoy and mooring components, and the HOBO data-loggers and weather stations used for corroboration. The validation of YSI-recorded values of pH, turbidity, and specific conductivity, however, did not produce overwhelmingly robust agreement with parameter values determined through NLET analyses. Importantly, it was not possible to test all parameters recorded by YSI sondes; dissolved oxygen, percent saturation of dissolved oxygen, chlorophyll-*a*, oxidation reduction potential, conductivity, and blue green algae had no corresponding analytic values through which comparisons could be made. Lack of agreement between YSI-recorded parameter values and NLET values could be an indictment of the difficulties involved with using YSI sondes to measure and record pH, turbidity, and specific conductivity - especially under-ice. Some of the calculated differences between buoy/mooring parameter values and whole-lake parameter values arise due to obvious sensor malfunctions (i.e., impossibly large negative values of turbidity and pH recorded by YSIs). Although all YSIs were appropriately calibrated prior to deployment in Noell Lake, their ability to accurately measure and record hydro-ecological parameters over consecutive under-ice and open-water seasons requires further investigation and scrutiny. Additionally, comparisons of YSI-recorded values of dissolved oxygen, percent saturation of dissolved oxygen, chlorophyll-*a*, oxidation reduction potential, conductivity, and blue green algae with parameter values determined through analytical techniques are needed.

3.5 Conclusions and Recommendations for Future Research Campaigns

Overall, automated buoy and subsurface mooring platforms were useful in recording and establishing continuous hydro-ecological baseline conditions of an upland tundra lake. The research campaign successfully measured and logged a multitude of physical and limnological measurements, uninterruptedly, over a three year period. Importantly, measurements recorded by sensors installed on the sequence of buoy and mooring components were deemed spatially representative of Noell Lake. The synthesis and analyses of recorded data afforded the integration and establishment of a number of important physical, biological, and chemical conditions of Noell Lake, in both the hypolimnion and epilimnion. As a result, this study was able to address a pervasive lack of continuous limnological data from lake systems in the western Canadian Arctic. Uninterrupted and recurrent sampling regimes are necessary to produce meaningful and instructive long-term datasets.

Recommendations to consider for future research campaigns using these devices include:

1) Few of the YSI sondes installed on the ALMS buoy recorded data consistently, and some of the devices did not record any data at all. The inclusion of YSI sondes on the buoy should be re-considered and potentially eliminated.
2) Of all the parameters recorded by YSI sondes in this research campaign, turbidity and oxidation reduction potential were the least reliable and accurate. The overall usefulness in including these parameters in future sampling campaigns using YSI sondes should be scrutinized.
3) Alternatively, analyzing and comparing time series of epilimnion and hypolimnion water temperature and dissolved oxygen (both percent saturation and concentration) generated valuable information on the mixing characteristics and stratification regime of the lake, the development of hypoxia in the hypolimnion, and the overall suitability of oxy-thermal habitats of Noell Lake. It is imperative that they be recorded in any future sampling campaigns. Given that changes to the persistence and strength of stratification regimes as well as changes to the mixing regimes of lakes are expected with increased climate warming in Arctic regions, it would be useful to gather as much information on these components as possible.
4) The instrumented subsurface mooring component was the most successful and least expensive used during this project. Greater focus on these systems, as opposed to the buoy and surface mooring components, may be beneficial.
5) YSI sondes are susceptible to damage from the formation of lake-ice. Acquiring accurate information pertaining to the ice phenology of the lake under investigation can ensure YSI sondes are installed at depths below the ice layer.
6) It would be useful to corroborate YSI-recorded measurements of dissolved oxygen, percent saturation of dissolved oxygen, chlorophyll-*a*, oxidation reduction potential, conductivity, and blue green algae with parameter values derived analytically from water samples collected at the same time as YSI measurements were recorded.

7) Greater focus must be paid to measuring potential changes in lake-ice duration and thickness. There is a paucity of Arctic lake-ice break-up, freeze-up, and thickness data; any future investigations should make the acquisition of these data a priority.
8) More detailed investigations into the vertical representativeness of buoy and mooring components are recommended.

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CHAPTER 4: CONTINUOUS ENVIRONMENTAL MONITORING OF NOELL LAKE, NORTHWEST TERRITORIES

Abstract

During the past three decades, the western Arctic of North America has sustained some of the most significant warming on earth. Research attempting to investigate climate-driven changes in northern lake ecosystems is complicated by a legacy of relatively instantaneous research initiatives predicated on defining the state of a given lake ecosystem given principally transitory lake-water samples analyzed to reveal traditional and physical parameters indicative of only the days or hours immediately before sampling. Not only are these “snapshot” samples limited in their temporal scope, but they also fail to reflect and integrate the other relevant biological, physical, and chemical conditions of the lake crucial to defining its state. The compounding effect of these constraints has resulted in an absence of meaningful baseline environmental data available for analysis and comparison. An array of instrument platforms and components, sensors, *in-situ* sampling, data-sources, and experiments were used to produce a continuous and comprehensive description of daily changes to the limnology and water quality of Noell Lake, NWT between the period September 30th, 2010 - July 2nd, 2013. Over the study period, Noell Lake displayed a dimictic mixing pattern. The major late summer/early autumn mixing event was detected on August 12th, 2012 followed by the development of persistent winter stratification until the spring mixing event, which occurred on June 10th, 2013. The development of hypoxic conditions occurred as of March 3rd, 2013 as the percent saturation of dissolved oxygen in the hypolimnion decreased below 30%. Augmentary nutrient limitation experiments suggest that autotrophic productivity in Noell Lake was nitrogen-limited.

Keywords: Arctic lake, Comprehensive limnology, Continuous Monitoring, Water Quality

4.1 Introduction

Arctic lakes are sensitive to environmental changes, with research suggesting that climate warming has prompted distinct changes to the surface area, water levels, and ecological components of various northern lakes over the past few decades (e.g., Rühland and Smol, 2005; Smith et al., 2005; Smol et al., 2005; Riordan et al., 2006; Smol and Douglas, 2007; Plug et al., 2008; Labrecque et al., 2009; Vincent, 2009; Williamson et al., 2009). Similarly, paelolimnological investigations have revealed that Arctic lakes have responded to changes in climate over the past several hundred years (Douglas and Smol, 1999; Rühland et al., 2003; Rühland and Smol, 2005; Smol et al., 2005). Given that polar regions will continue to experience the most significant warming (Kattsov et al., 2005; Prowse et al., 2006a), Arctic lakes, in particular, afford a unique ability to reveal climate-induced changes, as well as illuminate how other aquatic ecosystems, both Arctic and Temperate, are likely to respond moving forward (Lim and Douglas, 2003). As a result, it is evident that Arctic lakes represent crucial reference ecosystems to which past, recent, and future global environmental change can be compared (Lim and Douglas, 2003).

The quantity and circumpolar-representation of investigations of Arctic limnology have increased - Finland (e.g., Luoto, 2009); Greenland (e.g., Cremer et al., 2005); Norway (e.g., Løvik and Kjellberg, 2003); Alaska (e.g., LaPerriere et al., 2003); western Canadian Arctic (e.g., Ogbebo et al., 2009a,b); eastern Canadian Arctic (e.g., Westover et al., 2009); Sweden (e.g., Brunberg et al., 2002); Russia (e.g., Moiseenko et al., 2009) – however, ecological investigations of Arctic lakes are overwhelmingly outnumbered by those of Temperate systems (MacDonald et al., 2012). Furthermore, few enduring, continuous biological monitoring programs have been undertaken in lakes of northern latitudes (e.g., Douglas and Smol, 1993, 1994, 1995; Michelutti et al., 2003; ACIA, 2004).

Research campaigns centred on defining the state of a given lake ecosystem using mainly transient observations is common in northern limnology (Bailey et al., 2004). These momentary observations are limited to summer lake-water bulk samples analyzed to reveal limited chemical and physical parameters, indicative of only the days or hours immediately before sampling (Bailey et al., 2004). Not only are these “snapshot” samples limited in their temporal scope, but they fail to consider and integrate other significant physical, chemical, and ecological components of the lake crucial to defining its state.

Continuous and lasting monitoring programs of Arctic freshwater ecosystems are constrained by the high financial and logistical investments necessary in northern research. Uninterrupted, high-frequency sampling is challenging to undertake and sustain in Arctic lakes since they are generally far away from established centres of human activity (MacDonald et al., 2012). Moreover, northern research is technically challenging, subjected to extreme weather conditions, and study lakes are commonly only accessible by snowmobile and/or helicopter, imposing additional constraints on any research initiative (Hille, 2010).

Despite efforts to document the physical, biological, and chemical characteristics of water bodies across the Canadian Arctic increasing recently (i.e. Kokelj et al., 2005; Mesquita, 2008; Thompson, 2009; Kokelj et al., 2009; Hille, 2010), the aforementioned cost and logistical difficulties associated with undertaking northern research have limited these limnological investigations to sampling regimes unable to integrate the necessary temporal, chemical, physical, and biological parameters to accurately define the current state of these systems. Identifying and forecasting the effects of climate change in northern lakes, especially in light of additional anthropogenic and environmental stressors acting concomitantly on these ecosystems, are crucial to the stewardship and management of these aquatic resources. In this study, a range of instrument platforms, sensors, experiments, and arrays were used to generate a comprehensive and integrated suite of continuous baseline conditions, at a multitude of lake-depths, in both open-water and under-ice seasons. The various ecological, geochemical, hydrological, climatological, and physical components of an upland, Arctic tundra lake were sampled.

4.2 Methods

To produce a comprehensive description of the limnology of an upland, Arctic tundra lake, a multitude of ecological, geochemical, hydrological, environmental, and physical data sources were integrated.

a) Continuous, uninterrupted measurements of water temperature, conductivity, specific conductivity, concentration of dissolved oxygen, percent saturation of dissolved oxygen, oxidation reduction potential, total dissolved solids, pH, chlorophyll-*a*, and blue green algae were recorded during the period September 30th, 2010 – July 2nd, 2013. Measurements were recorded during consecutive under-ice and open-water seasons, and at two separate depths: (a) three-metres below the water surface (epilimnion) and (b) nine-metres below the water surface (hypolimnion).

b) Instantaneous depth-profiles of water temperature, conductivity, specific conductivity, concentration of dissolved oxygen, percent saturation of dissolved oxygen, oxidation reduction potential, pH, chlorophyll-*a*, and blue green algae were recorded at one-metre intervals during open-water (August 8th, 2012) and under-ice seasons (May 13th, 2012) at a multitude of locations around Noell Lake. All YSI sondes and sensors were calibrated prior to use in the field.

c) Additional successive chemical and nutrient water quality sampling analyses were undertaken: September 17th, 2011; May 13th, 2012; June 26th, 2012; November 19th, 2012; and July 15th, 2013. Water samples were collected at the same suite of locations that parameter depth-profiles were recorded by YSI sondes, and analyzed for specific conductivity, pH, alkalinity, fluorine, chlorine, sulphate, colour, dissolved organic carbon, dissolved inorganic carbon, hardness, calcium, magnesium, sodium, potassium, nitrogen dioxide, silicon dioxide, turbidity, dissolved phosphorous, ammonia, nitrate+nitrogen dioxide, organophosphate, particulate organic carbon, particulate organic nitrogen, total nitrogen, total phosphorous, and total dissolved nitrogen.

d) Nutrient-limitation experiments were undertaken in the open-water season of 2013 to generate further inferences into which nutrients may be constraining autotrophic and heterotrophic productivity in Noell Lake.

e) HOBO data loggers recorded uninterrupted depth profiles of water temperature, at one-metre intervals, down to a depth of 15 metres below the water surface for a full year.

f) Satellite images of lake-ice freeze-up and break-up were compiled and synthesized to establish sequences of lake-ice formation and decay in Noell Lake.

4.3 Time Series of Average Water Quality Conditions

The water quality of Noell Lake was studied by the Department of Fisheries and Oceans in 1982 (Read and Roberge, 1986) and 1986 (Anema et al., 1990a,b), and by Ogbebo et al. (2009a) in 2007 as part of a greater limnological investigation in the lower Mackenzie River Basin. Water quality analyses include the collection, and shipment of bulk water samples collected from Noell Lake to analytical laboratories for subsequent testing of chemical and nutrient parameters. Similarly, toward generating a more temporally continuous understanding of the limnology Noell Lake, as well as evaluate potential spatial variability in the limnological parameter of Noell Lake, water quality samples were collected at various locations and depths, during different months and years, and analysed for similar suites of chemical and nutrient parameters on five different days: September 17th, 2011; May 13th, 2012; June 26th, 2012; November 19th, 2012; and July 15th, 2013. Greater detail on sampling procedures, sample locations, and sample depths is provided in "Section 3.3.1 Spatial Homogeneity of Lake Measurements" of Chapter 3; raw data are available in the Appendix (Appendix K – Appendix O).

The means of reported chemical and physical water quality data of Noell Lake are included in Table 4-1, while reported nutrient, chlorophyll *a*, and Secchi depth data of Noell Lake are included in Table 4-2. The monthly means from 2011 to 2013 represent the mean of all individual sampling locations and depths from the given sampling day within that month. Work in Noell Lake facilitated a time series of geochemical and nutrient conditions in Noell Lake from August 1982 through July 2013. Raw data, with associated maximums, minimums, and standard deviations are included in Appendix K, L, M, N, and O.

Table 4-1: Inter-annual average concentrations for major ions and related water quality variables for Noell Lake. Average concentrations are in mg/L except for pH in pH units, water temperature in (^{o}C), and specific conductivity in (µs/cm). Raw data, maximums, minimums, and standard deviations are included in Appendix K, L, M, N, and O.

	August Mean 1982[b]	**Summer Mean 1986[c]**	**July 2007[a]**	**Sept. Mean 2011[d]**	**May Mean 2012**	**June Mean 2012**	**Nov. Mean 2012**	**July Mean 2013**
pH	--	7.6	8.2	7.4	7.3	7.2	7.2	7.3
Water Temp.	13.5	10.2	15.7	--	--	--	--	--
Diss. O.	9.0	--	9.8	--	--	--	--	--
Spec. Cond.	--	71.1	67.3	71.2	89.4	71.1	79.0	70.9
Mg	1.2	2.0	--	2.1	2.6	2.2	2.4	2.1
Na	1.9	2.4	2.5	2.4	3.0	2.5	2.8	2.5
K	0.6	1.0	--	1.0	1.2	1.1	1.1	1.1
Cl	2.4	2.1	1.7	1.7	2.2	1.8	2.1	1.8
Ca	4.7	7.0	7.1	7.0	8.7	6.9	7.4	7.3
SO_4	9.5	10.9	11.0	11.4	14.3	11.1	12.5	11.3
SiO_2	--	--	0.4	0.2	0.4	0.4	0.2	0.6
Fe	--	0.04	--	--	--	--	--	--
Mn	--	0.01	--	--	--	--	--	--

SOURCES: [a]Ogbebo et al., 2009a; [b]Read and Roberge, 1986; [c]Anema et al., 1990b

Table 4-2: Inter-annual average concentrations of nutrient, chlorophyll-*a*, and Secchi depth data for Noell Lake. Average concentrations are in µg/L unless designated otherwise. Raw data, maximums, minimums, and standard deviations are included in Appendix K, L, M, N, and O.

	August 1982[b]	**Summer Mean 1986[c]**	**July 2007[a]**	**Sep. Mean 2011**	**May Mean 2012**	**Nov. Mean 2012**	**July Mean 2013**
Secchi Depth (m)	--	--	2.8	--	--	--	--
Total Phosphorous	--	--	7.8	12.5	4.5	4.7	8.6
Diss. Phosphorous	70.0	--	1.8	2.7	2.7	2.0	2.3
Total Nitrogen	--	--	201.5	329.0	259.0	220.0	251.0
Total Diss. Nitrogen	1000.0	--	192.8	237.0	251.0	202.0	212.0
NH_3	--	--	6.3	5.6	9.8	4.6	12.9
$NO_3 + NO_2$	--	--	5.0	1.0	29.0	2.4	1.0
Chlorophyll a	0.3	0.8	1.6	--	--	--	--
Part. Organic Carbon	--	--	103.0	869.0	51.0	--	190.0
DOC (mg/L)	--	--	4.9	6.3	6.4	--	5.7
DIC (mg/L)	--	--	4.3	4.2	5.7	4.3	4.1

SOURCES: [a]Ogbebo et al. 2009a; [b]Read and Roberge 1986; [c]Anema et al. 1990b

4.4 Annual Cycle of the Limnology of Noell Lake

Continuous under-ice and open-water plots for average daily temperature, conductivity, specific conductance, percent saturation of dissolved oxygen saturation, concentration of dissolved oxygen, pH, chlorophyll *a*, oxidation-reduction potential, blue green algae, and total dissolved solids, as recorded by buoy and mooring components, are presented in the Appendices (Appendix A – Appendix J). However, for clarity of explanation, some figures may be incorporated and discussed in-text; other figures may be referred to but only provided in the Appendix. For the purpose of explaining the annual cycle of Noell Lake's hydro-ecology, the period July-2012 to July-2013 will be discussed.

4.4.1 Timing of Seasonal Stratification, Mixing-Regime Characteristics, and Depth-Profiles of Various Limnological Parameters of Noell Lake

Understanding seasonal changes to the vertical distribution of heat, in addition to the stratification- and mixing-regimes characterizing a given lake affords valuable understanding into the processes most strongly regulating the various components of its limnology. It is important to consider that water of similar temperatures, and therefore, densities mixes more readily than water of different densities (Wetzel, 1975). The integration and synthesis of continuous measurements of (1) water temperature, (2) percent saturation of dissolved oxygen, (3) concentration of dissolved oxygen recorded by buoy and mooring components, in addition to (4) water temperatures recorded by HOBO data loggers installed at one-metre intervals facilitates a multitude of unique and important discernments into the limnology of Noell Lake. More specifically, the thermal-depth profile, development of stratification, and the exact dates of mixing events and hypoxic conditions can be inferred with precision for the first time, facilitating detailed establishment of limnological conditions, processes, and thresholds in Noell Lake.

Furthermore, the identification of exact dates of mixing events resulted in the realization that water quality sampling occurred during periods in which Noell Lake was mixed, as well as during summer and winter stratification. This generated insights into instantaneous geochemical conditions of Noell Lake at several important stages in its annual limnological cycle. Additionally, depth profiles of *in-situ* limnological parameters recorded by YSI sondes are included as well.

Despite comparatively smaller transfers of heat from the air, condensation, inflows, and lake sediments - direct absorption of solar radiation by the water is the greatest and most important source of heat to lakes (Wetzel, 1975). The amount of light energy absorbed in a lake is highest at the lake surface, decreases exponentially with depth, and increases with organic matter content (Wetzel, 1975). Conversely, exchanges of thermal radiation and evaporative cooling at the lake surface account for the most significant losses of energy from a lake system (Wetzel, 1975). Only approximately 10% of observed thermal profiles in lakes can be explained by the direct absorption of solar radiation, the remaining redistribution of heat energy is a product of several processes including the mixing action of wind-energy, underwater currents, the morphometry of the lake basin, and losses of water through outflow (Birge, 1916; Wetzel, 1975).

Open-Water:

The temperature profile of Noell Lake is typical of an ice-covered, dimictic lake. With the breakup of ice-cover and return of increased inputs of solar radiation in late May, the lake surface warms faster than mixing forces can redistribute the heat to lower, cooler depths. As a result, warm air temperatures and high inputs of solar radiation (24 hours a day during summer) produce a strong thermal separation in the summer months, with temperature-derived differences in water density separating the warmer surface water from the cooler water at the lake bottom for most of July and half of August. At this point, it is useful to divide Noell Lake into three thermal regions, which become increasingly resistant to mixing with one another: (1) the epilimnion, (2) the thermocline, and (3) the hypolimnion. The thermocline refers to the zone encompassing a steep thermal gradient between the well-mixed and thermally homogeneous upper epilimnion, and comparatively colder and stagnant hypolimnion.

Epilimnion average daily water temperature ranged from a maximum of 19.2^{o}C on August 1st, 2012 to a minimum of 5.83^{o}C on October 7th, 2012 (Figure 4-1). Hypolimnion average daily water temperatures ranged from a maximum of 16.3^{o}C on August 12th, 2002 to a minimum of 5.8^{o}C on October 7th, 2012 (Figure 4-1). Below the mixing action of the wind, the hypolimnion is cooler and comparatively stagnant. This contributes to a fairly well-defined and persistent stratification between the epilimnion and hypolimnion until August 12th, 2012. It is important to note that the hypolimnion derives small inputs of heat from solar radiation, density currents, and turbulent conduction (Hutchinson, 1941).

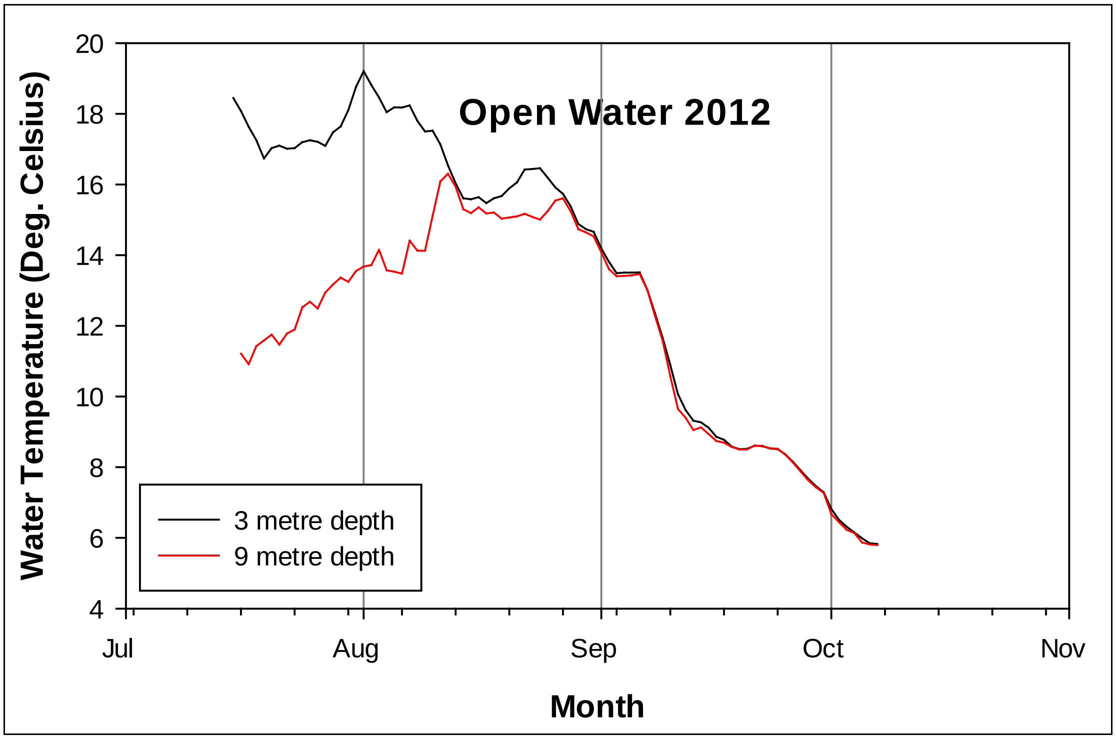


Figure 4-1: Temporal variation in average daily water temperature (°C) throughout the 2012 open-water period in Noell Lake.

In early August, as the ambient air temperature declines at a rate exceeding the offsetting inputs of heat from solar radiation, net losses of heat from the lake facilitates surface cooling. As cooled surface water becomes denser than the underlying warmer water in the epilimnion, it sinks into the thermocline region. Convective currents, in addition to wind-induced mixing, accelerate the sinking process of water from the well-mixed and isothermal epilimnion encroaching into the hypolimnion. As a result of these mixing and sinking processes, the temperature of the hypolimnion eventually converges with that of the epilimnion, thus, eliminating the thermal stratification. Once the lake water is isothermal, even moderate wind-action can cause a mixing of the entire volume of Noell Lake. Such an event took place on August 12th, 2012 – the day of the maximum daily average hypolimnion temperature.

Immediately after the August 12th, 2012 mixing event, Noell Lake appears to undergo another brief period of stratification on August 16th, 2012; evidence of additional stratification is supported by the sudden divergence between epilimnion and hypolimnion water temperatures. Surface warming is the likely cause of a distinct increase in epilimnion temperatures, while hypolimnion temperatures remain consistent. On August 26th, 2012, the epilimnion temperature cools to equal that of the hypolimnion and the stratification is eliminated once again. Similar to the August 12th, 2012 mixing event, evidence of whole-lake mixing can be inferred from the analyses of dissolved oxygen during the same open-water period (Figure 4-2), and the percent saturation of dissolved oxygen (Appendix D).

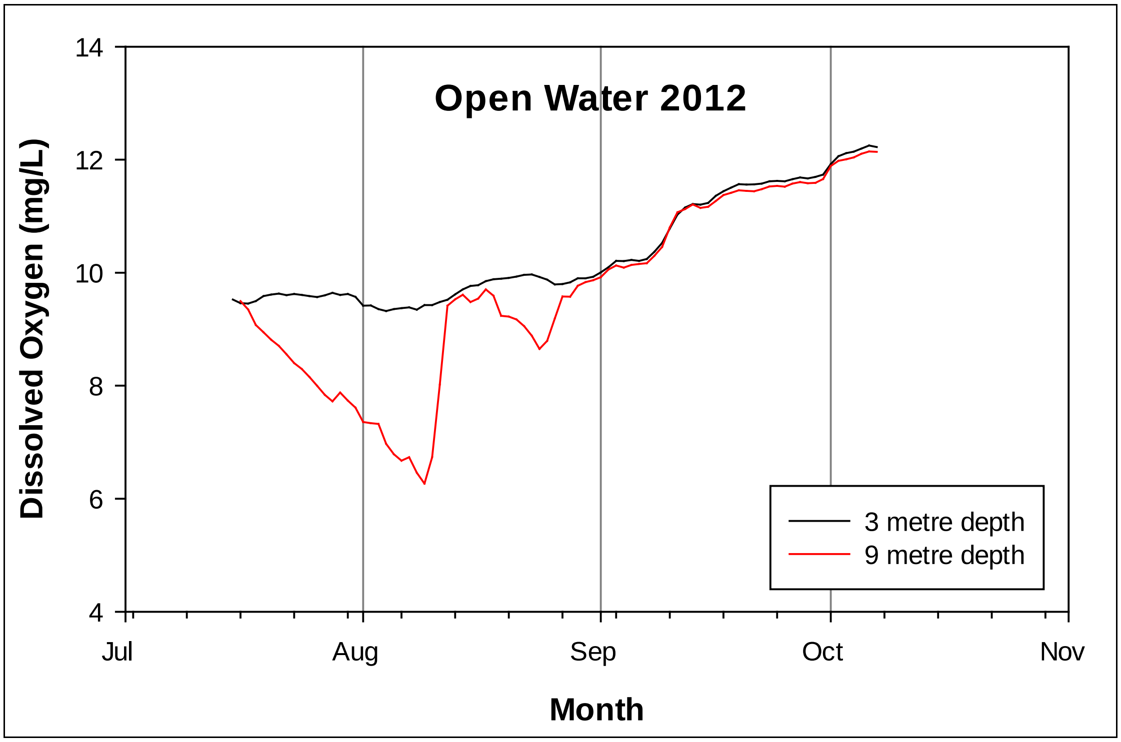


Figure 4-2: Temporal variation in dissolved oxygen (mg/L) throughout the 2012 open-water period in Noell Lake.

Figure 4-3 displays a time series of water temperature, at a multitude of depths, as recorded by the data loggers strapped to the cable of Instrumented Subsurface Mooring. These data facilitate further validation of the mixing and stratification processes suggested through the analyses of buoy and mooring data. The same mixing events and onsets of stratification are observable in the HOBO data logger data.

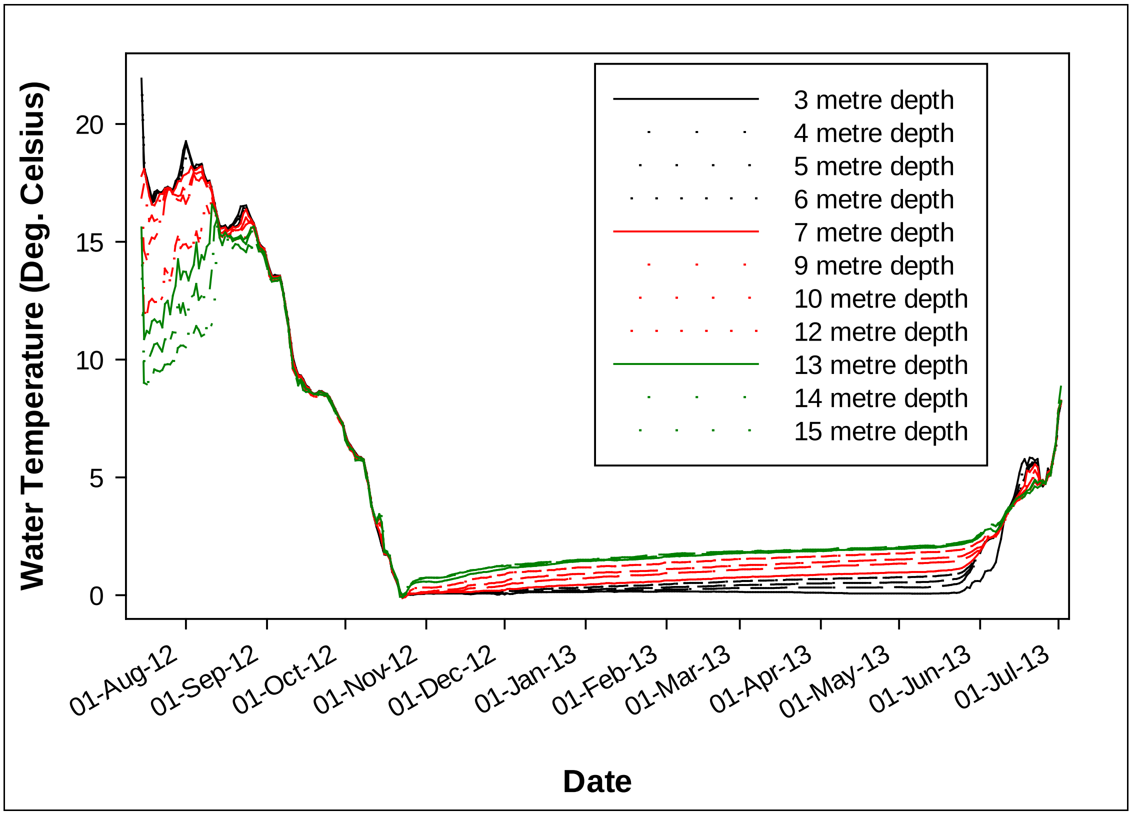


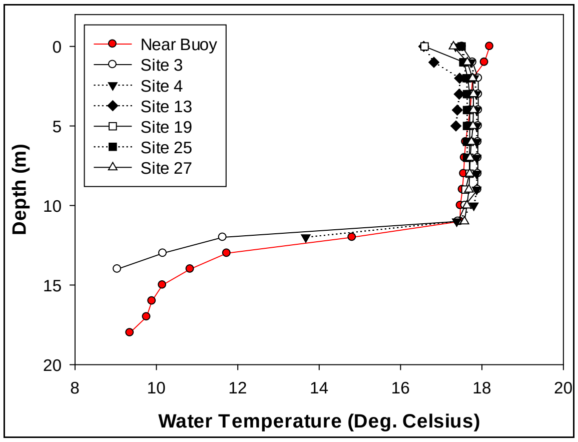
Figure 4-3: Water temperatures recorded by HOBO data loggers strapped at one-metre intervals to the cable of the instrumented subsurface mooring.

The lake remains relatively well-mixed and isothermal for the duration of the 2012 open-water period as the water temperatures of the epilimnion and hypolimnion continue to cool simultaneously. Since the epilimnion and hypolimnion are similar in temperature and density, Noell Lake remains fully mixed for several weeks at the end of the open-water season. This is corroborated through the analyses of buoy and mooring water temperature data (Figure 4-1), dissolved oxygen data (Figure 4-2), percent saturation of dissolved oxygen (Appendix D), as well as water temperature data recorded by HOBO data loggers (Figure 4-3). Full mixing of the lake water column persists from August 26th, 2012 until October 23rd, 2012 when the onset of lake-ice formation begins, and winter stratification begins.

Table 4-3: Geochemical conditions of Noell Lake under autumn mixed conditions (water samples collected on September 17th, 2011). Parameter concentrations are in mg/L with the exception of pH, which is in pH units. Number of observations for all parameters = 9.

Parameter	**Mean**	**Standard Deviation**	**Maximum**	**Minimum**
Specific Conductivity	71.2	4.36	74.00	59.70
pH	7.4	0.087	7.53	7.24
TP	0.01	0.018	0.060	0.0044
TN	0.3	0.16	0.73	0.24
DOC	6.3	2.88	14.00	5.30
DIC	4.2	0.46	4.50	3.00
Ca	7.0	0.27	7.22	6.32
Mg	2.1	0.058	2.19	1.99
Na	2.4	0.30	2.52	1.60
K	1.0	0.073	1.070	0.85
F	0.07	0.0030	0.070	0.060
Cl	1.7	0.13	1.78	1.39
SO_4	11.4	1.0	11.70	8.80

Additionally, on August 8th, 2012, prior to the August 12th, 2012 mixing event, a handheld (YSI 556) YSI Multiparameter Water Quality Sonde Assembly (YSI 6600 V2 - Yellow Springs Instruments, Idaho, US) was used to record at-depth measurements of water temperature, conductivity, specific conductivity, dissolved oxygen, percent saturation of dissolved oxygen, pH, oxidation reduction potential, chlorophyll-*a*, and blue green algae data. Vertical profiles were recorded at seven different locations around Noell Lake**.** Information on the exact locations of the various depth profiles in Noell Lake is included in “Section 3.3.1 Spatial Homogeneity of Lake Measurements” (Figure 3-3).



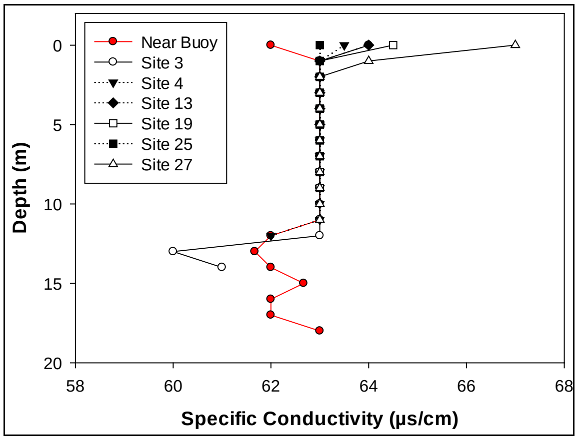
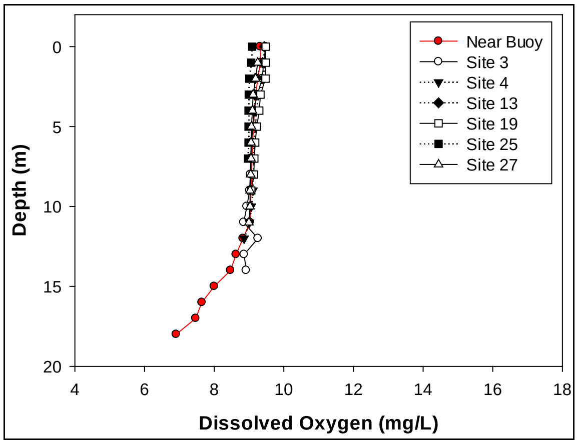


Figure 4-4: Open-water depth profiles of water temperature (°C) in Noell Lake, NWT recorded on August 8th, 2012.

Figure 4-5: Open-water depth profiles of specific conductivity (µs/cm) in Noell Lake, NWT recorded on August 8th, 2012.



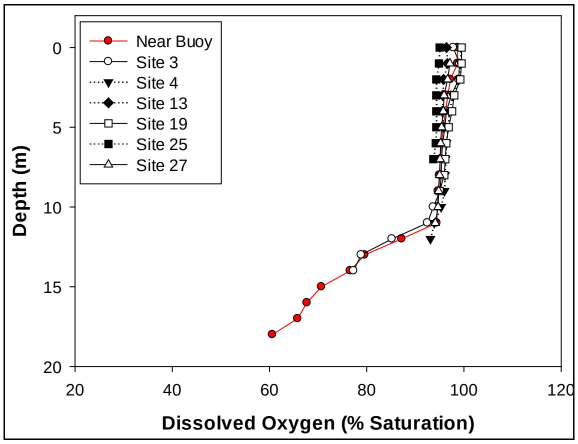
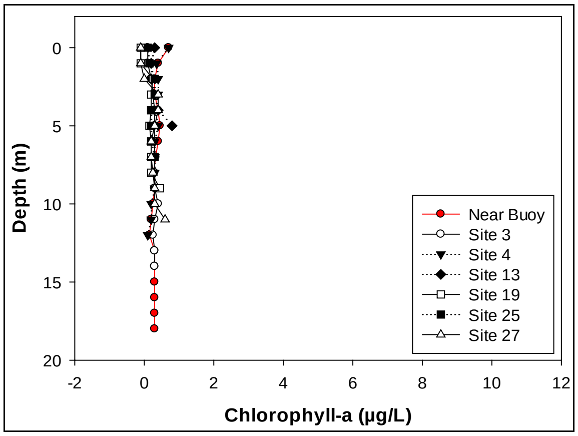


Figure 4-6: Open-water depth profiles of dissolved oxygen (mg/L) in Noell Lake, NWT recorded on August 8th, 2012.

Figure 4-7: Open-water depth profiles of dissolved oxygen (percent saturation) in Noell Lake, NWT recorded on August 8th, 2012.



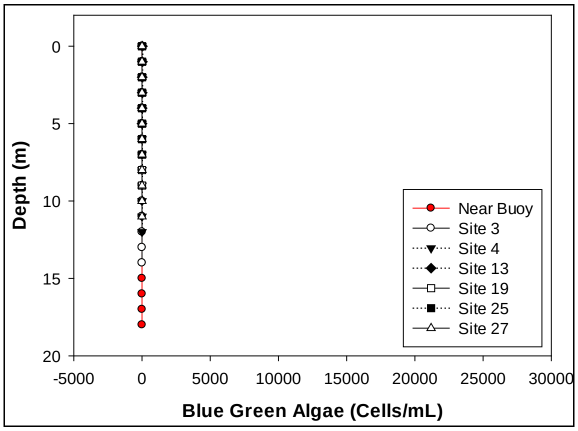


Figure 4-8: Open-water depth profiles of chlorophyll-*a* in Noell Lake, NWT recorded on August 8th, 2012.

Figure 4-9: Open-water depth profiles of blue green algae (cells/mL) in Noell Lake, NWT recorded on August 8th, 2012.

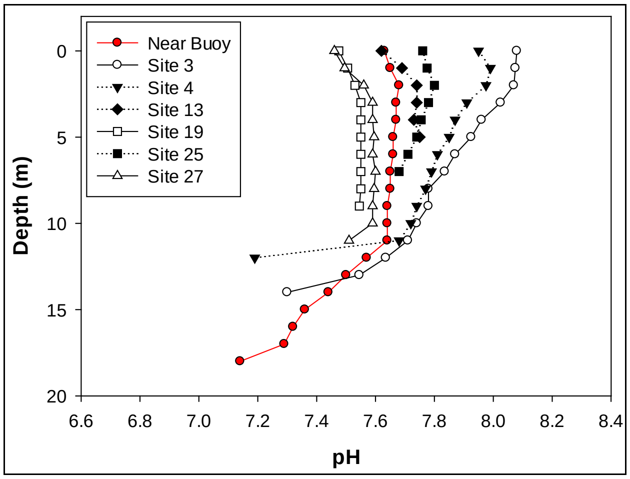


Figure 4-10: Open-water depth profiles of pH in Noell Lake, NWT recorded on August 8th, 2012.

<u>*Under-Ice:*</u>

During the 2012-2013 under-ice season, epilimnion and hypolimnion temperatures cool and reach the TMD in early October. Prior to ice formation, wind-induced mixing of the entire lake occurs in response to slight wind-action since the density gradient between water at 0°C and 4°C is relatively small. As a result, it possible for the isothermic conditions of the epilimnion and hypolimnion to approach 0°C, as is the case during the 2012-2013 under-ice season. Isothermic conditions are relatively common in medium- to large-sized lakes that are exposed to considerable wind action, such as Noell Lake (Wetzel, 1975). Once ice forms and covers Noell Lake, thermal exchanges with the atmosphere are reduced and wind-induced mixing is essentially eliminated. Water immediately below the newly formed ice is 0°C and is, in turn, underlain by water of increasing in density and temperature that approaches the circulation temperature of the lake prior to ice formation (Wetzel, 1975). As a result, winter stratification develops during the under-ice season in Noell Lake (Figure 4-11).

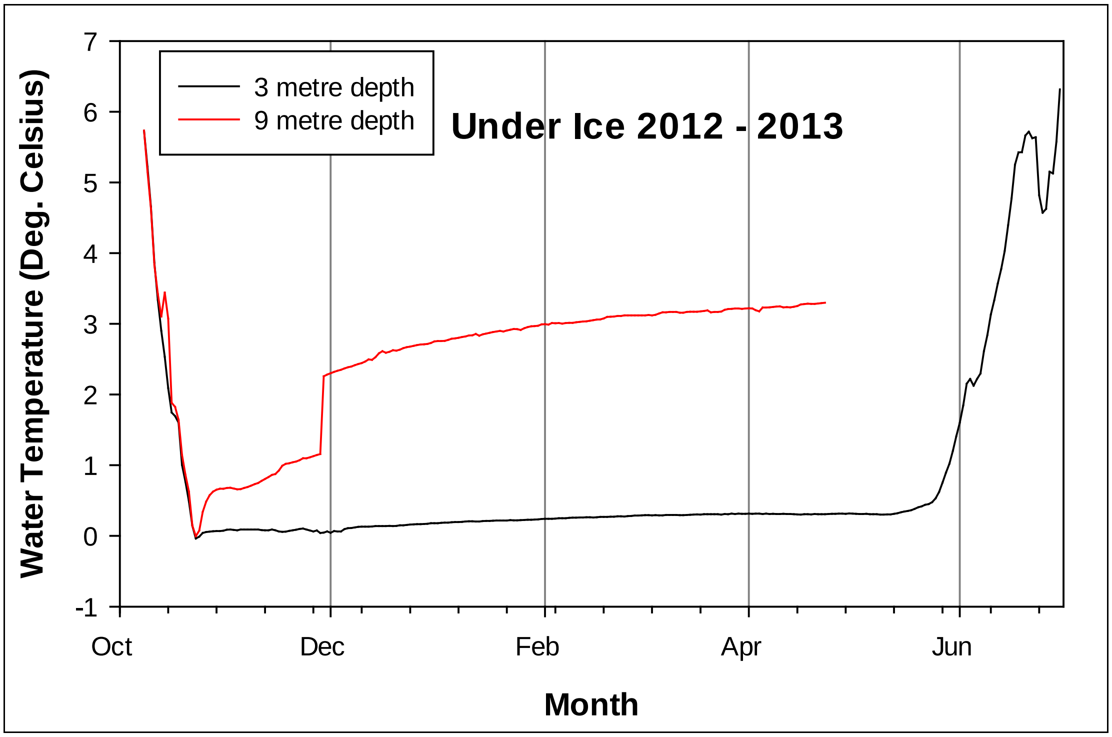


Figure 4-11: Temporal variation in average daily water temperature (°C) throughout the 2012-2013 under-ice period in Noell Lake measured by YSI sondes.

Differences between epilimnion and hypolimnion temperatures in winter are less pronounced than in the open-water season and, as a result, the density differences responsible for winter stratification are also a function of differences in hypolimnion and epilimnion salinity (4.4.2.2 Specific Conductivity). The under-ice water temperature profile of Noell Lake during the 2012-2013 under-ice season is fairly typical for a medium-sized, ice-covered lake. Water temperatures immediately below the ice remain at 0°C for the majority of the under-ice season, and are in turn underlain with water increasing in temperature, salinity, and density. Average hypolimnion temperatures during the three under-ice seasons range from 2.9°C to 3.3°C. The salinity- and temperature-induced stratification of Noell Lake persists for the duration of all three under-ice seasons and is corroborated by the decline of dissolved oxygen during the under-ice seasons (Figure 4-12).

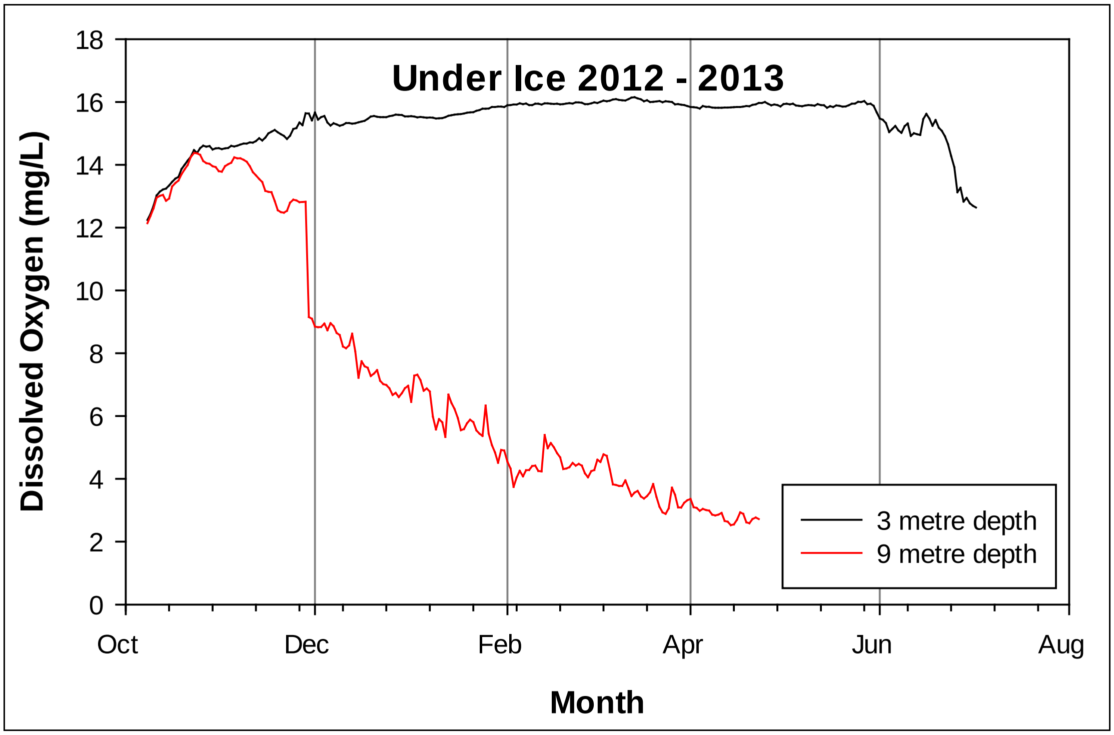


Figure 4-12: Temporal variation in dissolved oxygen (mg/L) throughout the 2012-2013 under-ice period in Noell Lake measured by YSI sondes.

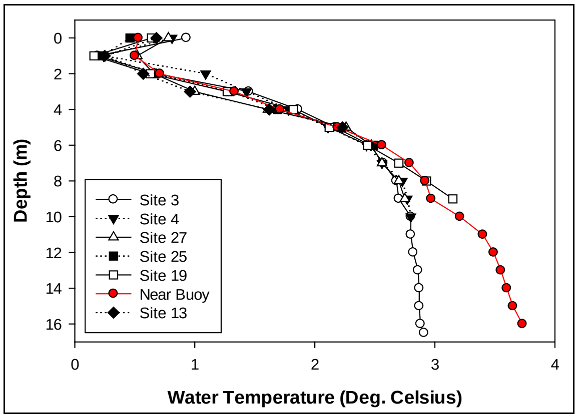
Under-ice heating of the water column continues due to several mechanisms; the relative importance of each mechanism changes over the course of the under-ice season. During the period in which Noell Lake is under-ice but still experiencing daylight, solar radiation can account for upwards of 75% of winter heating, especially if there is little snow cover on the ice (Wetzel, 1975). During the period of persistent darkness, heat is transferred to the water column from lake sediments; the magnitude of the thermal input from lake sediment depends on a multitude of factors including preceding summer thermal conditions and lake morphometry (Hutchinson, 1957). When the main volume of under-ice water is colder than 4°C, water confined to the littoral areas of the lake can be heated slightly through the ice; warmer, denser water will, thus, sink and flow via currents of profile-bound density along the lake sediments to the deeper portion of Noell

Lake (Wetzel, 1975). A sharp increase in epilimnion temperature is observable at the end of the 2012-2013 under-ice season (Figure 4-3). This is most likely related to the break-up of the ice cover and return of radiative warming from the sudden increase of available light energy. Winter stratification persists from October 23rd, 2012 until June 10th, 2013.

Table 4-4: Geochemical conditions of Noell Lake under stratified winter conditions (water samples collected on May 13th, 2012). Concentrations are all in mg/L except for pH. Number of observations for all parameters = 9.

Parameter	Mean	Standard Deviation	Maximum	Minimum
Specific Conductivity	89.4	5.7	101.0	83.1
pH	7.3	0.09	7.4	7.06
TP	0.004	0.0006	0.005	0.004
TN	0.3	0.02	0.3	0.2
DOC	6.4	0.4	7.2	5.9
DIC	5.7	0.6	6.7	5.1
Ca	8.7	0.5	9.7	8.2
Mg	2.6	0.2	3.0	2.5
Na	3.0	0.2	3.4	2.8
K	1.2	0.07	1.3	1.1
F	0.08	0.007	0.09	0.07
Cl	2.2	0.2	2.5	1.9
SO_4	14.3	1.3	16.5	12.4

On May 13th, 2012 a YSI water quality sonde sampling campaign was implemented on the frozen surface of Noell Lake to generate the first under-ice depth profiles of a multitude of limnological parameters. These represent the first instantaneous under-ice depth-profiles of limnological parameters recorded during a period of winter stratification in Noell Lake. Under-ice depth profiles of water temperature, pH, dissolved oxygen, percent saturation of dissolved oxygen, conductivity, specific conductivity, chlorophyll-*a*, blue green algae, and oxidation reduction potential were recorded and are displayed in Figures 13-21.



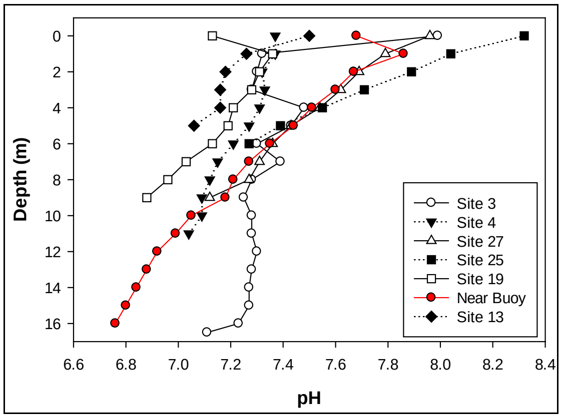
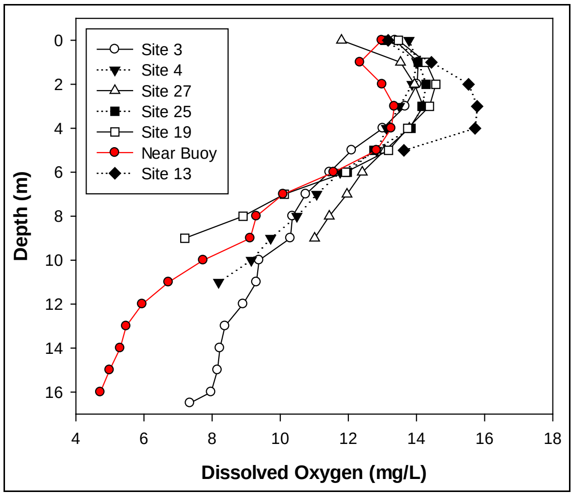


Figure 4-13: Under-ice depth profiles of water temperature (°C) in Noell Lake, NWT recorded on May 13th, 2012.

Figure 4-14: Under-ice depth profiles of pH in Noell Lake, NWT recorded on May 13th, 2012.



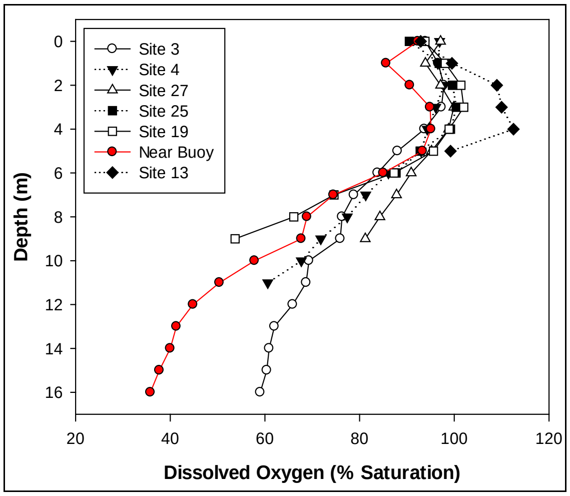
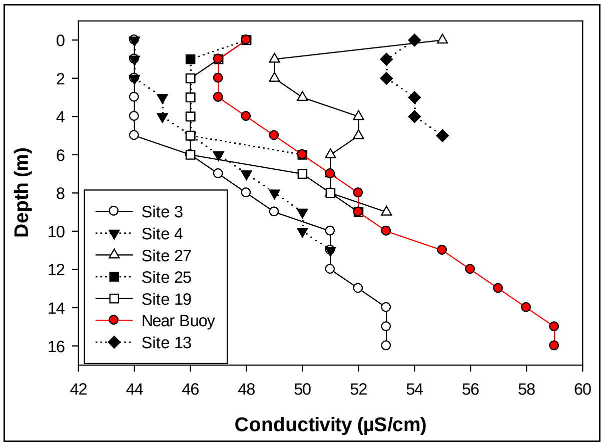


Figure 4-15: Under-ice depth profiles of dissolved oxygen (mg/L) in Noell Lake, NWT recorded on May 13th, 2012.

Figure 4-16: Under-ice depth profiles of dissolved oxygen (percent saturation) in Noell Lake, NWT recorded on May 13th, 2012.



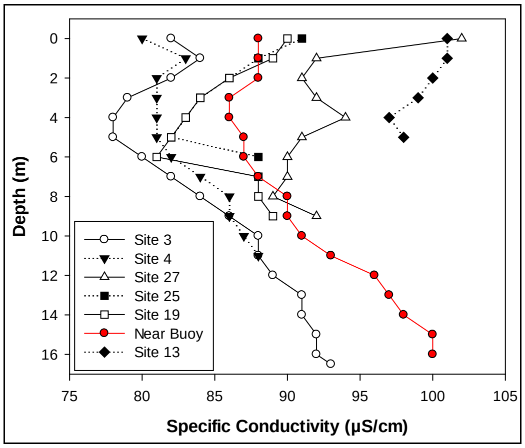
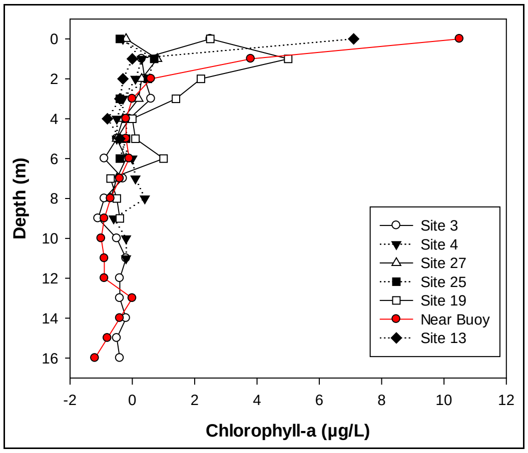


Figure 4-17: Under-ice depth profiles of conductivity (µs/cm) in Noell Lake, NWT recorded on May 13th, 2012.

Figure 4-18: Under-ice depth profiles of specific conductivity (µs/cm) in Noell Lake, NWT recorded on May 13th, 2012.



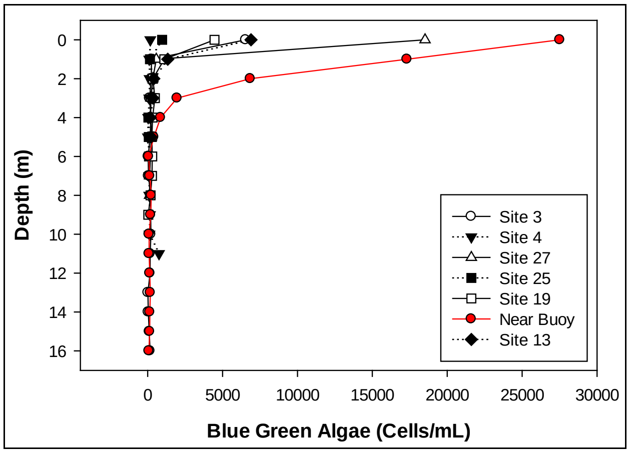


Figure 4-19: Under-ice depth profiles of chlorophyll-*a* in Noell Lake, NWT recorded on May 13th, 2012.

Figure 4-20: Under-ice depth profiles of blue green algae (cells/mL) in Noell Lake, NWT recorded on May 13th, 2012.

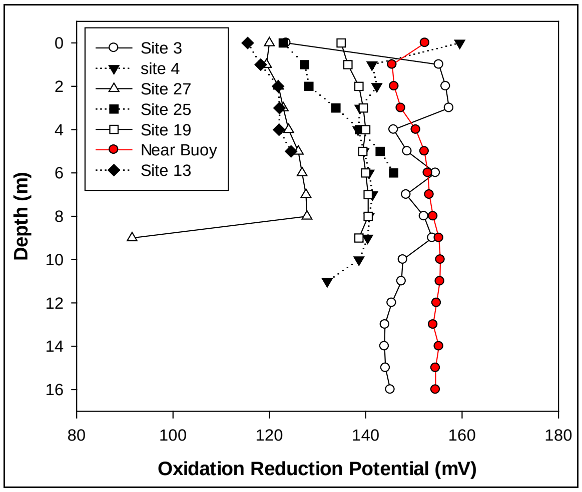


Figure 4-21: Under-ice depth profiles of oxidation reduction potential (mV) in Noell Lake, NWT recorded on May 13th, 2012.

4.4.2 Temporal Trends of Various Limnological Parameters in Noell Lake

In addition to water temperature and dissolved oxygen, a multitude of additional limnological parameters were recorded by the YSI sondes installed on the buoy and mooring components, in both the epilimnion and hypolimnion, and during both open-water and under-ice seasons.

4.4.2.1 Conductivity – Appendix B

The conductivity of lake water is a measure of its capacity to conduct an electrical current, which in turn reflects the ionic salt content of the water. Conductivity is strongly affected by temperature, increasing by approximately 2% per 1°C in warming (Wetzel, 1975).

Open-Water:

Conductivity in the open-water season appears to be regulated by its relationship with temperature. Broadly speaking, conductivity and temperature appear to follow corresponding trends throughout the entire 2012 open-water period (Figure 4-22). Observable increases in conductivity during the open-water season are also likely the consequence of simultaneous dilution effects; lake evaporation concentrates the salinity of lake water. However, changes in open-water conductivity appear to be most dependent on the changes in water temperature. Specific conductivity (section 4.4.2.2 – Specific

Conductivity) excludes the effect of temperature on conductivity by normalizing the salt content to a standard temperature. As a result, it provides a more robust indication of Noell Lake's salinity by isolating changes in the ionic salt content of Noell Lake from simultaneous changes to its water temperature.

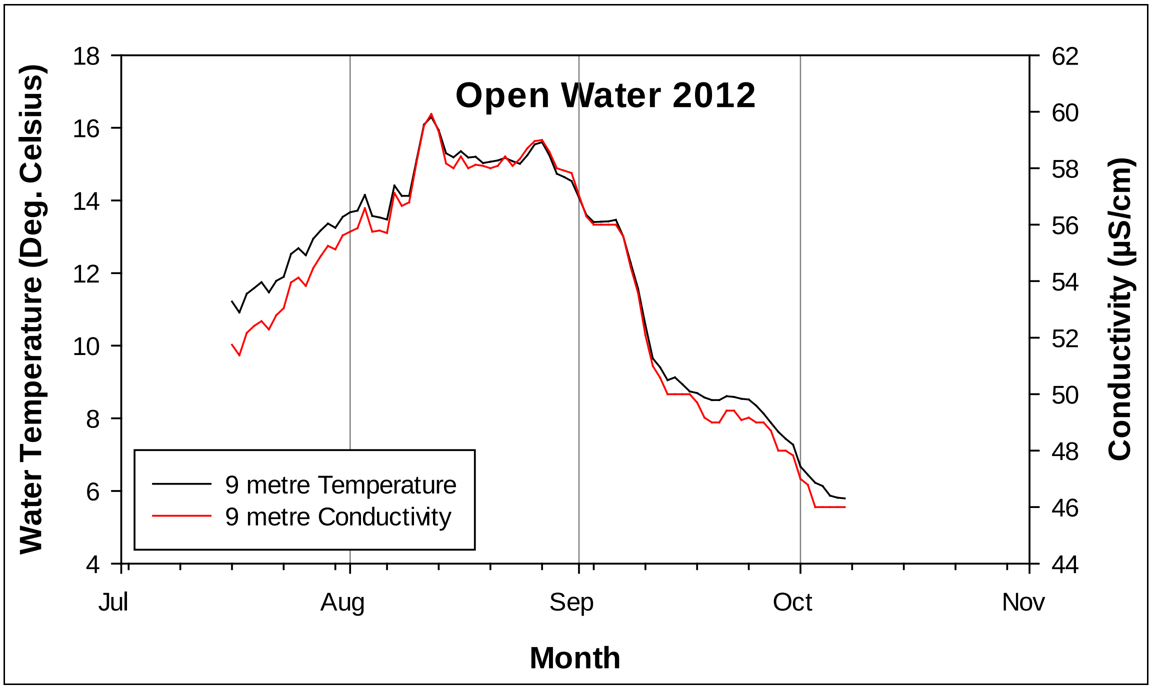


Figure 4-22: Time series of hypolimnion water temperature (°C) and conductivity (µS/cm) of Noell Lake during the open-water period July 16 – October 7, 2012.

Conductivity ranges from a minimum of 46 µS/cm in early October to a maximum of just below 60 µS/cm on August 8th, 2012. As mentioned previously in this section, conductivity and temperature are closely related and there is considerable similarity in the profiles of both limnological parameters in the open-water season of 2012. Furthermore, both parameters undergo their seasonal maximum (August 12th, 2012) and minimum (October 7th, 2012) on the same days of the open-water season.

Under-Ice:

The conductivity of Noell Lake appears to exhibit a similar trend throughout both under-ice seasons, with both the hypolimnion and epilimnion experiencing a series of "step" increases in conductivity, driven most likely by concomitant increases in under-ice water temperature (Figure 4-23). In both under-ice seasons, the conductivity of the hypolimnion exceeds that of the epilimnion. Conductivity in Noell Lake ranges from approximately 40 µS/cm to 57 µS/cm in the hypolimnion, and 40 µS/cm to 53 µS/cm in the epilimnion over the two under-ice seasons. As discussed previously, the temperature of the hypolimnion remains warmer than that of the epilimnion for the majority of the under-ice season; as a result, the conductivity of the hypolimnion exceeds the conductivity of the epilimnion for the entirety of both under-ice seasons. It is unclear what processes may explain the sudden spike in conductivity on December 28th, 2011.

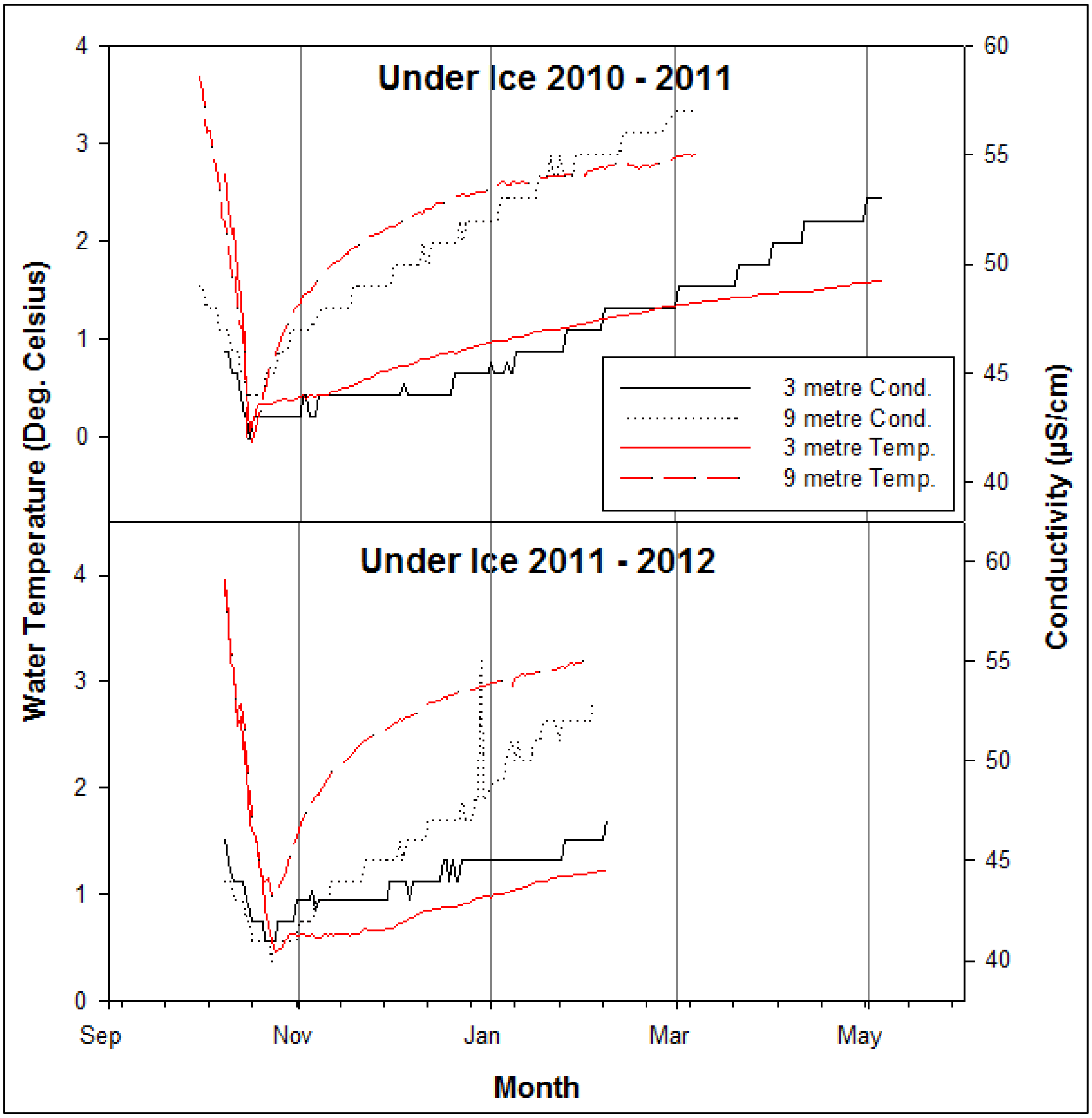


Figure 4-23: Time series of epilimnion and hypolimnion water temperature (°C) and conductivity (µS/cm) of Noell Lake during the under-ice periods (top) October 8, 2010 – May 6, 2011 and (bottom) October 8, 2011 – February 7, 2012.

4.4.2.2 Specific Conductivity – Appendix C

As a result of the conductivity of water being strongly regulated by temperature, attempts to relate and compare measurements of lake water ionic content between seasons and other lakes are problematic. By normalizing conductivity to a fixed temperature (25°C), the actual ionic content of the water can be isolated and valuable comparisons are possible. Measurements of the specific conductivity of a lake quantify the resistance of the water to electrical flow. Generally speaking, as the concentration of ionized salts in water increases, resistance to the flow of electrons decreases. Since specific conductance is defined as the reciprocal of specific resistance, increases in the ionic salt content of water are accompanied by increases in measured specific conductivity.

Open-Water:

The specific conductivity of Noell Lake steadily increases from 70 µS/cm at the beginning of the 2012 open-water season to just above 72 µS/cm by the end, in both the epilimnion and hypolimnion. Interestingly, there is a lag between the points at which the specific conductivity of the epilimnion and hypolimnion begin to increase; the hypolimnion appears to undergo slight step increases in specific conductivity before the epilimnion. Such events begin on July 17th, 2012 and August 3rd, 2012. The slight increases in specific conductivity during the 2012 open-water period are likely due to increases in the concentration of ionic salts due to lake water evaporation at the lake surface.

Under-Ice:

The specific conductivity of Noell Lake during the 2012-2013 under-ice season undergoes considerably larger increases, in both the epilimnion and hypolimnion, than the open-water season. Epilimnion specific conductivity increases from a minimum of 72 µS/cm at the beginning of the under-ice season to a sustained maximum of 93 µS/cm occurring over the period May 2nd to May 26th, 2013. Hypolimnion specific conductivity increases from a minimum of 72 µS/cm at the beginning of the under-ice season to a maximum of 85 µS/cm just prior to instrument failure on April 23rd, 2013. Additionally, unlike the 2012 open-water season, there is a clear divergence between the specific conductivity of the epilimnion and the hypolimnion. Unlike the trends observed in under-ice conductivity, which included hypolimnion conductivity exceeding that of the epilimnion, under-ice hypolimnion specific conductivity is actually *less* than that of the epilimnion. Since the temperature-effects that regulated the dynamics of under-ice conductivity have been standardized to a specific temperature, under-ice increases in the ionic salt concentration of Noell Lake are the likely cause of this pattern.

Considering specific conductivity reflects the level of dissolved salts in water, it would be useful to analyze the under-ice trends in the context of total dissolved solids. However, there was no temporal overlap between YSI sonde recordings of specific conductivity and total dissolved solids. Nevertheless, the analysis of under-ice trends of each parameter from different under-ice seasons can still facilitate valuable extrapolations. As ice forms, total dissolved solids are excluded from the ice mass, contributing to a concentration of total dissolved solids in the comparatively smaller remaining volume of unfrozen water in the epilimnion. Refer to section “4.4.2.9 Total Dissolved Solids” for additional explanations about this exclusion process. Total dissolved solids being excluded from the ice and re-dissolved in the epilimnion of Noell Lake are the likely cause of epilimnion specific conductivity exceeding that of the hypolimnion throughout the 2012-2013 under-ice season.

Density-difference induced stratification is not only a temperature based phenomena, but can also be produced by differences in salinity or combinations of both (Wetzel, 1975). More importantly, the stability of stratifications produced by salinity-driven density differences can be more persistent and difficult to disrupt than stratifications caused by only thermal density differences (Wetzel, 1975). The changes in under-ice specific conductivity are, thus, an important component of the development of the winter stratification.

4.4.2.3 Dissolved Oxygen (Percent Saturation) – Appendix D

Inputs of oxygen in lakes include physical processes such as atmospheric diffusion and wave-mixing, as well as through the photosynthetic processes of algae and other aquatic plants. Conversely, oxygen is consumed by the respiration of fish and other aquatic organisms, as well as aerobic bacteria and microbes through the decomposition of organic matter, especially at the lake bottom (Wetzel, 1975). Certain hypolimnetic chemical reactions, such as the reduction of nitrate to ammonia, can also consume dissolved oxygen (Wetzel, 1975). The rate at which a given lake-ecosystem utilizes available dissolved oxygen in relation to the rate it is being regenerated can, therefore, provide a relatively robust overall indication of the metabolism of the lake (Wetzel, 1975). Furthermore, the distribution of oxygen in Noell Lake affects the solubility of inorganic nutrients, which strongly impact living organisms in the lake. Given that any aquatic organism possessing aerobic respiratory biochemistry relies on dissolved oxygen for their metabolic activity, the dynamics of oxygen circulation in lakes is pivotal to their comportment, dispersal, and growth within the lake (Wetzel, 1975).

The solubility of oxygen in water is affected nonlinearly by temperature, pressure, and salinity (Wetzel, 1975). Broadly speaking, as temperature increases oxygen becomes less soluble in water; conversely, the solubility of oxygen in water increases with pressure. Increasing salinity reduces the solubility of oxygen in water, although freshwater lakes are rarely saline enough to impact the solubility of oxygen significantly (Wetzel, 1975). The pressure affecting oxygen solubility in Noell Lake is a combination of (a) the atmospheric pressure it is exposed to at its given altitude, (b) local meteorological conditions, and (c) the hydrostatic pressure overlying a particular depth within the lake water column (Wetzel, 1975).

The dissolved oxygen of water is said to be 100% saturated when it is in equilibrium with the oxygen of the overlying atmosphere. However, gases diffuse slowly in water and, consequently, conditions of "supersaturation" can occur for a multitude of reasons (Wetzel, 1975). Lakes can become supersaturated with respect to oxygen in circumstances where oxygen is being produced at rates which exceed the combination of oxygen-consumption and oxygen "escape" into the atmosphere. Additionally, conditions of supersaturation can occur if lake water temperature changes rapidly, and there has not been enough time to facilitate a re-equilibration of oxygen in the lake with respect to the atmosphere.

Open-Water:

Through visual analysis of the trends in dissolved oxygen, evidence of both summer stratification and intermittent mixing events are notable. The epilimnion has a fairly consistent range of dissolved oxygen percent saturation, with a minimum of 97.41% occurring on September 30th, 2012 and a maximum of 102.68% occurring on July 31st, 2012. The epilimnion remains well oxygenated throughout the 2012 open-water season since free exchange with the atmosphere facilitates a consistent resupply of oxygen into the well-mixed depths of the epilimnion. Two separate periods of supersaturation (July 28th - August 2nd and August 20th – August 24th) occur in the epilimnion of Noell Lake during the 2012 open-water period, and align with sharp increases in water temperature (Figure 4-24). It is important to note that the amount of oxygen a given volume of water can dissolve decreases as temperature cools, and oxygen diffuses slowly in water (Wetzel, 1975). As temperature increases on the days prior to the periods of supersaturation, the epilimnion oxygen content is now supersaturated with respect to the preceding cooler temperatures; there has not been sufficient time to re-equilibrate to the new temperature. Periods of slight supersaturation such as these are common in lakes, as are the slight undersaturations caused by biochemical oxidations consuming oxygen at rates which exceed resupply mechanisms. Undersaturation can occur when temperature cools abruptly, leaving the water under-saturated with respect to the preceding warmer temperatures (Wetzel, 1975).

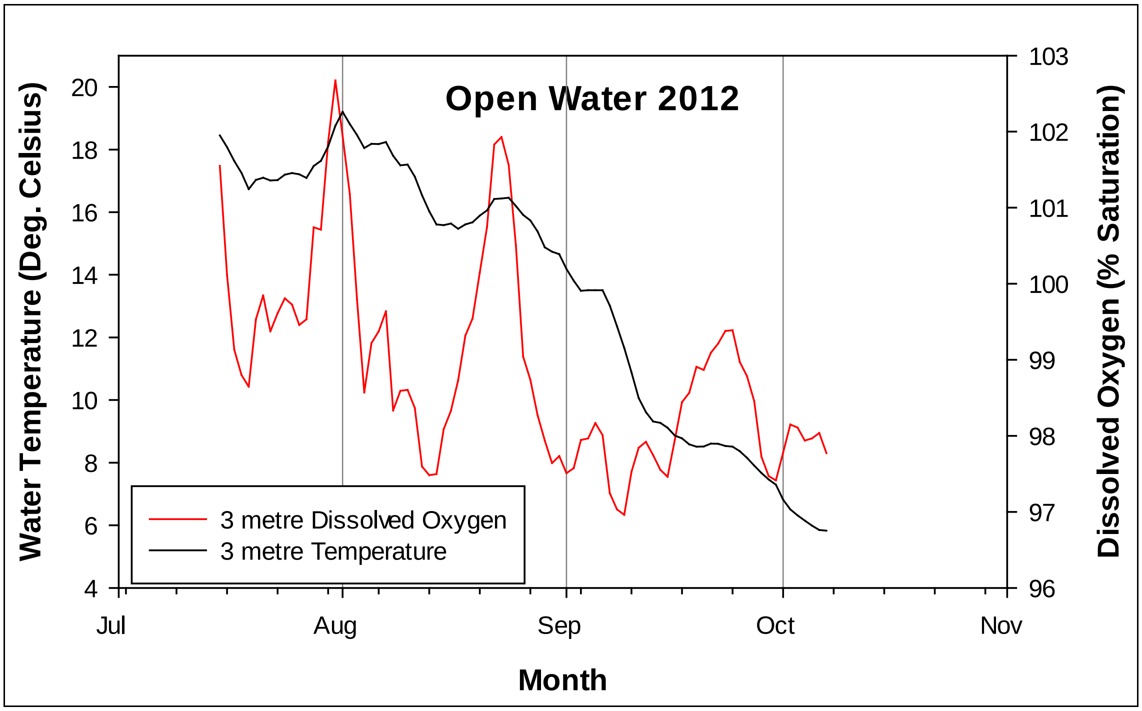


Figure 4-24: Time series of epilimnion dissolved oxygen (% saturation) and temperature (°C) of Noell Lake during the open-water period July 16 – October 7, 2012.

As discussed in previous sections, the stratification that develops in the 2012 open-water season has particularly important consequences for the dynamics of dissolved oxygen. The percent saturation of hypolimnion dissolved oxygen decreases from 86.6%

on July 16th, 2012 to 61.0% by August 9th, 2012. A mixing event occurs on August 10th, 2012 which increases hypolimnion dissolved oxygen percent saturation to 96.6%. Since the hypolimnion is, in effect, closed-off from interacting with the atmosphere while the stratification persists, resupply of dissolved oxygen is not possible. Loss of hypolimnetic oxygen is a consequence of several processes, whose relative importance varies from lake to lake. The consumption of oxygen during the oxidation of organic matter, produced in the more productive epilimnion, in the water column, and at the sediment-water interface accounts for a large portion of oxygen depletion during the open-water season. Although plant and animal respiration can account for considerable losses of dissolved oxygen in the hypolimnion, bacterial respiration associated with the decomposition of sedimenting organic matter accounts for the majority of oxygen losses (Wetzel, 1975). It is important to note that losses of oxygen due to bacterial respiration are intensive at all depths of the water column, however, losses in the hypolimnion are not offset by renewal processes such as photosynthesis and circulation that occur in the epilimnion. Furthermore, the chemical oxidation of dissolved organic matter in the hypolimnion accompanies oxygen consumption by bacterial respiration, animals, and plants (Wetzel, 1975).

The diffusion of oxygen from the epilimnion into the depleted hypolimnion is slow, and thus, a mixing event is the likely cause of the sudden increase in hypolimnetic dissolved oxygen saturation beginning on August 10th, 2012. Well-oxygenated epilimnion water is able to mix with the hypolimnion, thus, resupplying the depleted zone with dissolved oxygen. However, stratification is able to re-establish on August 18th, 2012 and the same hypolimnetic oxygen-depletion processes produce a decline to 85.8% by August 24th, 2012. Another mixing event on August 25th, 2012 eliminates the summer stratification for the duration of the 2012 open-water season and oxygen concentrations remain at just below saturation in accordance with existing solubility-temperature relationships. These mixing events align well with mixing events inferred through analysis of trends in water temperature (Section 4.4.1).

Under-Ice:

Noell Lake exhibits a fairly typical depth-profile of percent saturation of dissolved oxygen profile for a dimictic, ice-covered lake in the hypolimnion, and a curious trend in the epilimnion. As such, trends in the hypolimnion will be discussed first.

With the formation of ice cover on Noell Lake, the system is essentially sealed-off from exchanges with the atmosphere (Wetzel, 1975). Due to the same processes of hypolimnetic oxygen discussed in the “Open-Water” section (i.e., animal, plant, and bacterial respiration; chemical oxidation), oxygen is consumed resulting in the percent saturation of dissolved oxygen in the hypolimnion declining from just below complete saturation to 19.4% by the end of the open-water season. The bottom layers of the lake are slowly depleted of oxygen throughout the winter with oxygen conditions approaching and possibly experiencing hypoxia; hypoxia refers to dissolved oxygen conditions being detrimental to aerobic organisms. Furthermore, the density-driven movement of water from the epilimnion down to the hypolimnion is sufficiently slow that any oxygen dissolved in the water is depleted by the time it reaches lower depths (Wetzel, 1975). More importantly, the shift from aerobic to anaerobic conditions in the hypolimnion excludes a major volume of the lake from habitation by a majority of aquatic plant and animal species (Wetzel, 1975). The resulting dissolved oxygen curve, in which the hypolimnion is anaerobic, is referred to as “clinograde” (Wetzel, 1975).

Given the temperature and depth of the epilimnion, it appears that oxygen concentrations remain constant at saturation for the entire under-ice period. This is surprising considering a comparatively “fixed” concentration of dissolved oxygen is likely to decrease as it is consumed by the various oxidative processes occurring in the lake, especially since the regenerative processes of mixing and photosynthesis occur at lower rates in the colder temperatures (Wetzel, 1975). There are two possible explanations for this trend, and both processes could be operating simultaneously in Noell Lake.

As depth increases, the degree of dissolved oxygen supersaturation necessary to cause bubble growth, and thus, facilitate re-equilibration also increases (Wetzel, 1975). Below 1 metre, given stable water column conditions and the absence of turbulent mixing, a remarkably large degree of supersaturation may be required to cause the necessary bubble growth (Wetzel, 1975). Due to overlying hydrostatic pressure, oxygen produced at depths below 1-4 metres can remain dissolved. Overlying lake ice also increases the hydrostatic pressure impacting water at these depths, and can increase the overall amount of dissolved oxygen in the water (1975). In some instances dissolved oxygen can accumulate several hundred per cent relative to surface pressure and still not exceed the absolute saturation at a particular depth.

An additional component in explaining the under-ice trends in epilimnion dissolved oxygen percent oxygen is evident in the analysis of the concentration of dissolved oxygen (Section 4.4.2.4 Dissolved Oxygen (mg/L)). The overall concentration of epilimnion dissolved oxygen increases during the 2012-2013 under-ice season. This suggests that photosynthetically active organisms were present and resupplying the epilimnion with oxygen. Phillips and Fawley (2002) suggest that even in late winter, under conditions of maximum ice thickness, some lakes can experience supersaturated dissolved oxygen conditions; experiments performed on North Dakota lakes found that under-ice blooms of *P. aciculiferum* may have produced elevated levels of under-ice dissolved oxygen.

In summary, increasing hydrostatic pressure from overlying lake ice, stable under-ice water column conditions, and photosynthetic regeneration of dissolved oxygen may be the cause of supersaturated dissolved oxygen conditions persisting in the epilimnion of Noell Lake throughout the entire 2012-2013 under-ice season.

4.4.2.4 Dissolved Oxygen (mg/L) – Appendix E

Unlike dissolved oxygen percent saturation, which measures the amount of dissolved oxygen in lake water relative to the atmosphere, the dissolved oxygen (mg/L) content of a lake measures the actual concentration of dissolved oxygen. Both parameters follow similar trends and are influenced by the same competing processes of oxygen-consumption and -regeneration.

Open-Water:

Epilimnion concentrations of dissolved oxygen are consistent throughout the 2012 open-water period ranging from 9.5 mg/L at the start of the season to 12.2 mg/L at the end. Overall, this general increase in dissolved oxygen reflects sustained photosynthetic production of oxygen in the well-mixed epilimnion throughout the season, as well as decreasing water temperatures beginning on August 1st, 2012 which increase the overall volume of dissolvable oxygen that can be dissolved in water.

The concentration of dissolved oxygen in the hypolimnion throughout the open-water season experiences identical trends to hypolimnion percent saturation of dissolved oxygen. At the beginning of the 2012 open-water season, concentrations of dissolved oxygen are 9.5 mg/L. Hypolimnetic oxygen-depletion processes, in addition to summer stratification preventing the mixing of well-oxygenated epilimnion water to lower depths, cause a persistent decline in dissolved oxygen until a mixing event on August 10th, 2012 causes dissolved oxygen to climb as high as 9.7 mg/L on August 17th, 2012. Similar to the trends in dissolved oxygen percent saturation, the re-establishment of the summer stratification stops mixing with the epilimnion and the concentration of hypolimnion dissolved oxygen declines to 8.6 mg/L by August 24th, 2012. Another mixing event on august 25th, 2012 removes the summer stratification for the duration of the 2012 open-water period and hypolimnion concentrations of dissolved oxygen equal those of the epilimnion, rising above 12.1 mg/L, the highest values of the season.

Under-Ice:

Persistent hypolimnetic oxygen-depletion processes, in conjunction with temperature- and light-constrained processes of hypolimnetic oxygen-regeneration, result in a persistent depletion of hypolimnetic oxygen over the course of the 2012-2013 under-ice season in Noell Lake. The concentration of hypolimnetic oxygen declines from as high as 14.2 mg/L on November 5th, 2012 to 2.5 mg/L on April 14th, 2012. During the 2012-2013 under-ice season, Noell Lake approaches and most likely experiences hypoxic conditions in the lower depths of the hypolimnion. Winter habitat of aquatic plants and animals are likely affected.

As discussed in detail in the previous section (Section 4.4.2.3 Dissolved Oxygen (% Saturation)), the surprising maintenance of elevated levels of dissolved oxygen in the epilimnion throughout the 2012-2013 under-ice season is most likely explained by increasing hydrostatic pressure from overlying lake ice, stable under-ice water column conditions, and photosynthetic regeneration of dissolved oxygen throughout the entire 2012-2013 under-ice season. As a result, epilimnion concentrations of under-ice dissolved oxygen increase from 12.2 mg/L on October 8th, 2012 to over 16.0 mg/L by early-March 2013.

4.4.2.5 pH – Appendix F

Open-Water:

The pH of the epilimnion changes little throughout the open-water season of 2012, oscillating between 7.3 and 7.5. As is common in oligotrophic lakes, the pH of the hypolimnion is slightly lower than that of the epilimnion (Wetzel, 1975). Furthermore, the pH of the hypolimnion appears to be more dynamic throughout the open-water period than the pH of the epilimnion. Interestingly, there is a distinct increase in hypolimnion pH beginning on August 11th, 2012. In lakes with relatively stagnant water, increased photosynthetic activity associated with the blooming of algae can increases the pH of the lake water (Wetzel, 1975). It is evident that there are increases in both blue green algae (BGA) and Chl-*a* commencing in early August that may correspond with sudden increases in pH in mid-August (Figure 4-25). Similarly, as BGA and Chl-*a* both undergo decreases on August 21st, 2012, there is a corresponding decrease in hypolimnion pH. As Chl-*a* increases steadily from the end of August onward, hypolimnion pH also increases and remains consistently high for the duration of the 2012 open-water period. There are correlations amongst pH and both BGA and Chl-*a* in the open-water season of 2012 in Noell Lake.

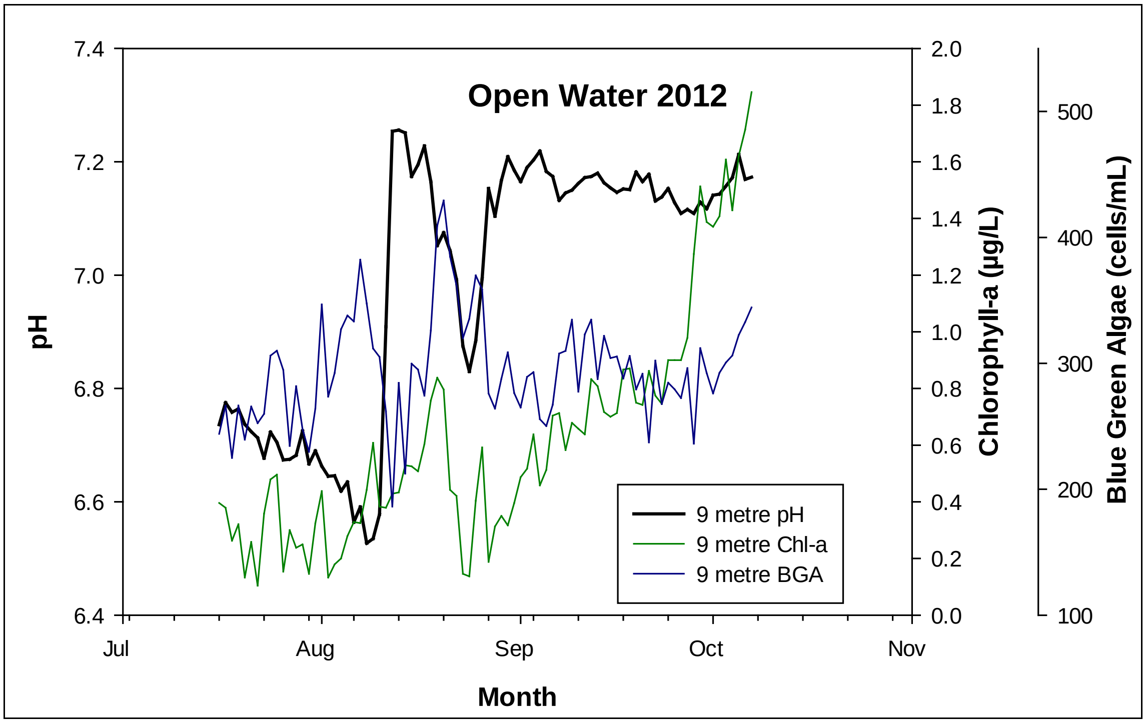


Figure 4-25: Time series of hypolimnion pH, chlorophyll-*a* (µg/L), and blue green algae (cells/mL) of Noell Lake during the open-water period July 16 – October 7, 2012.

Under-Ice:

There is little consistency between the pH of the epilimnion and hypolimnion amongst any of the 2010-2011, 2011-2012, or 2012-2013 under-ice seasons. Sensor malfunctions could explain the seemingly atypical pH patterns of the under-ice seasons. According to YSI probe information manuals, excessively low pH levels can be associated with nutrient deficiencies, metal toxicities, or other impediments to aquatic organisms. Of the under-ice pH values that appear to have been recorded by functional YSI water quality sonde components (e.g., 2010-2011 epilimnion; 2011-2012 hypolimnion; 2012-2013 epilimnion and hypolimnion), there appears to be a consistent trend of moderately decreasing pH over all three under-ice seasons. Hadley et al. (2013) suggest that the accumulation and trapping of respired CO_2 lowers lake water pH over the under-ice season.

4.4.2.6 Chlorophyll-*a* – Appendix G

Chlorophyll-*a* (Chl-*a*) is a measure of the photosynthetic capacity of the water column as determined by the concentration of chlorophyll cells in phytoplankton, and to a lesser extent, cyanobacteria (http://www.ysi.com/index.php).

Open-Water:

No YSI sonde successfully recorded open-water epilimnion Chl-*a*.

Under-Ice:

There is reason to suspect that the Chl-*a* values from the 2012-2013 under-ice season are unreliable; it is unlikely that Chl-*a* values approaching 5 µg/L occurred under-ice given the thick ice cover persisting at that time, and considering the Chl-*a* concentrations of the two under-ice seasons prior did not exceed 1 µg/L. As a result, the under-ice Chl-*a* plots for the 2010-2011 and 2011-2012 under-ice seasons were re-plotted to improve the scale of the Chl-*a* values (Figure 4-26). Chl-*a* values in both under-ice seasons undergo gradual decreases over the winter period. This is likely due to a combination of colder temperatures constraining metabolic activity, increasing coverage of snow and ice further limiting the availability of light necessary for photosynthesis, and the persistent consumption of oxygen and other resources in the depths below the epilimnion.

Epilimnion concentrations of Chl-*a* ranged from -0.05 to 0.4 µg/L during the 2010-2011 under-ice season and 0.15 to 0.7 µg/L during the 2011-2012 under-ice season. Epilimnion concentrations of Chl-a were generally higher than hypolimnion due to oxygen- and light-availability being highest just under the ice. Epilimnion concentrations ranged from -0.2 to 0.2 µg/L during the 2010-2011 under-ice season and from 0 to 0.5 µg/L during the 2011-2012 under-ice season.

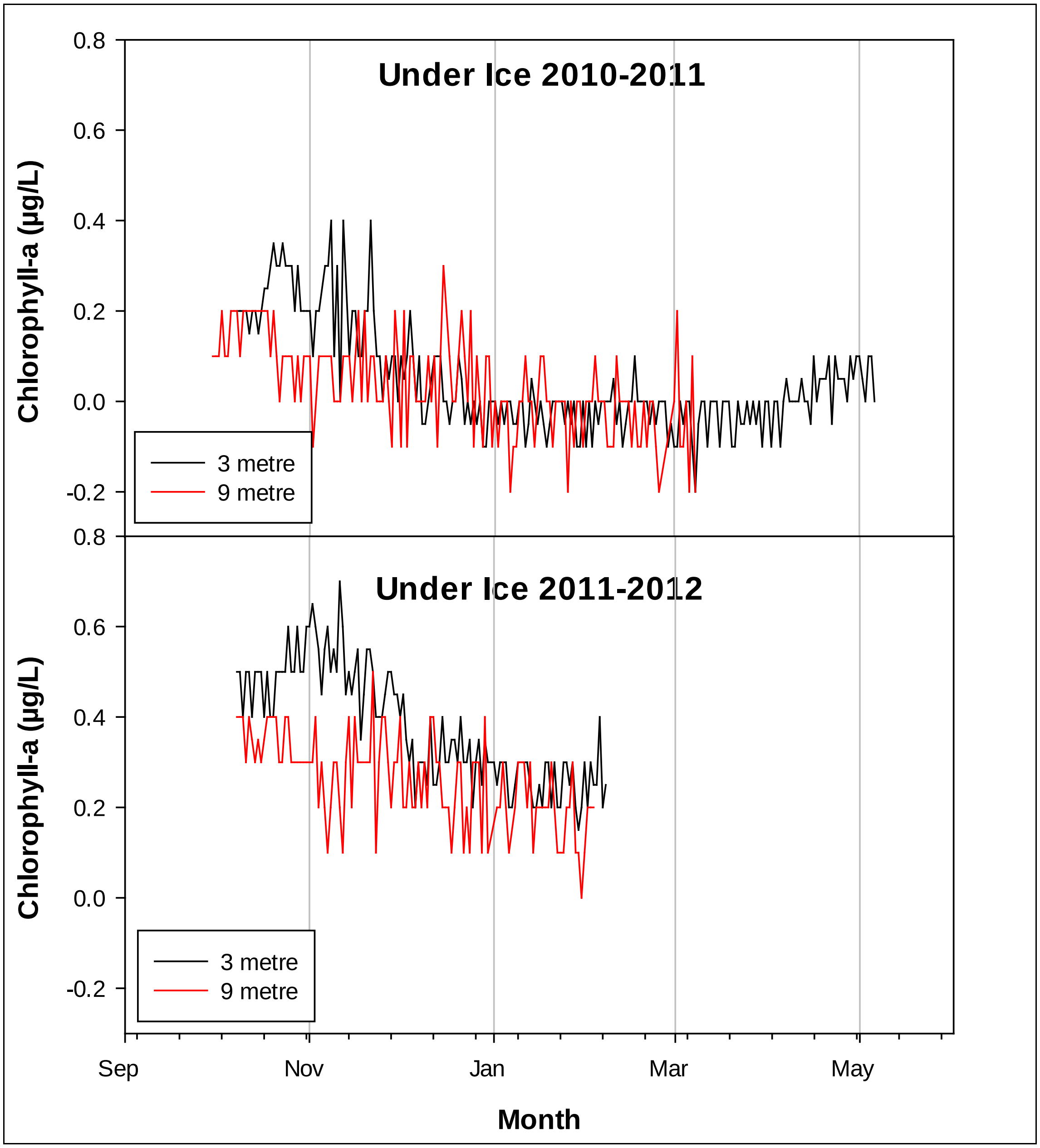


Figure 4-26: Time series of epilimnion and hypolimnion chlorophyll-a (µg/L) of Noell Lake during the under-ice periods (top) October 8, 2010 – May 6, 2011 and (bottom) October 8, 201l – February 7, 2012.

4.4.2.7 Oxidation Reduction Potential (ORP) – Appendix H

As oxygen is dissolved in water, a redox potential is created due to exchanges of electrons and free energy accompanying the reduction reaction (Stumm, 1966; Stumm and Morgan, 1970). The oxidation reduction potential (ORP) is a measurement of the oxidizing or reducing potential of the ionic constituents in a given quantity of water, given their ability to accept or lose electrons. In natural bodies of water, relatively few elements react consistently in redox processes; these elements include carbon (C), oxygen (O), nitrogen (N), sulfur (S), iron (Fe), and manganese (Mn) (Wetzel, 1975). ORP has particularly important implications for the biogeochemical cycling of micronutrients vital to aquatic flora and fauna (Wetzel, 1975). While insufficient concentrations of Fe and Mn can inhibit photosynthetic activity, elevated levels of Fe can be toxic to aquatic organisms (Wetzel, 1975). Similarly, high concentrations of Mn can inhibit green and blue-green algal populations (Gerloff and Skoog, 1957; Patrick et al., 1969).

Processes relating to bacterial and photosynthetic metabolism control spatio-temporal shifts in lake oxidation-reduction states (Wetzel, 1975). Broadly speaking, well oxygenated water contains predominantly sulfate ions while anoxic water induce the reduction of sulfate into hydrogen sulfide. ORP is greatly affected by pH, and more specifically, the activity of hydroxyl and hydrogen ions. Generally, as pH increases, ORP decreases. Furthermore, redox potential is not greatly affected by changes in dissolved oxygen content until conditions of anoxia are met (Wetzel, 1975).

More importantly, measurements of redox potential in natural waters are not well-suited for quantitative comparison and interpretation but rather qualitative and relative comparisons (Wetzel, 1975). Assessing ORP is useful for monitoring changes *in* a lake system rather than deriving conclusions based on absolute values.

Open-Water:

ORP increases from greater than 260 mv at the beginning of the open-water season to 280 mv by the end of the open-water season in 2012 (Figure 4-27). Allgeier et al. (1941) and Kjensmo (1970) suggests that well-oxygenated water generally has ORP values in the 300-500 mv range, however, it also stated that lower redox potentials are commonly observed in lake systems with elevated levels of dissolved organic compounds; particularly though lake systems with *Sphagnum* moss in the catchment vegetation. The Noell Lake catchment is well-covered by such mosses (Quinton and Marsh, 1999). Interestingly, ORP does not appear to follow the trend suggested by Wetzel (1975); instead, ORP and pH appear to exhibit a positive correlation with one another.

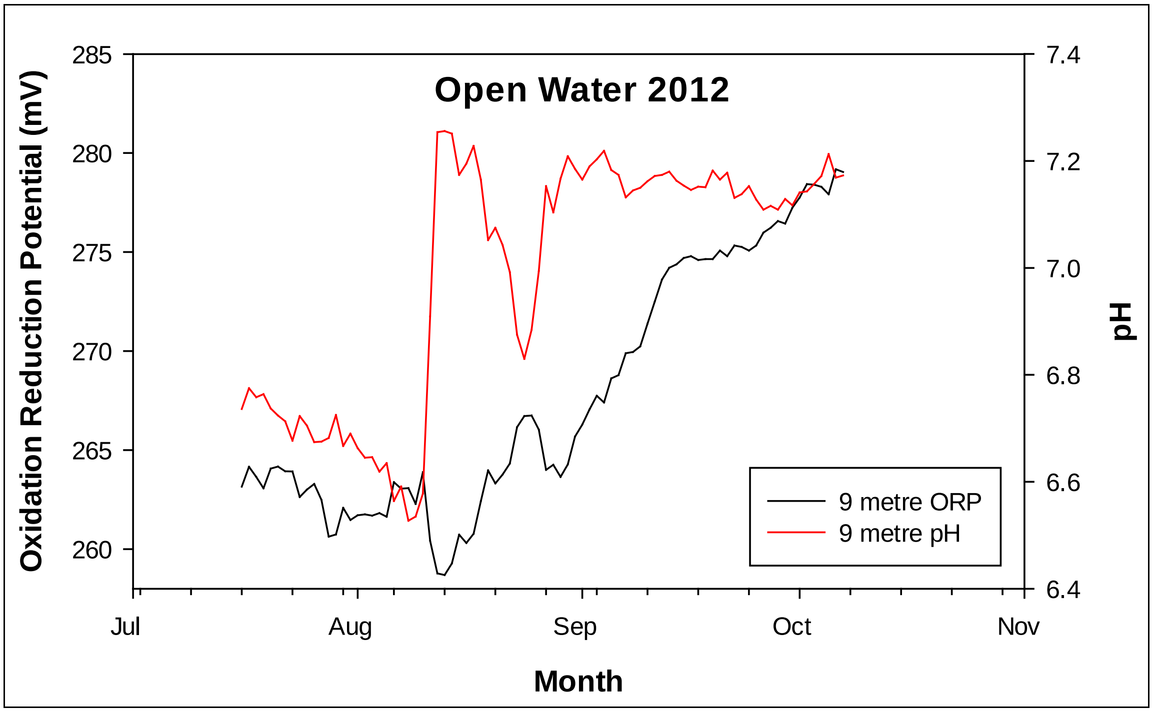


Figure 4-27: Time series of hypolimnion oxidation reduction potential (mV) and pH of Noell Lake during the open-water period July 16 – October 7, 2012.

Under-Ice:

The under-ice ORP trends of the 2010-2011, 2011-2012, and 2012-2013 seasons are dissimilar. During the 2010-2011 under-ice season, epilimnion ORP values increase gradually from 251.9 to 297.5 mV while hypolimnion values increase from 194.3 to 305.6 mV. In contrast, epilimnion ORP values during the 2011-2012 under-ice season decrease from 308.9 to 243.0 mV. Hypolimnion ORP values ranged from -102.3 to -3.4 mV during the 2011-2012 under-ice season; the presence of hydrogen sulfide results in a large negative ORP values (http://www.ysi.com/index.php). Hypoxic conditions may have been reached in the hypolimnion, which could also explain the incredibly low ORP values; unfortunately, there are no corresponding dissolved oxygen measurements during this time to corroborate the hypothesis. Epilimnion ORP was not recorded during the 2012-2013 under-ice season; hypolimnion values began at 279.7 mV at the beginning of the 2012-2013 under-ice season, peaked abruptly on December 2nd, 2012 at 378.7 mV, and then gradually declined to 336.6 mV by the end of the winter season.

4.4.2.8 Blue-Green Algae (BGA) – Appendix I

Blue-Green Algae (BGA) is a measurement of an algal group known as Cyanophyta or Myxophyceae, depending on the classification system being referenced (Wetzel, 1975). This algal group is the only group that are prokaryotic in cell structure, with the large majority of BGA being members of the coccoid family Chroococcaceae, and the filamentous families Oscillatoriaceae, Nostocaceae, and Rivulariaceae (Wetzel, 1975). Most, if not all, BGA groups exist in a variety of asymmetrical mucilaginous sheath formations (Wetzel, 1975). These organisms have important implications for a number of parameters such as dissolved oxygen, nitrogen, and carbon in addition to affecting overall ecosystem health and water quality (http://www.ysi.com/index.php).

Open-Water:

Despite considerable inter-annual and lake-specific variability in the distribution of cyanobacteria (Wetzel, 1975), there is a well-established correlation between the abundance of phytoplanktonic algae and heterotrophic bacteria (Wetzel, 1975; Vincent, 2000). Research undertaken by Vincent and Vincent (1982) reported that *Synechococcus* and *Synechocystis* were principal components of the deep water chlorophyll *a* maximum in the oligotrophic Lake Vanda, Antarctic. Upon visual inspection of (Figure 4-28), it is evident that the summer peak of both BGA and chlorophyll-*a* appear to occur at approximately the same time. Similarly, a number of other peaks within the open-water season of both BGA and Chl-*a* appear to align with one another. Furthermore, BGA content appears to change quite rapidly throughout the 2012 open-water season of Noell Lake. This is relatively typical of dimictic lakes (Rasumov, 1962; Saunders, 1971) since rapid increases and decreases in BGA content at a given depth are common in dimictic lakes (Overbeck, 1968; Saunders, 1971).

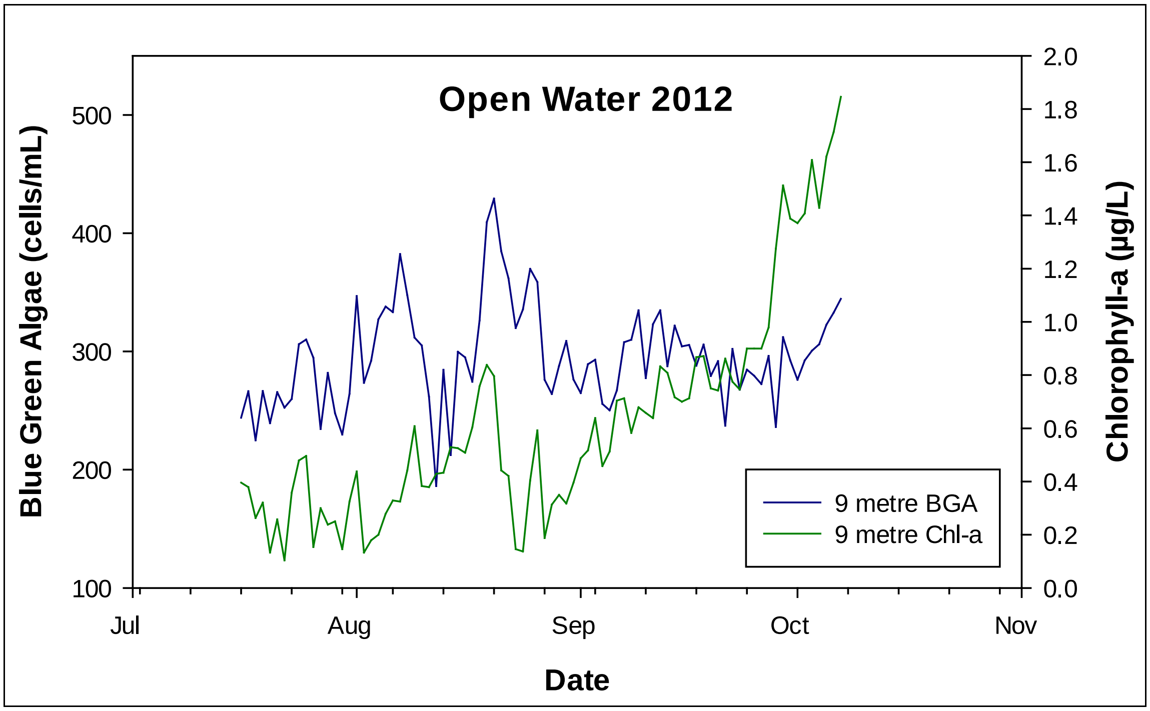


Figure 4-28: Time series of hypolimnion blue green algae (cells/mL) and chlorophyll-*a* (µg/L) of Noell Lake during the open-water period July 16 – October 7, 2012.

<u>*Under-Ice:*</u>

No YSI sondes successfully recorded BGA during any of the under-ice seasons.

4.4.2.9 Total Dissolved Solids (TDS) – Appendix J

Total dissolved solids (TDS) represent the combined total of all organic and inorganic materials dissolved in the lake water.

<u>*Open-Water:*</u>

No YSI sondes successfully recorded TDS during any of the open-water seasons.

<u>*Under-Ice:*</u>

The 2010-2011 and 2011-2012 under-ice seasons in Noell Lake exhibit similar patterns with respect to the TDS content of the water in the epilimnion. During the 2010-2011 season, the TDS content of Noell Lake increases from approximately 52 µs/cm to 62 µs/cm; the TDS content of the epilimnion in the 2011-2012 increases from 50 µs/cm to 56 µs/cm. However, the YSI sonde stopped recording on February 7th, 2012 during the

2011-2012 season, otherwise TDS most likely would have increased further. By comparison, the maximum value of 62 µs/cm during the 2010-2011 season was reached on May 6th, 2011.

Research by Zhang et al. (2012) suggests that seasonally ice-covered lakes in arid regions undergo a process of TDS migration from the ice to the unfrozen lake water below. Similar to the TDS trends characterizing Noell Lake, Zhang et al. (2012) reported that the under-ice TDS content of water in Ulansuhai Lake, China was significantly higher than during the preceding open-water period. During the freeze-up process, upwards of 80% of TDS are excluded from the newly formed ice cover and dissolved into the water below (Zhang et al., 2012) through two processes: (1) crystallography theory and (2) liquid-solid phase equilibrium theory (for a more detailed description of these processes, refer to Zhang et al., 2012).

4.5 Lake-Ice Freezeup and Breakup Sequences

4.5.1 Introduction

The National Aeronautics and Space Administration installed Moderate Resolution Imaging Spectroradiometer (MODIS) on both the Terra (EOS AM) and Aqua (EOS PM) satellites that were launched into orbit in 1999 and 2002 respectively. Combining data from both instruments, the entire Earth is imaged every 1 to 2 days. MODIS images are designed to capture large-scale global dynamics including changes in the Earth's cloud cover as well as processes occurring in the oceans, the lower atmosphere, and on land. MODIS images deliver spatial resolution as low as 500 m spatial resolution until 2008 and then improved to 250 m from then on. Through visual analysis of MODIS Terra and Aqua photographs, the annual freeze-up and break-up dates for Noell Lake, NWT can be inferred and important sequences of lake-ice freezeup and breakup can be observed.

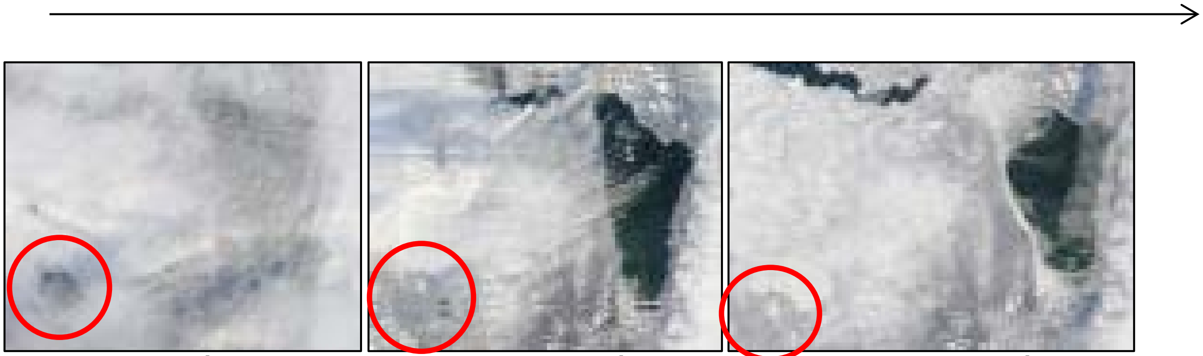
4.5.2 Methods

MODIS daily images were accessed at: http://rapidfire.sci.gsfc.nasa.gov/cgi-bin/imagery/realtime.cgi. Images from expected freezeup and breakup days were inspected until appropriate sequences were determined. Importantly, freezeup and breakup sequences can span a number of days.

4.5.3 Results

Freeze-up Sequence:

October 4th, 2004. **October 8th, 2004.** **October 10th, 2004.**



October 11th, 2004. **October 12th, 2004.** **October 15th, 2004.**

Figure 4-29: Example of a freeze-up sequence on Noell Lake, NWT (inside the red circle) occurring between October 4th and October 15th, 2004; Sitidgi Lake is visible to the right of Noell Lake.

Despite the persistent presence of cloud-cover throughout the freezeup sequence, the successive formation of complete ice-cover on Noell Lake is discernible (Figure 4-29). Due the persistence of cloud cover in the region, there are considerable gaps between image days. However, the general progression of ice-formation is still visible, especially in the larger Sitidgi Lake to the right of Noell Lake. Ice-formation begins at the margins of the lake and progresses inward.

Break-up Sequence

June 15th, 2008.

June 17th, 2008.

June 18th, 2008.

June 20th, 2008.

June 22nd, 2008.

Figure 4-30: Example of a break-up sequence on Noell Lake, NWT occurring between June 15th and June 22nd, 2008; Sitidgi Lake is visible to the right of Noell Lake.

The complete breakup and melting of lake-ice occurred over a one-week period in 2008. Ice decay begins at the margins of Noell Lake, and progresses inward (Figure 4-30). It is interesting to note that the same concentric-decay does not appear to occur on Sitidgi Lake. Instead, ice-decay appears to occur simultaneously across the lake-ice cover.

4.6 Nutrient Limitation

4.6.1 Introduction

While Ogbebo et al. (2009b) reported both nitrogen (N) and phosphorous (P) co-limitation in Noell Lake in both open-water (August 2006, July 2007) and under-ice (March 2007) seasons, there was disagreement amongst the reported nutrient-limitation depending on the method used to infer nutrient-deficiency. For example, while reported summer protein to carbohydrate, particulate carbon (PC) to particulate phosphorous (PP), and particulate nitrogen (PN) to PP ratios suggested that Noell Lake did not suffer from any nutrient deficiencies throughout the open-water period, total nitrogen (TN) to total phosphorous ratios suggested P-limitation; conversely, PC to PN ratios suggested N-limitation. As a result, a definitive conclusion on the summer nutrient status of Noell Lake was not possible. Winter data suggested that Noell Lake was phosphorous and nitrogen co-limited, which is consistent with studies done previously by Anema et al. (1990a,b) and Ramlal et al. (1991), who also reported nitrogen and phosphorous co-limitation in Noell Lake.

4.6.2 Methods

Nutrient Diffusing Substrata (NDS) were assembled by sealing four 37 mL plastic snap-cap vials (Dynalab – Rochester, NY), containing four different nutrient treatments, to a plastic test-tube rack. The four treatments were: (1) an N addition (0.5M $NaNO_3$), (2) a P addition (0.5M KH_2PO_4), (3) an N+P combined addition (0.5M $NaNO_3$ + 0.5M KH_2PO_4), and (4) a control solution comprised of agar. The NDS surfaces were silica discs (LECO Instruments Ltd. – St. Joseph, MI) and fused across the tops of the plastic snap-cap vials to cover the agar completely. The test-tube racks and snap-cap vials used are shown in (Figure 4-31).

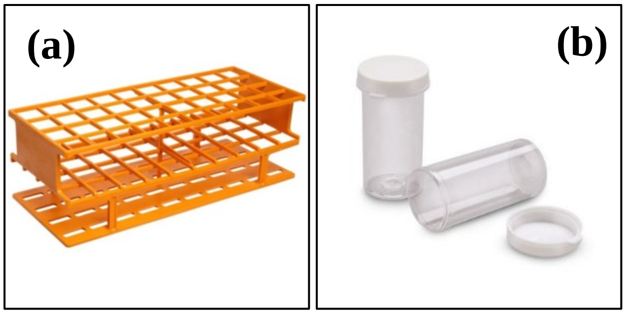


Figure 4-31: (a) Plastic test-tube rack and (b) snap-cap vials used for the assembly of the NDS apparatus.
(a) http://www.amazon.com/Nalgene-5970-0120-Acetal-Plastic-Unwire/dp/B003OBYZOY
(b) http://www.amazon.com/CLEAR-POLYSTYRENE-SNAP-VIAL-DRAM/dp/B001L7S4RW

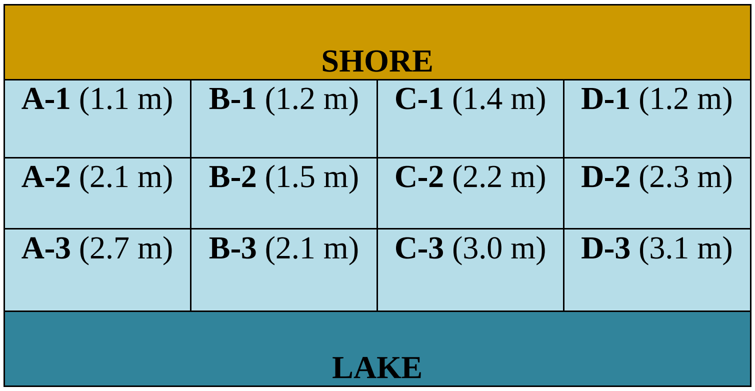
To begin preparing the NDS, eight Erlenmeyer flasks were assembled; four to contain control agar solution, and the other four to contain nutrient solutions. 10 g of agar was placed into four of the Erlenmeyer flasks designated as controls, followed by 250 mL of double-deionized water. These four flasks were then covered in tin foil and placed in an autoclave sterilizer. For the four flasks designated to contain nutrient solutions, no agar was added. Instead, each flask was filled with 250 mL of double deionized water followed by one flask receiving 21.25 g of $NaNO_3$, another flask receiving 34.45 g of KH_2PO_4, another flask receiving both, and the control flask receiving no nutritive additions. The flasks were mixed well to ensure complete dissolution, and then autoclaved for one hour.

Once removed from the autoclave, each of the four nutrient flasks was combined with one of the agar solution flasks and placed on a hot plate set to 152^{o}C. 48 silica discs were placed on hot plates set to 300^{o}C. Next, the 4 agar-nutrient solutions were poured into the appropriately labeled plastic tubes until half-filled. Next, forceps were used to lift silica discs off the hot plate and place into the open end of a tube with agar-nutrient solution. Once the silica disc had fused to the plastic, the tube was turned over, silica side-down, and placed on tinfoil to cool. After several hours, the tubes were placed in a fridge. The following day, caulking was used to seal one of each of the tubes (N, P, N+P, Control) to the plastic test-tube trays. The trays were placed in sealed coolers and transported to Noell Lake. Anchoring the various NDS in place involved connecting the components to barbeque baskets filled with barbeque briquettes through the use of plastic ties prior to deployment (Figure 4-32). On each barbeque basket, a HOBO sensor (HOBO

Water Temp Pro v2) was installed to record hourly water temperature at each NDS location. Refer to (Figure 4-34) for a display of recorded water temperatures at each NDS throughout the experimental period.

Figure 4-32: NDS (right) and briquette-filled barbeque basket (left) just prior to deployment in Noell Lake, NWT. Note the HOBO logger on the left side of the basket.

Twelve nutrient limitation experiments were divided and deployed along 4 transects running perpendicular to the northeast shore of Noell Lake, NWT (Figure 4-33). Deployment occurred on July 19th, 2012 and retrieval was on September 26th, 2012. Nutrient Diffusing Substrate (NDS) were designed following protocols developed by Tank and Dodds (2003). In each of the 4 transects, 3 NDS were placed at increasing depths from the shoreline ranging from 1 to 3 metres. Upon retrieving the NDS, it was observed that one of the plastic test-tube racks was missing the N, P, and N+P treatments. The tubes containing the various nutrient treatments and silica discs were placed in coolers immediately upon retrieval from the lake and sent to the University of Calgary for laboratory analysis of chlorophyll-*a* and ash-free dry-mass.



SHORE			
A-1 (1.1 m)	**B-1** (1.2 m)	**C-1** (1.4 m)	**D-1** (1.2 m)
A-2 (2.1 m)	**B-2** (1.5 m)	**C-2** (2.2 m)	**D-2** (2.3 m)
A-3 (2.7 m)	**B-3** (2.1 m)	**C-3** (3.0 m)	**D-3** (3.1 m)
LAKE			

Figure 4-33: Diagram of the four transects of nutrient diffusing substrate experiments in Noell Lake, NWT. Included are the installation depths and HOBO data-logger serial numbers.

4.6.3 Results and Discussion

The nutrient-limitation experiments undertaken in July of 2012 revealed that algal production is limited primarily by nitrogen in Noell Lake, with some secondary phosphorous co-limitation present.

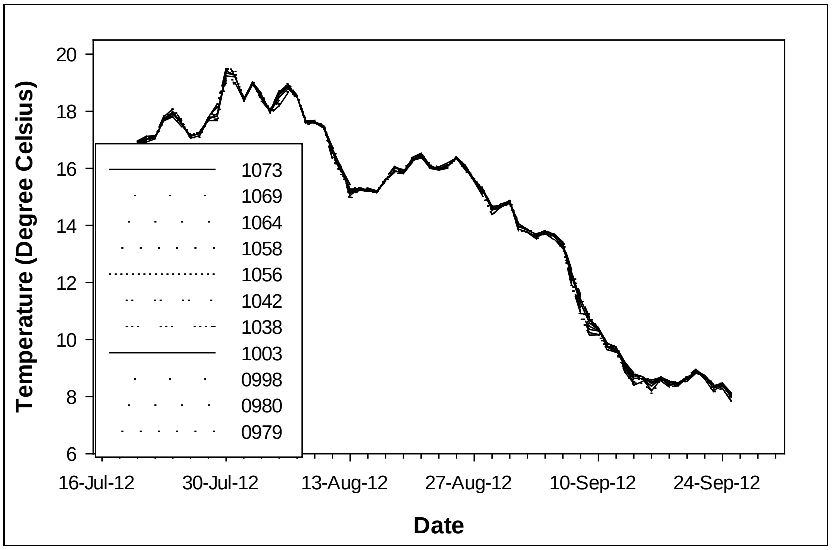


Figure 4-34: Water temperatures recorded by HOBO data-loggers installed on all 12 NDS barbeque baskets in Noell Lake, NWT.

Autotrophs:

Chl-*a* concentrations of the silica discs removed from the control treatments had a mean of 1.2 ± 0.9 μg/cm^2 and ranged from a minimum of 0.06 μg/cm^2 to a maximum 3.3 μg/cm^2; mean Chl-*a* concentration for the N addition was 4.2 ± 3.08 μg/cm^2, and ranged from a minimum of 0.7 μg/cm^2 to a maximum of 8.8 μg/cm^2; mean Chl-*a* concentration for the P addition was 1.2 ± 0.6 µg/cm^2, and ranged from a minimum of 0.3 µg/cm^2 to a maximum of 2.3 µg/cm^2; mean Chl-*a* concentrations for the N+P addition was 4.8 ± 1.8 µg/cm^2, and ranged from a minimum of 1.7 µg/cm^2 to a maximum of 7.2 µg/cm^2 (Table 4-5) and (Figure 4-35).

Table 4-5: Summary table for chlorophyll-*a* accumulation (µg/cm^2) on silica discs fused onto plastic snap-cap vials containing different treatments of nutrients: Control, Nitrogen (N), Phosphorous (P), Nitrogen and Phosphorous (N+P). Number of samples (N), mean, maximum (Max.), minimum (Min.), and standard deviation (S.D.).

Nutrient Treatment	N	Mean	Max.	Min.	S.D.
Control	12	1.2	3.4	0.1	0.9
N	11	4.2	8.8	0.7	3.1
P	11	1.2	2.3	0.3	0.6
N+P	11	4.8	7.2	1.7	1.8

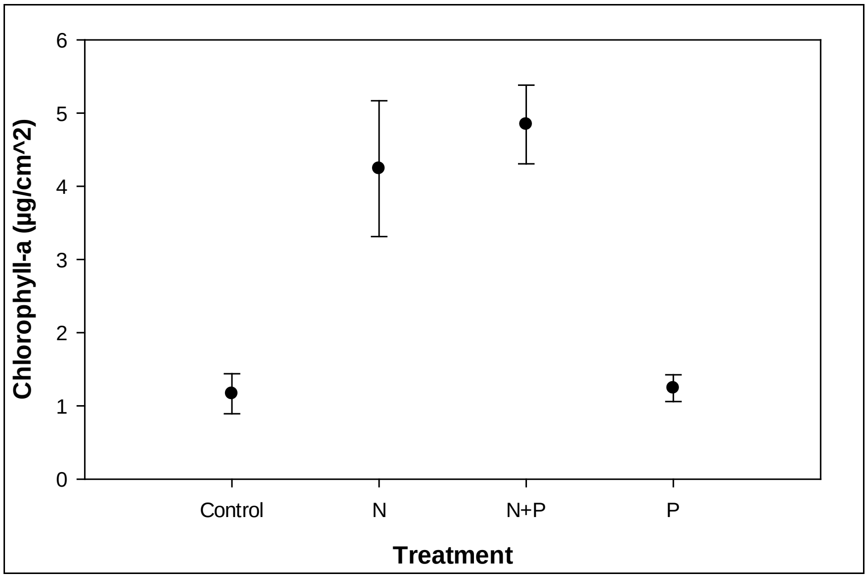


Figure 4-35: Mean concentration of chlorophyll-*a* by treatment level. Additions of N and N+P resulted in significant increases of chlorophyll-*a* relative to the control. Error bars represent the standard error of the mean.

Despite numerous attempts to transform chlorophyll-*a* data to pass tests of normality (Shapiro-Wilk) and equal variance, these assumptions were never met. As a result, a One Way ANOVA test was not possible. To isolate which groups differed from one another, pairwise multiple comparison procedures (Dunn's Method) were used (Table 4-6).

Table 4-6: Results of the Pairwise Multiple Comparison Procedures (Dunn's Method). P-values below 0.05 are bolded and indicate a statistically significant difference between treatment groups.

Comparison	Difference of Ranks	Q-Statistic	P-value < 0.05
Control vs. N	15.76	2.88	**Yes**
Control vs. N+P	20.95	3.82	**Yes**
Control vs. P	2.49	0.45	No
N vs. N+P	5.18	0.92	No
N vs. P	13.27	2.37	No
N+P vs. P	18.45	3.29	**Yes**

Evidence of nitrogen-limitation was found in Noell Lake with suspected secondary phosphorous co-limitation present as well.

Heterotrophs:

Out of the NDS deployed, ash-free dry mass (AFDM) concentrations of the silica discs removed from the control treatments had a mean of 8.9 mg/cm^2, and ranged from a minimum of 4.5 mg/cm^2 to a maximum 15.9 mg/cm^2; mean AFDM concentration for the N addition was 7.4 mg/cm^2, and ranged from a minimum of 3.9 mg/cm^2 to a maximum of 18.1 mg/cm^2; mean AFDM concentration for the P addition was 9.7 mg/cm^2, and ranged from a minimum of 4.9 mg/cm^2 to a maximum of 21.9 mg/cm^2; mean AFDM concentrations for the N+P addition was 11.3 mg/cm^2, and ranged from a minimum of 5.3 mg/cm^2 to a maximum of 18.1 mg/cm^2 (Table 4-7) .

Table 4-7: Mean ash-free dry mass (AFDM) that accumulated on silica discs fused onto plastic snap-cap vials containing different treatments of nutrients: Control, Nitrogen (N), Phosphorous (P), Nitrogen and Phosphorous (N+P). Number of samples (N), mean, maximum (Max.), minimum (Min.), and standard deviation (S.D.).

Nutrient Treatment	N	Mean	Max.	Min.	S.D.
Control	12	8.9	15.9	4.5	4.0
N	11	7.3	18.1	3.9	4.4
P	11	9.7	21.8	4.9	4.5
N+P	11	11.3	18.1	5.3	4.5

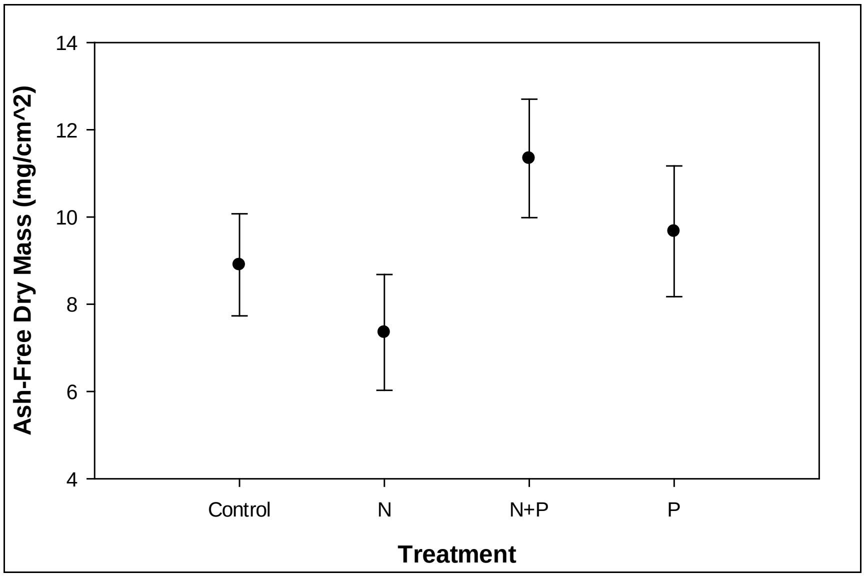


Figure 4-36: Mean concentration of ash-free dry mass by treatment level. No additions of nutrients differed significantly from the control. Error bars represent the standard error of the mean.

Through visual inspection of the mean and standard error plots, it is evident that the differences in the mean values among treatment groups are not great enough to exclude that differences are due to random sampling variability (Figure 4-36); as a result, multiple pairwise comparison tests were not performed on the AFDM data. Although the N- and N+P-treatments may differ significantly from one another, no nutrient treatment differs significantly from the control. As a result, it is not possible to suggest which nutrient may be most strongly limiting heterotrophic productivity in Noell Lake, NWT. Instead, other factors such as temperature and light may incur more robust controls on heterotrophic productivity.

4.7 Conclusion

Future hydro-ecological change in Noell Lake can now be compared to the continuous data generated through this study. A number of novel findings pertaining to previously unknown limnological properties of Noell Lake were obtained:

1. Ranges and temporal variation of current chemical, physical, and biological properties of the lentic system were defined: for both open-water and under-ice seasons; in the epilimnion and hypolimnion; and over

successive years. Continuous measurements of water temperature, conductivity, specific conductivity, percent saturation of dissolved oxygen, concentration of dissolved oxygen, pH, chlorophyll-*a*, oxidation reduction potential, blue green algae, and total dissolved solids were recorded over the period September 2011 through July 2013.

2. Increased understanding of the temporal variation in correlations amongst several limnological and physical parameters in Noell Lake was possible. Interesting relationships include blue green algae and chlorophyll-*a*, oxidation reduction potential and pH, as well as water temperature and conductivity.
3. Analysis of both the concentration and percent saturation of dissolved oxygen generated by the buoy and mooring components generated detailed insight into the development of hypoxic conditions in the hypolimnion during the under-ice season of 2012-2013. The percent saturation of dissolved oxygen falls below an important ecological threshold of 30% on March 3, 2013 in Noell Lake; furthermore, the concentration of dissolved oxygen decreases below 3 mg/L on March 22, 2013. It is suggested that aquatic life can no longer be sustained at concentrations of dissolved oxygen below 2 mg/L although different organisms certainly have different thresholds (Vaquer-Sunyer and Duarte, 2008). However, zooplankton and fish are surviving under the ice which might indicate possible “refugia” in Noell Lake, facilitating the survival of these organisms.
4. Analysis of water temperature data recorded by the buoy and mooring components, and hobo-data loggers generated detailed information on daily thermodynamic processes. HOBO data loggers continuously recorded temperature at 1-metre intervals down to 15 metres in Noell Lake for an entire year; buoy and mooring components continuously recorded epilimnion and hypolimnion water temperatures for one open-water season (2012) and three under-ice seasons (2010-2011, 2011-2012, 2012-

2013). The integration and analyses of water temperature data facilitated the investigations of detailed changes to the thermal profile of Noell Lake.

5. The analyses of water temperature and dissolved oxygen data generated detailed information on seasonal stratification and mixing dynamics. Two mixing events are evident during the 2012 open-water season: August 12th and August 26th, 2012. The mixing events are corroborated by the convergences of epilimnion and hypolimnion water temperature, concentration of dissolved oxygen, and percent saturation of dissolved oxygen. Furthermore, the water temperature data recorded by the HOBO data loggers also corroborate the dates of these mixing events. Additionally, while the YSIs installed on buoy and mooring components did not record data long enough to capture mixing during spring turn-over, the HOBO data loggers did. Through analyses of the HOBO data loggers strapped at 1-metre intervals down to 15 metres in depths, mixing events are discernible on June 10th, 2013 and June 23rd, 2013. These represent an important establishment of 2 separate mixing periods in Noell Lake; one in June and another in August. It is also interesting that both mixing periods contained two separate mixing events with brief re-stratification occurring intermittently. Winter stratification persists from October 23rd, 2012 until June 10th, 2013.
6. Analysis of under-ice total dissolved solids (TDS) illuminated the importance of lake-ice formation with respect to the exclusion of ionic constituents from the ice into the underlying water. Importantly, under-ice TDS can increase to levels higher than during the open-water season.
7. An additional successful module of the limnological investigation of Noell Lake was the data generated through the various water quality sampling campaigns. Water samples were collected during a multitude of open-water and under-ice seasons and shipped to NLET laboratories for further chemical and nutrient analyses. Results from these analytical analyses provide a sequence of robust geochemical “snap-shots” of instantaneous conditions in Noell Lake at a multitude of depths and locations around the

lake. A multitude of means, minimums, maximums, and standard deviations of specific conductivity, pH, alkalinity, F, Cl, SO_4, colour, dissolved organic carbon, dissolved inorganic carbon, hardness, Ca, Mg, Na, K, NO_2, SiO_2, turbidity, dissolved phosphorous, NH_3, NO_3NO_2, organophosphate, particulate organic phosphate, particulate organic nitrogen, total nitrogen, total phosphorous, and total dissolved nitrogen were generated, facilitating an important assessment of the spatial and temporal variability of a multitude of parameters in Noell Lake. Furthermore, results from these analyses include similar parameters to historical analyses undertaken previously in Noell Lake. For example, various time series exist for specific conductivity, Mg, Na, K, Cl, Ca, SO_4, and SiO_2 with measurements being collected in 1982, 1986, 2007, 2011, 2012, 2013. These reflect important on-going changes in the geochemical baseline conditions of Noell Lake.

8. The results of nutrient-limitation experiments undertaken in July of 2013 indicate that Noell Lake fits within current paradigms proposing high rates of nitrogen-deficiency in Arctic-tundra lakes due to the high phosphorous-adsorbing capacity of lake sediments in the region (Levine and Whalen, 2001).

4.7.1 Ecological Relevance to Users of Noell Lake

The first ever establishment of continuous baseline data of Noell Lake was initiated during a comparatively undisturbed period in the history of the lake; construction of the aforementioned all-season road and commencement of the 2012 forest fire during had yet to occur. In general, the findings discussed above infer broader implications to users of Noell Lake. Dissolved oxygen and water temperature data suggest that the development of hypoxic conditions could affect fish distribution in the lake, especially under ice. Additionally, deep holes in the lake that have been identified as important to local fishing may not be optimal locations for fish.

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CHAPTER 5: TEMPORAL AND SPATIAL TRENDS OF CHLOROPHYLL-*A* IN HIGH-LATITUDE LAKES ACROSS THE NORTHERN HEMISPHERE (1965-2010)

Abstract

Recent climate impact assessments on Arctic regions converge on concluding that since 1980, Arctic region air temperatures have been increasing at approximately twice the global rate of warming. Arctic lakes in particular continue to demonstrate distinct changes to their hydro-ecology as climate-regime changes occur. Changes to lake-ice duration and thickness, water temperature, concentrations of dissolved organic carbon and nutrients, and irradiance regimes are likely altering primary productivity and algal biomass in Arctic lentic systems. Despite the growing availability of northern hemisphere limnological data, no temporally-integrative and spatially-comprehensive analyses of changes to lake productivity, as measured by algal biomass, have been undertaken. Published data on algal biomass, total phosphorous, total nitrogen, dissolved organic carbon, and latitude was synthesized to determine whether average algal biomass had experienced significant variations over the past few decades in response to physical and biogeochemical changes affecting northern aquatic ecosystems. Datasets were compiled through the amalgamation of 51 academic sources, obliged data-acquisition requests, and data retrieved from the World Lake Database; this resulted in data for over 1000 lake-years, ranging in latitudes from 50^{o}N to 83^{o}N. Average algal biomass observed during the open-water seasons have decreased over the period 1965-2010, especially in lower Arctic regions. 4 major hypotheses were outlined, based on existing empirical evidence as well as model projections, for the decreasing concentrations of Chl-*a* observed in high latitude systems of the northern hemisphere. Recent temporal trends in algal biomass suggest that Arctic lakes are experiencing distinct and dramatic changes to their primary productivity, which has important implications for lake trophic structure.

Keywords: Arctic lakes; Chlorophyll-*a*; Dissolved nutrients; Ultraviolet radiation; Dissolved organic carbon; Mixing and stratification; Lake ice cover

5.1 Introduction

In recent decades, high latitude systems of the northern hemisphere have warmed more than any other region on the earth (Burrows et al., 2011); these systems are expected to continue being subjected to the most pronounced increases in temperature (Diffenbaugh and Giorgi, 2012; Collins et al., 2013). Consequently, ecosystems whose boundaries are situated completely or partially within these regions have reflected some of the most robust and distinct responses to recent climate warming (Vaughan et al., 2013) and will continue to be some of the most vulnerable moving forward (Gonzalez et al., 2010). Of these vulnerable northern ecosystems, the Intergovernmental Panel on Climate Change (IPCC) has identified Arctic lakes as being one of the aquatic habitats most susceptible to direct climate effects, especially rising air temperatures (Scholes et al., 2013), since the features of Arctic limnological systems are contingent on air temperatures being persistently $<0^{o}$C for the majority of the year (Vincent et al., 2012). A multitude of independent and associated changes are likely affecting, and will continue to affect, the productivity and food web structure of high latitude lakes in the northern hemisphere (Wrona et al., 2006; Smol and Douglas, 2007; Vincent and Laybourn-Parry, 2008). The scientific community, however, is still in relatively early stages of understanding how changes to Arctic lake hydro-ecology will manifest themselves with regard to global extent, regional variability, and concomitant changes to lake trophic structure (Lenters et al., 2011).

Warming of the well-mixed surface water is likely disturbing the commencement, persistence, and stability of summer thermoclines (Winder and Schindler, 2004); decreases in the overall duration and thickness of lake ice-cover is likely altering irradiance regimes (Bonilla et al., 2009), although coincident increases in coloured dissolved organic matter (CDOM) may be interceding in the effects of increased ultraviolet radiation (UVR); and the compounding effects of permafrost thaw, increased industrial development, and alterations to hydrological regimes in Arctic regions are likely affecting the biogeochemistry of lake water (Frey and McClelland, 2009; Kokelj et al., 2009). The amalgamation of these changes, in addition to a multitude of other simultaneous climate- and anthropogenic-stressors, have affected, and are likely to continue affecting the phenology and morphology of Arctic freshwater biota (Daufresne et al., 2009; Winder et al., 2009), especially plankton (Larsen et al., 2014), since many species currently live at the extremes of their environmental tolerances (Nõges et al., 2010). However, while recent assessments of terrestrial productivity suggest that the Arctic terrestrial biosphere has greened dramatically since 1982 in relation to increases in air temperature and extended growing seasons (Xu et al., 2013), no spatially-comprehensive and temporally-integrative analyses have investigated temporal trends in the productivity of the abundant lakes of the circumpolar Arctic. Predicting the character and magnitude of changes to the various components of Arctic lake ecosystems is complicated by a history of variable lake responses to climate change (Vincent et al. 2012). Lakes spanning a diverse range of environmental conditions, differing in size, underlying bedrock geology (Keatley et al. 2008), local topographical features, ice phenology (Michelutti et al. 2006), watershed nutrient-availability, and productivity (Finkelstein and Gajewski 2007; Finkelstein and Gajewski 2008), facilitate the development of unique limnological systems capable of dissimilar responses to warming.

Conclusions founded on empirical investigations (Palmer et al., 2014), analyses of satellite imagery (Schneider and Hook, 2010; 2012), and modelling exercises (Kirillin, 2010) converge on the finding that surface-water temperatures of large inland water bodies have warmed in recent decades in response to warmer air temperatures, especially in the mid- and high-latitudes of the northern hemisphere (Schneider and Hook, 2010; Larsen et al., 2014; Vaughan et al., 2013). Schneider and Hook (2012) utilized spaceborne thermal infrared imagery generated during the period 1985-2009 to produce time series of surface-water temperatures for 169 of the largest inland water bodies worldwide; results indicated that summer night-time surface-water temperatures of the investigated water bodies have warmed at an average rate of $0.045 \pm 0.011\ ^{o}$C/yr, with warming being the most pronounced in mid- to high-latitudes of the northern hemisphere (Schneider and Hook, 2012). A separate 12-lake investigation of surface water temperatures (seven in south-central Ontario and five in north-central Wisconsin) over a 25-year period (1981-2005) revealed that the investigated lakes experienced increases in temperature and thermal stability, contributing to an overall decrease in mixing depths (Palmer et al., 2014). Furthermore, Palmer et al. (2014) determined the best predictors of mixing depth to be dissolved organic carbon and spring air temperature, suggesting that both broad-scale climatic factors - in combination with lake-specific characteristics - modify the thermal responses of individual lakes to climate warming.

As ambient air temperatures increase, associated increases in incident longwave radiation, coupled with decreases in the overall loss of latent and sensible heat, facilitate an overall warming of lake surface-temperatures (Livingstone, 2003). However, warming does not occur homogenously along the lake-depth profile; Palmer et al. (2014) reported that while, on average, nearly all of the measured increases in fall air-temperatures were mirrored in increases in surface and epilimnion temperatures, only half of the increases in air-temperature resulted in metalimnion warming, and hypolimnion temperatures were unaffected altogether. The compounding effects of (a) rates of surface-water warming exceeding those of lower depths and (b) water density increasing nonlinearly with temperature, contribute to stronger thermal gradients of density and salinity in lake water columns in response to increases in air temperature (Palmer et al., 2014). Correspondingly, modelling exercises have also reported evidence of stronger stratification in lakes in response to climate warming, with associated transitions to dimictic mixing regimes in some lakes (Kirillin, 2010). Thermal responses to increases in atmospheric heat inputs differ between dimictic and polymictic lakes, and can be distributed unequally in the seasonal cycle (Kirillin, 2010). Decreasing lake-ice cover, as a consequence of increases in air temperatures, has amplified the warming of water bodies, causing lakes to warm more rapidly than air temperature in some cases (Larsen et al., 2014).

Planktonic biomass is affected by warming lake water, and changes to plankton species composition, in relation to increasing water-temperatures, have been reported by several investigations (Christoffersen et al., 2008; Heino et al., 2009; Jansson et al., 2010). A paleolimnological investigation of a lake located in the Siberian Arctic indicated that warm, ice-free summer conditions were associated with highest primary productivity, while periods of perennial ice cover were associated with the lowest (Melles et al., 2007). Furthermore, 6-14^{o}C increases in lake water temperature were associated with growth-rate increases exceeding 10-fold in some diatom species (Doyle et al., 2005). However, the responses of phytoplankton and algal communities to temperature- and mixing-regime changes are complex; for example, decreases in vertical mixing due to stronger stratification can provoke lower production and/or higher plankton mortality in lakes (Kirillin, 2010).

Assessing recent changes to lake-ice phenology is complicated by a legacy of sparse empirical datasets on which analyses can be undertaken, limited lakes with long term records, discontinued records (Prowse et al., 2011a, b), and methodological inconsistencies (Beltaos and Prowse, 2009). However, available evidence indicates that average seasonal ice-cover is decreasing in both thickness and duration in the mid- to high-latitudes of the northern hemisphere (Vaughan et al., 2013). Benson et al. (2012) examined trends in freeze- and break-up dates for 150-, 100-, and 30-year periods in 75 northern hemisphere lakes ending in 2005; the most recent 30-year period exhibited the most pronounced decrease in lake-ice duration, with break-up occurring 1.9 days earlier per decade and freeze-up occurring 1.6 days later per decade. Similarly, a separate analysis undertaken by Jensen et al. (2007) concluded that break-up occurred 2.1 days per decade earlier, and freeze-up occurred 3.3 days per decade later, in 65 water bodies located between Minnesota and New York. Furthermore, Jensen et al. (2007) reported that rates of change in ice phenology are greater over the more recent period of 1975-2004 than over the period 1846-1995. Wang et al. (2012) reported that average ice-cover duration decreased by 71% over the period 1973-2010. Trends in northern hemisphere lakes indicating shorter under-ice seasons, as a result of later freeze-up and earlier break-up, are corroborated by analyses in Switzerland (Livingstone, 1997; Magnuson et al., 2000), Russia (Livingstone, 1999; Magnuson et al., 2000), Finland (Magnuson et al., 2000; Korhonen, 2006), Japan (Magnuson et al., 2000), the United States of America (Magnuson et al., 2000; Hodgkins et al., 2002), Canada (Magnuson et al., 2000; Duguay et al., 2006), Germany (Livingstone et al., 2010), and Sweden (Weyhenmeyer et al., 2005; Livingstone et al., 2010). Although exceptions exist, the rates of change in the timing of lake-ice phenology are: greater for break-up than freeze-up, greater at higher altitudes, and greater during more contemporary time periods than distant time periods (Vaughan et al., 2013).

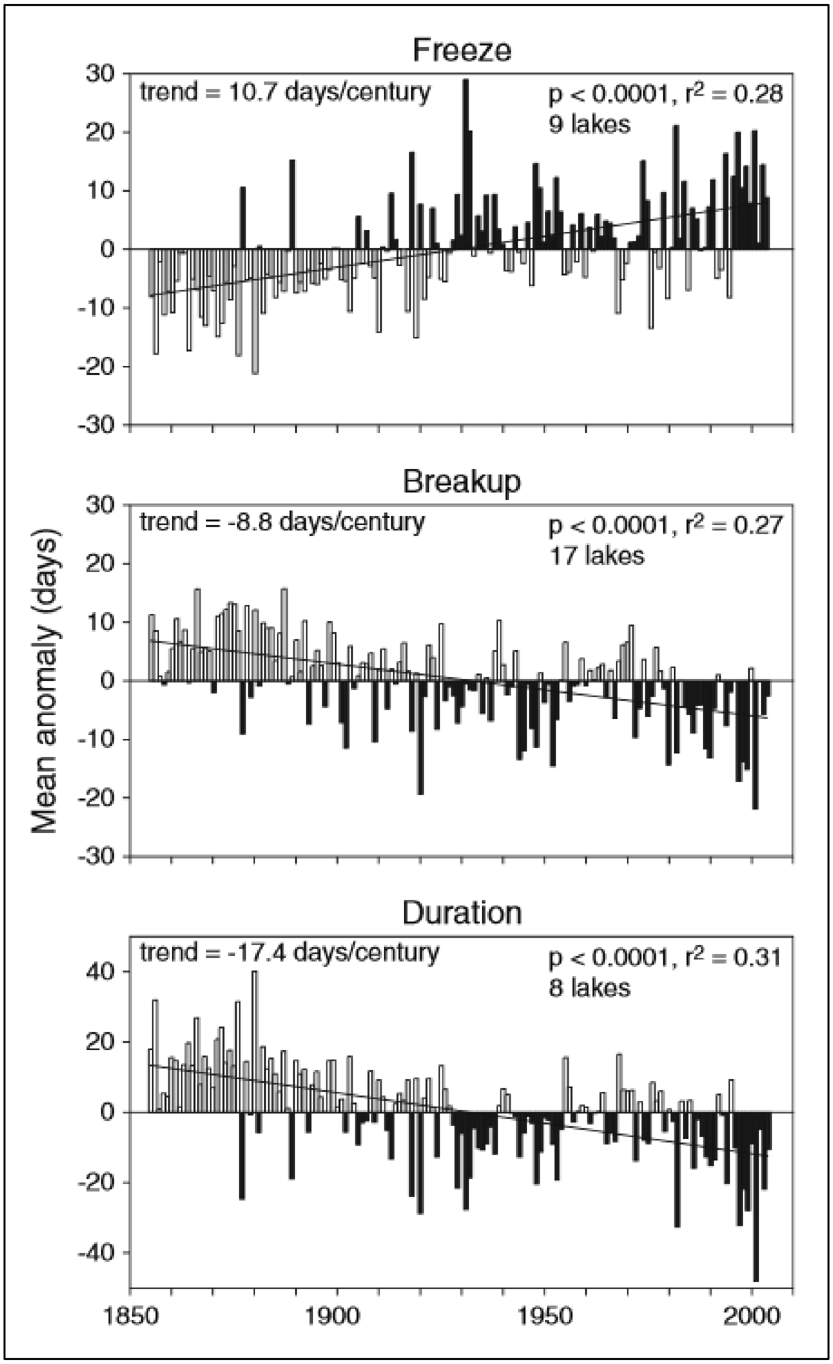

Figure 5-1: Trends for the 150 years from 1855-6 through 2004-5 in mean annual values of freeze day (*top*), breakup day (*middle*), and ice-cover duration (*bottom*), expressed as anomalies from the 150-year mean. Years with earlier than average freeze, later than average breakup, and longer than average duration are given as gray bars, and years with later than average freeze, earlier than average breakup, and shorter than average duration are black bars. The linear trend is provided as is the slope, p-value, r^2, and the number of lakes for each ice measure. The year shown is the beginning year of the winter season. *Figure from Benson et al. (2012).*

Decreased persistence of ice-cover in lakes, as well as reduced ice-cover thickness, have expansive impacts on the limnology of northern lake ecosystems (Vincent et al., 2008; Mueller et al., 2009; Prowse et al., 2011b). Annual rates of primary production may be amplified through increased light-availability for photosynthetic activity or by expanding suitable habitats for the active photosynthetic community into deeper parts of the lake, thus increasing the total volume of the lake capable of facilitating net primary production (Vincent et al., 2011). For example, Antoniades et al. (2009) reported increases in the activity and magnitude of deep-layer photosynthetic sulfur bacteria in Lake A on Ellesemere Island, Nunavut in response to decreased coverage of snow and ice. Furthermore, wind-induced mixing may increase as ice-free conditions last longer, entraining nutrients from deeper layers of the lake into the photosynthetic zone (Vincent et al., 2012). Vincent et al. (2009) reported that a remarkably warm 2008 caused melting of the persistent ice-cover of Lake A; as the halocline deepened, an associated upwelling of nutrients stimulated mixolimnion phytoplankton production (Veillette et al., 2011). Research by Smol et al. (2005) and Rühland et al. (2008) suggest that lakes distributed across the Canadian Arctic have experienced increases in both productivity and freshwater diatom diversity over the past several hundred years due to an overall decrease in ice-cover duration. However, changes did not manifest themselves at similar times or similar scales (Devlin and Finkelstein 2011); while some diatom communities began responding to climate warming in the mid-1800s (Douglas et al. 1994; Antoniades et al. 2005), discernible changes in other lakes did not occur until the twentieth century (Perren et al., 2003; Finkelstein and Gajewski, 2008).

An important consequence of earlier lake-ice break-up are the potentially deleterious effects of increased ultraviolet radiation (UVR; 280-400 nm) on aquatic organisms unable to adjust their photoprotection strategies (Rautio et al., 2011). UVR directly affects the deoxyribonucleic acid (DNA), fecundity, sex ratio, developmental and growth rates, pigmentation, feeding and swimming behaviour, and survival of aquatic invertebrates in lakes (Perin and Lean 2004). In Arctic landscapes, coloured dissolved organic matter (CDOM) and sustained ice- and snow-cover are the only forms of physical protection from UVR that northern aquatic organisms utilize (Laurion et al. 1997).

However, many Arctic lakes are shallow and clear, with concentrations of protective dissolved organic carbon (DOC) regularly being less than 2 mg/L (Rautio and Korhola 2002). As a result, detrimental intensities of UVR are a ubiquitous and pervasive threat to aquatic organisms across the circumpolar Arctic (Vincent and Laybourn-Parry 2008).

Despite ultraviolet-B radiation (UVBR; 280-320 nm) being more damaging on a per photon basis than ultraviolet-A radiation (UVAR; 320-400 nm) (Cullen et al., 1992), the comparatively greater attenuation of UVBR in the water column can render UVAR more constraining to integrated water column productivity (Hiriart-Baer and Smith, 2004). The photosynthetic capacity, biological composition, and ability of phytoplankton to utilize available nutrients are negatively affected by UVR (Harrison and Smith, 2009), and earlier ice melting can increase UVR-exposure by an order of magnitude in polar aquatic systems (Vincent et al., 2007). Even in the 1950s, high UVR was the reported cause of phytoplankton-scarcity in aquatic ecosystems of northern Patagonia and Swedish Lapland (Thomasson, 1956). Through analyses of fossil pigment reconstructions, Leavitt et al. (2003) reported declines in algal biomass of 10- to 25-fold associated with corresponding periods of increased UVR-exposure. More recently, the lack of *Daphnia sp.* in ponds located in Finnish Lapland was related to high UVR (Rautio and Korhola, 2002). The experimental removal of snow off a perennially ice-covered lake resulted in a sixteen-fold increase in biological UVR exposure and only a thirteen-fold increase in photosynthetically active radiation (PAR) (Belzile et al., 2001).

In addition to increases of UVR resulting from diminishing lake-ice cover, stratospheric ozone-depletion may have caused simultaneous increases in levels of UVR affecting the ecology of Arctic lakes (Hartmann et al., 2013). Recent assessments of global stratospheric ozone converge on the conclusion that stratospheric ozone has declined from pre-1980 values (Hartmann et al., 2013), with Arctic and other high latitude locations (>60^{o}) having experienced particularly pronounced ozone-depletion (Harrison and Smith, 2011; McKenzie et al., 2011). Weatherhead et al. (2005) estimate that anthropogenic activity, relating to the production of ozone-depleting substances (ODSs) such as chlorofluorocarbons (CFCs), has resulted in stratospheric ozone losses as high as 7% over the Arctic, with associated increases in UVR-B affecting the Arctic biosphere (Arróniz-Crespo et al., 2011). With the establishment and enforcement of the

Montreal Protocol in 1987 and 1989 respectively, production of ozone-depleting substances was successfully reduced (McKenzie et al., 2011). However, recovery of the Arctic ozone layer is gradual and may not fully recuperate until 2050 (World Meteorological Organization, 2007). Increased exposure to UVR from changing ice phenology, in addition to ozone-depleting processes, may have caused significant changes to the productivity of Arctic lakes. However, concomitant increases in concentrations of dissolved organic matter in lakes, related to the greening of Arctic landscapes (Xu et al., 2013), complicate predicting the effects of increased UVR on Arctic lake primary productivity.

Feedback mechanisms related to intensifying decreases in land- and ocean-surface albedo are likely to continue increasing terrestrial winter temperatures until 2100 (Callaghan et al., 2005). As winter periods warm at a greater rate than summer periods, the seasonality of temperature over the Arctic is diminished, contributing to longer seasons during which Arctic and boreal vegetation can grow (Snyder, 2013). Consequently, there is growing evidence of increased plant growth across the tundra in response to Arctic warming (Tape et al., 2006; Hudson and Henry, 2009; Grant et al., 2011). In recent years, various analyses have reported a “greening” of high latitude regions (Myneni et al., 1997; Tucker et al., 2001; Jia et al., 2003), with the greatest greening occurring along the Beaufort Sea, high Arctic of North America, and eastern European Arctic (Zhang et al., 2008; Pouliot et al., 2009; Bhatt et al., 2010; Forbes et al., 2010; Walker et al., 2011; Epstein et al., 2012; Macias-Fauria et al., 2012).

Xu et al. (2013) assessed responses in plant productivity, as measured by Normalized Difference Vegetation Index (NDVI), and related them to changes in summer warmth index, as measured by the sum of the monthly-mean temperatures above freezing (expressed as ^{o}C/month). Changes in NDVI were inferred from satellite observations imaged during the period 1982-2012, and displayed positive trends with the summer warmth index which have increased by an average of 5^{o}C/month over the whole Arctic (Xu et al., 2013). Despite spatial heterogeneity in the magnitude of greening trends, approximately one third of the circumpolar Arctic has shown increased plant productivity during the three decades investigated (Xu et al., 2013). Substantial increases in terrestrial organic carbon production associated with the northward expansion and densification of shrubs and trees in northern hemisphere high latitude landscapes can result in increased export of allochthonous dissolved organic matter to lakes and other Arctic aquatic systems (Wrona et al., 2006; Vincent et al., 2012).

The effect of changes in allochthonous DOC on Arctic lake productivity is complex, and while decreases in photosynthesis can occur due to increased shading by CDOM and additional terrigenous constituents (Watanabe et al., 2011), offsetting increases in photosynthesis can also occur as phytoplankton are better protected from damaging UVR (Vincent et al., 2012). Additionally, changes in DOC have also been suggested to affect the thermal stratification and heat budgets of lakes (Caplanne and Laurion, 2008). Climate-driven tree line movement caused changes to the content of CDOM in lakes, with paleolimnological investigations suggesting associated shifts in UVR and PAR irradiance (Pienitz and Vincent, 2000; Saulnier-Talbot et al., 2003). Additional complications to projecting limnological and ecological impacts relating to increases in both allochthonous dissolved organic matter and inorganic nutrients in Arctic lakes stem from resultant impacts on heterotrophic grazing and microbial food web processes (Vincent et al., 2012). Northern lake ecosystems are expected to become net emitters of carbon dioxide since aforementioned biogeochemical changes are likely to facilitate increases in heterotrophic grazing and microbial activity (Sobek et al., 2003; Sobek et al., 2005). Additionally, lake respiration is expected to increase at a rate that exceeds photosynthesis in the future (Vincent et al., 2012).

It is difficult to predict how lake ecosystems of northern latitudes will respond to changes in lake-surface temperature, thickness and duration of ice-cover, UVR, dissolved organic matter content, and heterotrophic grazing since changes to any given parameter can interact with simultaneous changes to other limnological parameters and processes. For example, while temperature and UVR have been identified as distinct factors likely regulating changes to phytoplankton community structure (Williamson et al., 2010), they also moderate the resultant effect of one another (Rae and Vincent, 1998; Roos and Vincent, 1998). The ultimate impact of UVR and temperature on phytoplankton community structure is, therefore, likely a combination of not only species- and organism-specific sensitivities to UVR and temperature, but also on zooplankton grazing

(Williamson et al., 2010), which in turn is also impacted by variations in UVR and temperature (Winder and Schindler, 2004).

Williamson et al. (2010) suggest that biotic factors such as zooplankton grazing are potentially significant variables to consider when investigating the effect of changing abiotic factors on phytoplankton community structure. Furthermore, ecological responses and sensitivities to abiotic factors such as UVR have been demonstrated to differ not only among taxa, but among life stages as well (Vincent and Laybourn-Parry 2008). Additionally, biotic factors such as grazing intensity may be influenced by the capacity of changing abiotic factors (e.g., lake-water temperature and UVR) to cause seasonal and/or vertical mismatches between phytoplankton and zooplankton (Winder and Schindler, 2004, between the availability of food resources (Leech and Williamson et al., 2001; Cooke et al., 2008), and between predator species (Williamson et al., 1999; Boeing et al., 2004).

Warming of the epilimnion in northern high latitude lakes - exacerbating decreases in ice-cover thickness and duration (Livingstone and Adrian, 2009; Weyhenmeyer et al., 2011) and affecting the thermocline (Winder and Schindler, 2004) - is interacting with changes to allochthonous DOC and inorganic species; the combination of these changes is likely to affect phytoplankton (Parker et al., 2008; Winder et al., 2009; Yvon-Durocher et al., 2011) with important repercussions to productivity, food web processes, as well as water quality (Scholes et al., 2014). Analyses of *in-situ* parameters measured consistently in limnological investigations over extensive spatial and temporal spans offer particularly important opportunities to investigate how specific limnological parameters have responded to climate- and anthropogenic-stressors in vulnerable northern ecosystems.

One of the most consistently measured parameters in limnological investigations, and has recurrently been the only *in-situ* biological parameter measured, is chlorophyll-*a* (Chl-*a*). Autotrophic cells and colonies, referred to as phytoplankton, are a crucial component of the base of the pelagic food chain in lakes. Cullen (1982) determined that robust agreement exists between planktonic primary production and algal production, and since Chl-*a* is comparatively easier to measure than algal biomass, Chl-*a* has served as the principal limnological variable to use as a trophic-state indicator (Cullen, 1982; Boyer et al., 2009). Chl-*a* is considered to respond to changes in limnological conditions (Wetzel, 2001) and its integration with measured concentrations of nutrients (e.g., phosphorous and nitrogen) facilitates a relatively robust assessment of the trophic status of a given freshwater lake (Mitchell and Prepas, 1990). The pervasive utilization of Chl-*a* as an index of the trophic-condition and productivity of lakes began with the acknowledgement of its suitability as an indicator of photoautotrophic biomass (algal biomass) and, thereby, primary productivity (Steele, 1962). Using measured concentrations of Chl-*a* as an estimate of algal (or phytoplankton) biomass is now ubiquitous in aquatic ecology (Vollenweider, 1968) and it began being measured in lakes as early as the 1930s.

The integration and analyses of observations and modelling exercises related to the timing and magnitude of primary production in the Arctic Ocean suggest that overall primary productivity has likely increased in response to climate warming in this region, and also trended towards earlier occurrences of peak productivity (Larsen et al., 2014). The onset of spring phytoplankton blooms are a function of changing light, hydrographic conditions, and the break-up of sea-ice (Wassman, 2011). Kahru et al. (2011) reported that earlier phytoplankton blooms were detected in approximately 11% of the area of the Arctic Ocean, and bloom events corresponded to diminished concentrations of early summer sea-ice. Additionally, for the period 1988-2007, both satellite observations and model estimates converge on the indication that increases in marine phytoplankton productivity were likely consequences of decreasing summer sea-ice extent (Zhang et al., 2010). It should be noted, however, that interceding processes related to zooplankton grazing (Olli et al., 2007) and/or the exhaustion of nutrients due to stronger stratifications and changing mixed layer depth (Wassman, 2011; Tremblay et al., 2012) may offset increases in Arctic Ocean primary production later in the year.

Permafrost thaw and degradation is expected to affect Arctic limnology primarily through processes associated with the deepening of the active later. As the active layer of the catchment area surrounding a lake deepens, previously non-existent infiltration pathways develop. As a result, accelerated geochemical weathering of subsurface materials that were formerly isolated from infiltrating water is likely to result in increased inputs of nutrients (Hobbie et al., 1999) and organic carbon into Arctic lakes, thereby increasing microbial produced carbon dioxide and methane (Walter et al., 2006; Mazéas et al., 2009; Laurion et al., 2010). An analysis of the water chemistry of 46 permafrost thaw ponds in the Canadian Arctic and subarctic revealed that permafrost thaw was accompanied by increases in the nutrient concentrations of the water (Breton, et al., 2009). Rates of geochemical weathering may be intensified further by projected increases in precipitation expected to affect the terrestrial regions of North America, Europe, and Asia (Kattsov et al., 2005). As a result, concentrations of TP and TN are expected to have increased in recent decades. Similar to nutrient levels, inputs of DOC are likely to increase with deepening active layers and increased runoff (Wrona et al., 2006b). Expected and observed increases in tundra vegetation are also likely to cause changes to DOC dynamics in Arctic catchments and DOC concentrations in Arctic lakes (White et al., 2007; Xu et al., 2013).

While certain lakes have had phytoplankton biomass sampled at regular intervals over the past thirty to fifty years, there have been no comprehensive analyses undertaken to investigate historical trends in Chl-*a* on large datasets of circumpolar Arctic lakes, spanning a vast array of northern limnological ecosystems. Broadly speaking, the compounding effects of reductions in the extent and duration of Arctic lake ice-cover, warmer ambient air- and water-temperatures, and enhanced dissolved nutrient (total phosphorous (TP) and total nitrogen (TN)) and DOC supplies derived from more biogeochemically active catchments have the potential to enhance lentic productivity in Arctic lakes (Bonilla et al., 2005; Prowse et al., 2006a; Bonilla et al., 2009; Antoniades et al., 2011). It should be noted however, paleolimnoligcal investigations indicate that any contemporary changes in diatom community structure are likely to manifest themselves differently across, and even within, disparate regions of the circumpolar Arctic (Vincent et al., 2012). Overall, trends in concentrations of open-water Chl-*a* in Arctic lakes were expected to reflect those of Arctic marine environments and experience increases.

The aim of this analysis was two-fold:

(1) By canvasing the existing bulk of available and suitable Arctic limnological data, general limnological characteristics (i.e., lake depth, altitude, dissolved oxygen, dissolved organic carbon, dissolved inorganic carbon, water temperature, conductivity, chemical cations and anions, and ice-cover characteristics) of a large and diverse a set of

Arctic lake systems were characterized and compared. The limnological systems differed in latitudinal band (i.e., 50.00-59.99, 60.00-60.99, 70.00-79.99, 80.00-89.99 ^{o}N), ACIA region, and year in which the lakes were sampled (1965-2011);

(2) Furthermore, investigations were undertaken to infer whether changes to Arctic lake primary productivity (as measured by algal biomass (Chl-*a*)) and geochemistry (TP, TN, DOC), caused by concomitant changes to Arctic climate (i.e., air temperature, precipitation) and cryosphere (i.e., ice-cover duration and thickness, permafrost melt), can be inferred through analysis of an assembled dataset of circumpolar Arctic limnological parameters.

In total, data for 1069 lake-years from 727 lakes was generated through the compilation of 52 separate academic sources, one obliged data request, and the World Lake Database. Latitude ranged from 50^{o} to 83^{o}N and the lakes were distributed primarily across the circumpolar Arctic and, to a lesser extent, the subarctic. All lake measurements were taken during the open-water seasons of the period 1965-2010.

5.2 Methods

The dataset used for the analyses was generated using three different approaches: (1) a wide spectrum of published scientific literature pertaining to northern limnological systems was examined for the inclusion of pelagic, open-water Chl-*a* as well as other key limnological (i.e., conductivity, dissolved oxygen, water-temperature, nutrient concentrations, DIC/DOC, cation and anion concentrations, turbidity) and physical (i.e., latitude, longitude, altitude, lake depth, lake area, ice-cover duration) variables; (2) data-acquisition requests were sent to various researchers and scientists inviting any available *in-situ*, open-water Chl-*a* measurements, as well as any other hydro-ecological parameters, from northern limnological systems; and (3) appropriate limnological data accessible through the World Lake Database (http://wldb.ilec.or.jp/) were added to the database.

A total of 52 separate academic sources, one accommodated data request (Westover et al., 2009), and data from 47 different lakes from the World Lake Database were used, resulting in data for 1069 lake-years' worth of *in-situ* Chl-*a* measurements (727 different lakes), as well as additional limnological and environmental observations.

To encompass as many ice-covered, high latitude lake systems of the northern hemisphere as possible, lakes situated at 50^{o}N or above were included in the analyses; however, Arctic limnological systems (latitudes ≥ 60^{o}N) remained the main focus of the investigation and subsequent analyses.

It should be noted, however, that the true southern boundary of the Arctic is not 60^{o}N, but is rather a reflection of a number of physiographic, climatic, solar, and cryospheric parameter limits. For the purpose of analyses, the Arctic is specified by rudimentary cut-off of 60^{o}N. Similarly, despite the Subarctic not truly being confined to only latitudes within 50.00 – 59.99^{o}N, it has been defined as such for the purpose of analysis. Arctic lakes represented 908 of the lake-years (596 lakes), with the remaining 160 observations (139 lakes) coming from subarctic regions (50 - 59^{o}N). Lakes were distributed across the circumpolar Northern hemisphere, with locations in 9 countries: Canada, Finland, Greenland, Lithuania, Norway, Poland, Russia, Sweden, and the United States of America.

Epilimnetic summer values (during the open-water season) of Chl-*a* were used. Due to the relatively unproductive nature of Arctic lentic systems, Chl-*a* measurements corrected for pheophytin are often below detection limits; furthermore, correcting Chl-*a* measurements for pheophytin is a relatively recent analytical practice, only beginning in the 1990s. As a result, mainly uncorrected Chl-*a* values were included in the data set. Eliminating biases stemming from particular lakes being sample disproportionately more often within a season, or amongst years, was accomplished by averaging lakes sampled multiple times in the same year. Through the examination of the residuals, Flanagan et al. (2003) determined that the inclusion of point observations of Chl-*a* did not influence the results, consequently point observations were also included in the analyses. Considering that there may be intra- and inter-annual fluctuations in limnological conditions, data may not be representative of stable conditions. However, these measured values represent an *in-situ* “snapshot” of typical summer concentrations appropriate for this kind of survey.

When provided, corresponding TN, TP, and DOC measurements were also included in the data set and averaged if needed. Although not included in these analyses, additional physical (altitude, water temperature, maximum lake depth, mean lake depth, lake area, ice-cover thickness, ice-cover duration, and secchi depth) and geochemical measurements (dissolved oxygen, conductivity, dissolved inorganic carbon, manganese, sodium, potassium, calcium, magnesium, chloride, sulphate, carbonate, aluminium, barium, copper, iron, and silica) were included in the dataset as well.

In cases where both ponds and lakes were investigated in a limnological research campaign, lakes were defined as having a mean depth of at least 2 m or deeper; any water-bodies with mean depths shallower than 2 m were excluded from analyses. All GPS coordinates were converted into Decimal Degrees if not provided in that format. If any reported values were below the detection limits of the chemical analysis, they were given a value of half the reported detection limit. Unfiltered TP and TN measurements were included since they were overwhelmingly more common in the scientific literature and in previous limnological investigations.

All statistical analyses were conducted using SigmaStat and SigmaPlot packages. All parameters were $\log_{10}$-transformed to reduce coefficients of skewness and kurtosis. *In-situ* measurements of Chl-*a*, TP, TN, and DOC (1966 – 2011) from Arctic and Subarctic lakes were plotted against time and compared. Data was then separated into latitudinal bands and re-plotted against time in to investigate the possibility of disparate responses to climate-warming based on latitude.

5.3 Results

5.3.1 Overall Temporal Trends in Chlorophyll-*a*, Total Phosphorous, Total Nitrogen, and Dissolved Organic Carbon

Temporal trends of the concentrations of open-water Chl-*a*, TP, TN, and DOC are shown in Figure 5-2, Figure 5-3, Figure 5-4, and Figure 5-5 respectively. Arctic lakes are represented by blue dots and Subarctic lakes are represented by green dots.

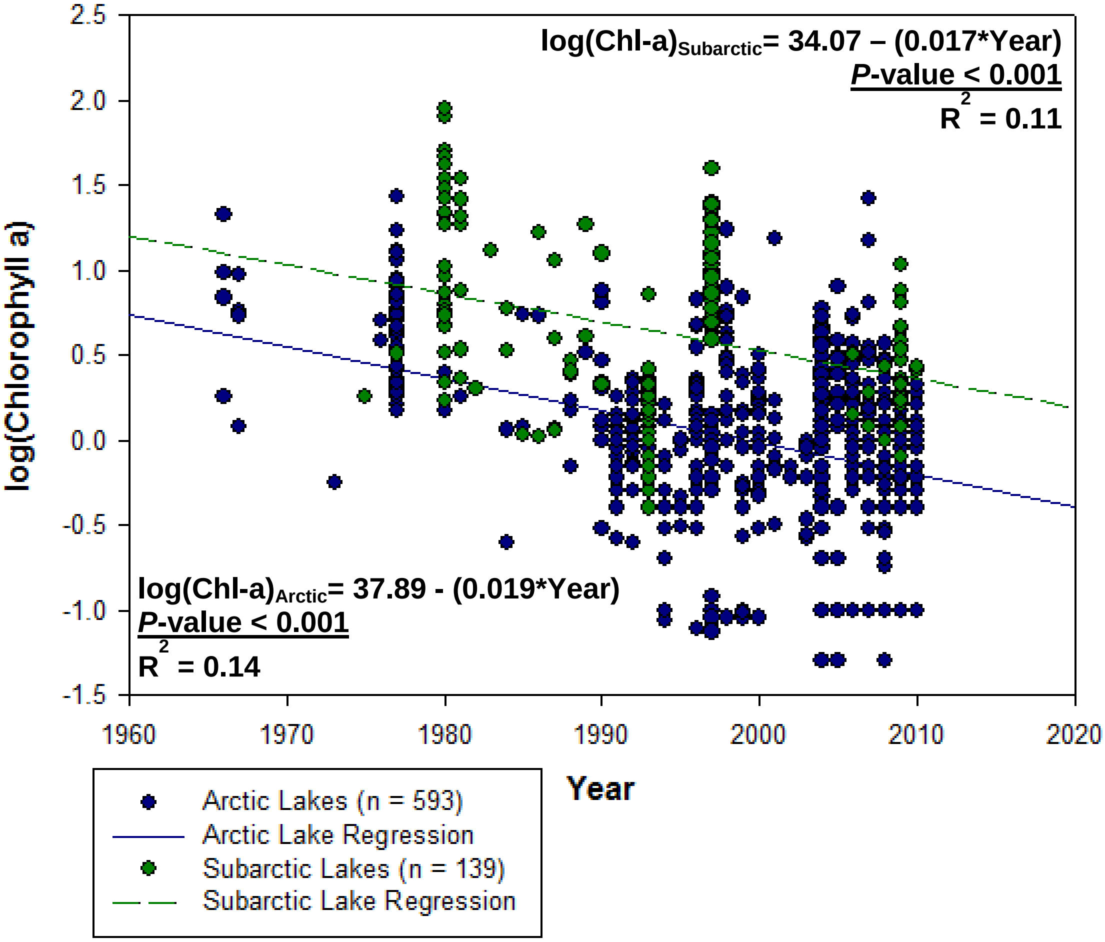

Figure 5-2: Linear trends of averaged *in-situ* observations of open-water Chl-*a* for Arctic and subarctic limnological systems for the period 1960-2010. Blue circles represent Arctic lakes and green circles represent subarctic lakes. Chl-*a* values were log-transformed, and were originally units of (µg/L). Numbers of individual lakes represented (n) are included in the legend. Significant *P*-values ($P < 0.05$) are underlined.

The average concentrations of open-water Chl-*a* decrease during the period 1965-2010 in both Arctic and Subarctic lakes (Figure 5-2). Furthermore, average values of Chl-*a* were lower in Arctic lakes than Subarctic lakes during any given year throughout the observed time period. Given the slopes, standard errors, and sample sizes of the respective regression lines, it was determined that there was insufficient evidence ($P > 0.05$) to conclude that the slopes of Arctic and Subarctic lakes were significantly different ($t = 0.0033$, d.f. = 728, $P = 0.997$). It is, therefore, possible that one or more mechanisms may be affecting both groups of lakes simultaneously, contributing to the observed decreases in algal biomass over time. Next, similar temporal analyses of key environmental drivers were undertaken to investigate the possibility that changes to these parameters may be driving observed change in algal biomass. Environmental drivers included key dissolved nutrients - total phosphorous (Figure 5-3) and total nitrogen (Figure 5-4) - and dissolved organic carbon (Figure 5-5).

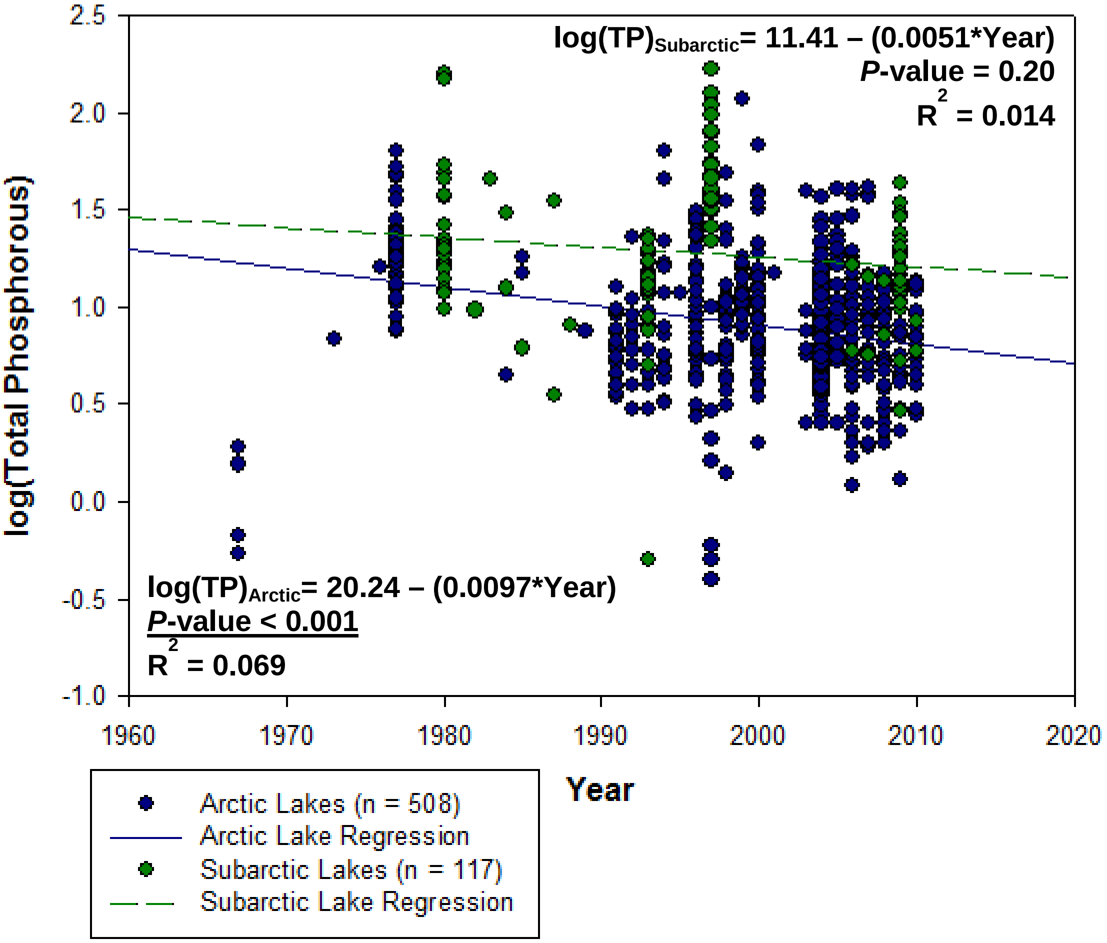

Figure 5-3: Linear trends of averaged *in-situ* observations of open-water TP for Arctic and subarctic limnological systems for the period 1960-2010. Blue circles represent Arctic lakes and green circles represent subarctic lakes. TP values were log-transformed, and were originally units of (µg/L). Numbers of individual lakes represented (n) are included in the legend. Significant *P*-values ($P < 0.05$) are underlined.

While there is no significant change in the average open-water concentrations of TP in Subarctic lakes, there is a persistent depression in the concentrations of Arctic open-water TP in Arctic lakes during the period 1965-2010 (Figure 5-3). Importantly, established coupling between TP and Chl-*a* (Flanagan et al., 2003) indicates that decreasing TP in Arctic lakes could be influencing observed depressions in algal biomass. More importantly, the lack of analogous decreases in Subarctic TP despite decreasing algal biomass suggests important latitudinal-differences in potential causal mechanisms affecting northern hemisphere lakes.

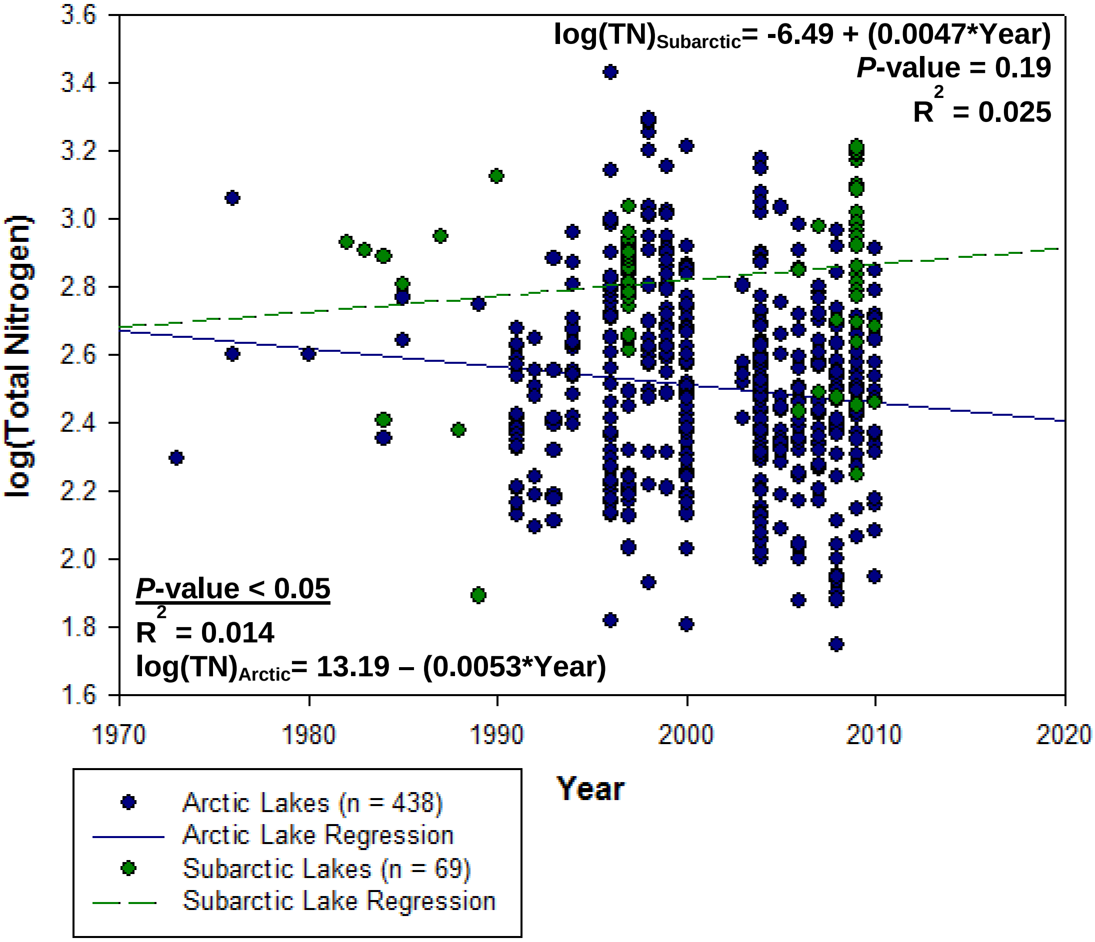

Figure 5-4: Linear trends of averaged *in-situ* observations of open-water TN for Arctic and subarctic limnological systems for the period 1970-2010. Blue circles represent Arctic lakes and green circles represent subarctic lakes. TN values were log-transformed, and were originally units of (µg/L). Numbers of individual lakes represented (n) are included in the legend. Significant *P*-values ($P < 0.05$) are underlined.

Similar to the observed trends in TP (Figure 5-3), the concentrations of average open-water TN decrease in Arctic lakes but exhibit no significant change in Subarctic lakes during the period 1970 to 2010 (Figure 5-4). Decreasing dissolved nutrient components, including both TP and TN, indicate an important link with observed decreases in Arctic lake Chl-*a*. Furthermore, given that Subarctic lakes experienced insignificant changes in concentrations of dissolved nutrients, it is evident that important differences exist between both groupings of lakes, possibly based on latitudinal differences (Flanagan et al., 2003).

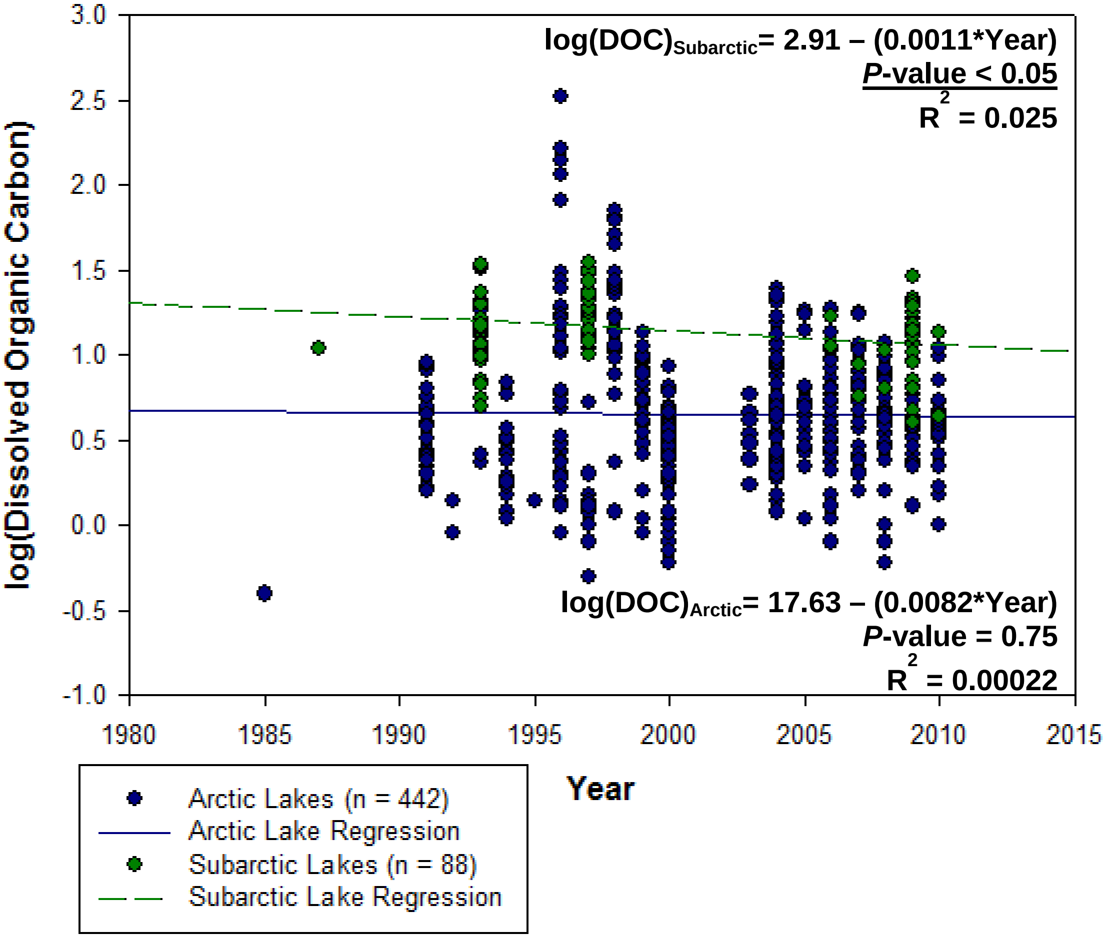

Figure 5-5: Linear trends of averaged *in-situ* observations of open-water DOC for Arctic and subarctic limnological systems for the period 1980-2010. Blue circles represent Arctic lakes and green circles represent subarctic lakes. DOC values were log-transformed, and were originally units of (mg/L). Numbers of individual lakes represented (n) are included in the legend. Significant *P*-values ($P < 0.05$) are underlined.

The concentrations of open-water DOC do not undergo significant changes in Arctic lakes during the period 1980-2010 but display a decreasing trend in Subarctic lakes (Figure 5-5). Similar to the dissolved nutrient components, there is a clear divergence between Arctic and Subarctic lakes in relation to observed trends in dissolved organic carbon. As a result, it was useful to partition limnological data into additional latitudinal sub-groupings to further investigate the possibility of disparate responses in Chl-*a*, as well as key environmental drivers, depending on the latitude of a given lake.

The following section includes plots of Chl-*a*, TP, TN, and DOC separated into latitudinal bands comprising high-Arctic (70.00-89.99 Degrees North), low-Arctic (60.00-69.99 Degrees North), and Subarctic (50.00-59.99 Degrees North) regions.

5.3.2 Latitudinal Effect on Temporal Trends in Chlorophyll-*a*, Total Phosphorous, Total Nitrogen, and Dissolved Organic Carbon

Average algal biomass of lakes located within the high-Arctic latitudinal band (70.00-89.99 Degrees North) did not undergo significant changes; conversely, lakes located within low-Arctic (60.00-69.99 Degrees North) and Subarctic latitudinal bands (50.00-59.99) did experience significant decreases (Figure 5-6). It is evident that separating limnological data into more detailed latitudinal bands generated important information pertaining to disparate responses in lakes situated within higher-Arctic verses lower-Arctic regions. Furthermore, it was necessary to separate the environmental drivers (TN, TP, and DOC) into separate latitudinal bands as well to investigate the potential for latitude-based differences in temporal trends.

Given the slopes, standard errors, and sample sizes of each regression line, it was determined that there was insufficient evidence ($P > 0.05$) to conclude that the slope of the low Arctic (60.00-69.99 Degree North) latitudinal band was significantly different from that of the Subarctic (50.00-59.99 Degree North) band ($t = 0.00080$, d.f. = 159, $P = 0.999$) (Figure 5-6). Because the slopes and intercepts of the Subarctic and low-Arctic latitudinal bands are similar, a similar mechanism may be driving the observed changes in these regions. However, it is important to note that the temporal range of the high-Arctic data (70.00-89.99 Degrees North) does not extend as far back in the historical record as that of the low-Arctic and Subarctic latitudinal bands (50.00-59.99 and 60.00-69.99 Degrees North). As a result, an additional Chl-*a* analysis was undertaken to investigate temporal trends during the period 1990-2010. This facilitated a standardization of the time period under investigation (Figure 5-10). Interestingly, given this new time-frame, there are no significant changes in average algal biomass in any of the latitudinal bands.

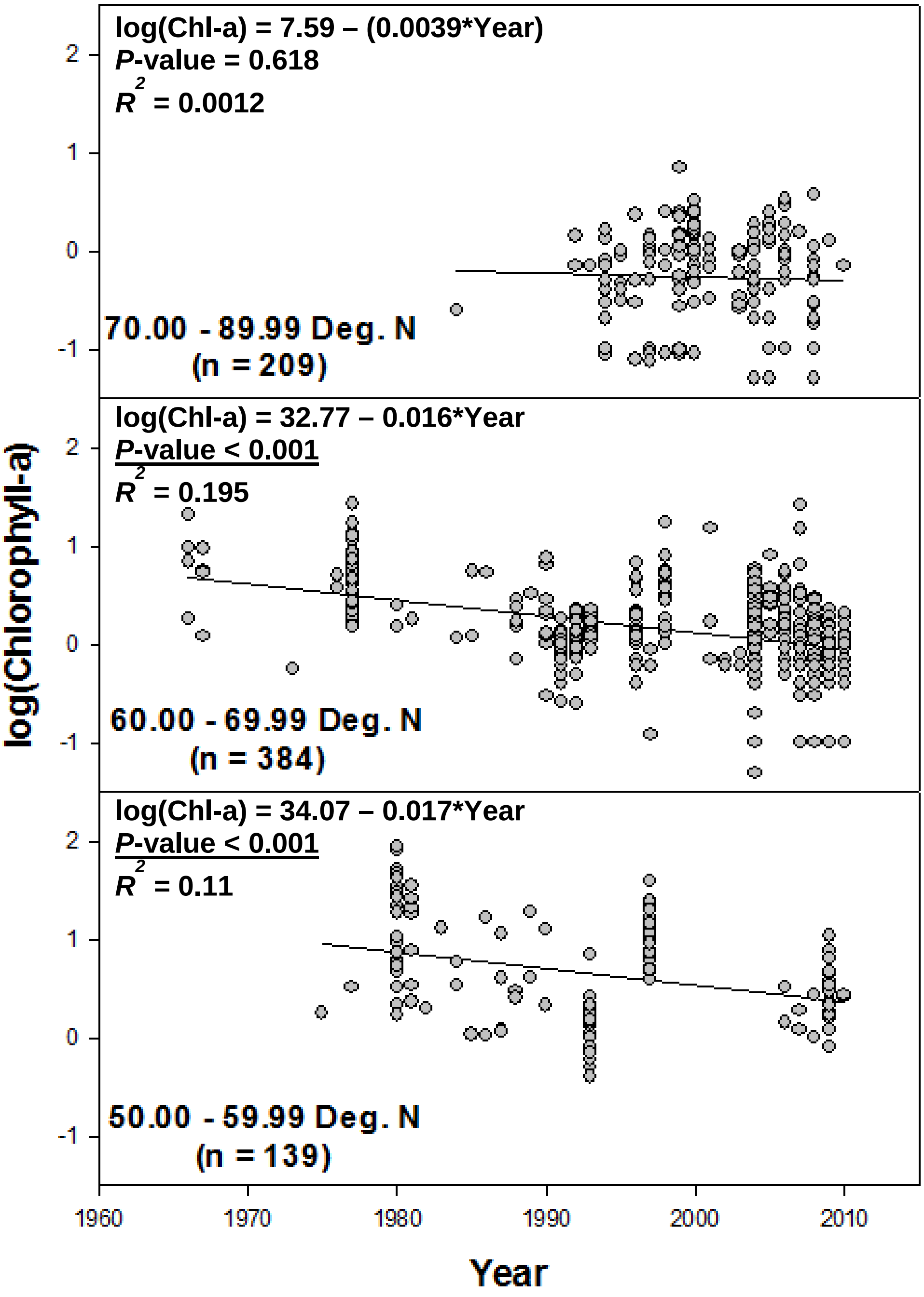

Figure 5-6: Linear trends of averaged *in-situ* observations of open-water Chl-*a* across different latitudinal bands for the period 1960-2010. Chl-*a* values were log-transformed, and originally units of (µg/L). Number of observations (n) are included under the latitudinal band (Deg. N). Significant *P*-values ($P < 0.05$) are underlined.

The separation of lakes into latitudinal bands exposed distinct and disparate trends in open-water TP depending on the band under investigation (Figure 5-7). While the Subarctic (50-59.99 Degree North) and high-Arctic (70.00-89.99 Degree North) do not exhibit changes over time, the low-Arctic (60.00-69.99 Degree North) latitudinal band exhibits a significant decreasing slope.

Similar to the latitudinal separation of Chl-*a* and TP, average open-water concentrations of TN also displayed latitude-dependent differences in temporal trends. Higher-Arctic regions (70.00-89.99 Degree North) displayed a significant decrease in TN while Subarctic and low-Arctic regions (50.00-59.99 Degree North and 60.00-69.99 Degre North) displayed insignificant changes over time (Figure 5-8). These findings, in addition to the latitudinal analysis of Chl-*a* and TP, reiterate the importance of separating lakes into latitudinal bands (Figure 5-6).

Similar to the latitudinal separation of Chl-*a*, TP, and TN – temporal trends of average open-water concentrations of DOC also differed between latitudnal bands. While the Subarctic and low-Arctic regions (50.00-59.99 Degree North and 60.00-69.99 Degree North) displayed significant decreasing trends, the higher-Arctic regions (70.00-89.99 Degree North) displayed insignificant temporal trends (Figure 5-9). Given the slopes, standard errors, and sample sizes of each regression line, it was determined that there was insufficient evidence ($P > 0.05$) to conclude that the slope of the low-Arctic latitudinal band was significantly different from that of the Subarctic latitudinal ($t = 0.0079$, d.f. = 348, $P = 0.994$) (Figure 5-9). Given that the slopes were statistically similar, perhaps a similar mechanism(s) is/are driving the trend.

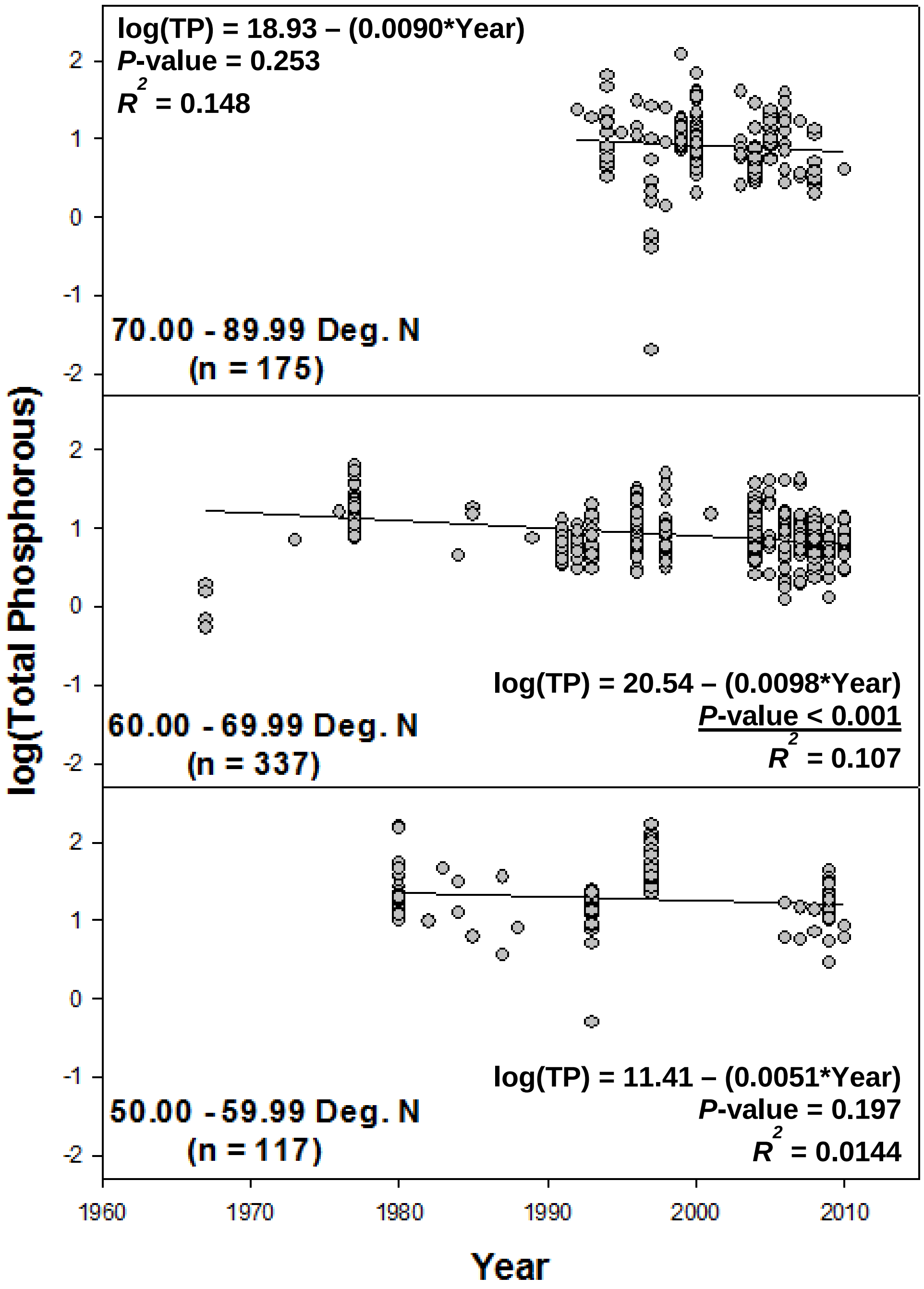

Figure 5-7: Linear trends of averaged *in-situ* observations of open-water TP across different latitudinal bands for the period 1960-2010. TP values were log-transformed, and were originally units of (μg/L). Number of observations (n) are included under the latitudinal band (Deg. N). Significant *P*-values ($P < 0.05$) are underlined.

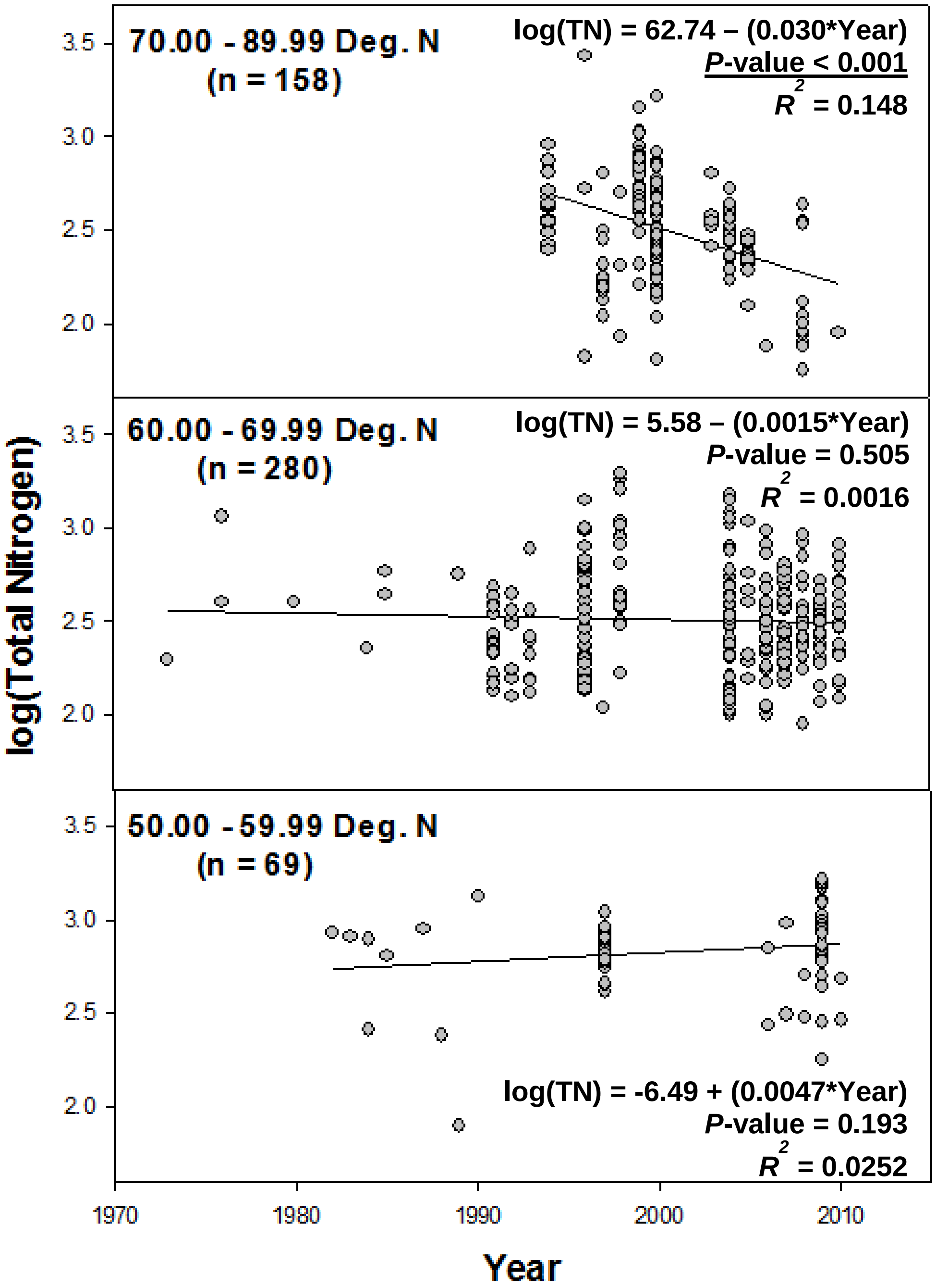

Figure 5-8: Linear trends of averaged *in-situ* observations of open-water TN across different latitudinal bands for the period 1970-2010. TN values were log-transformed, and were originally units of (µg/L). Number of observations (n) are included under the latitudinal band (Deg. N). Significant *P*-values ($P < 0.05$) are underlined.

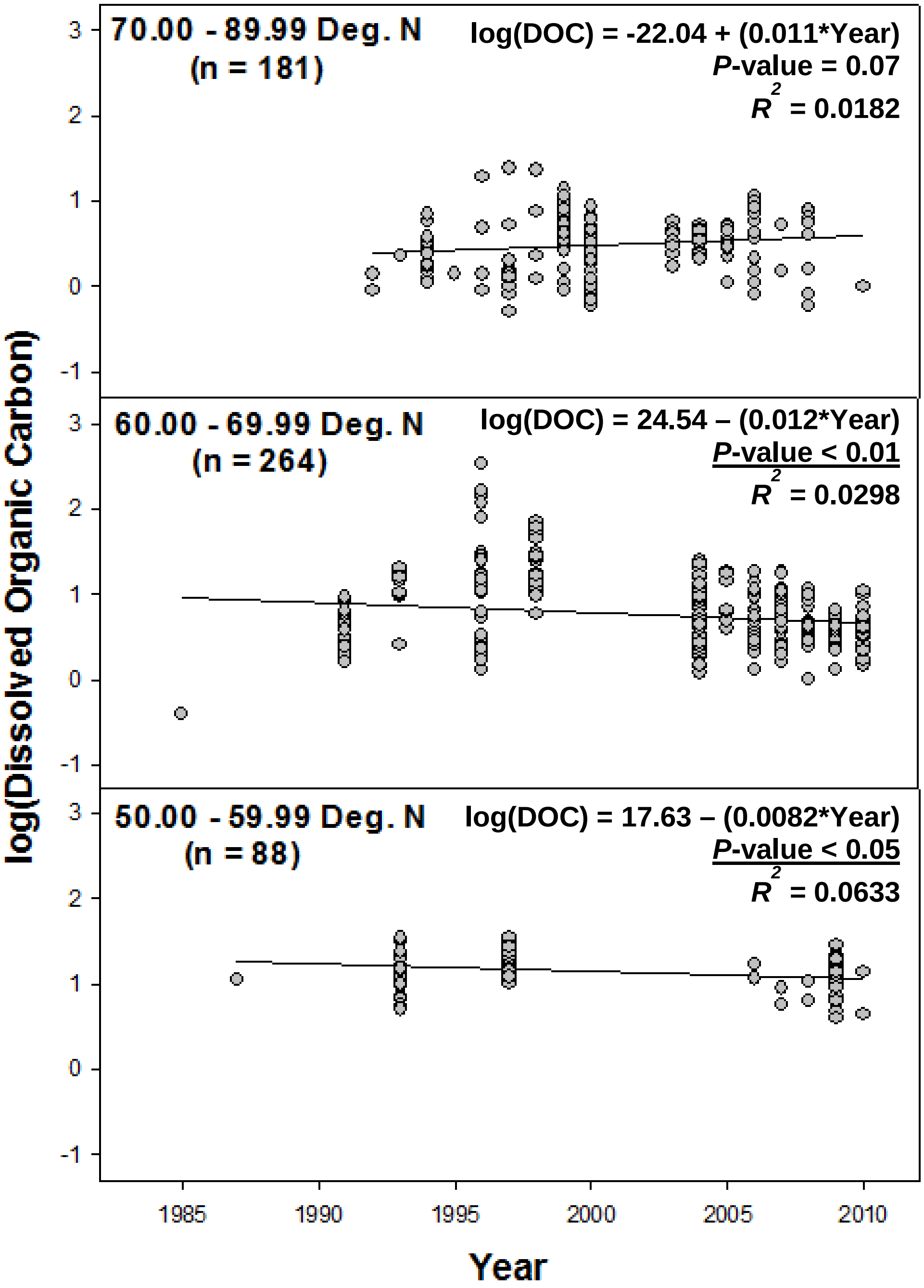

Figure 5-9: Linear trends of averaged *in-situ* observations of open-water DOC across different latitudinal bands for the period 1985-2010. DOC values were log-transformed, and were originally units of (mg/L). Number of observations (n) are included under the latitudinal band (Deg. N). Significant *P*-values ($P < 0.05$) are underlined.

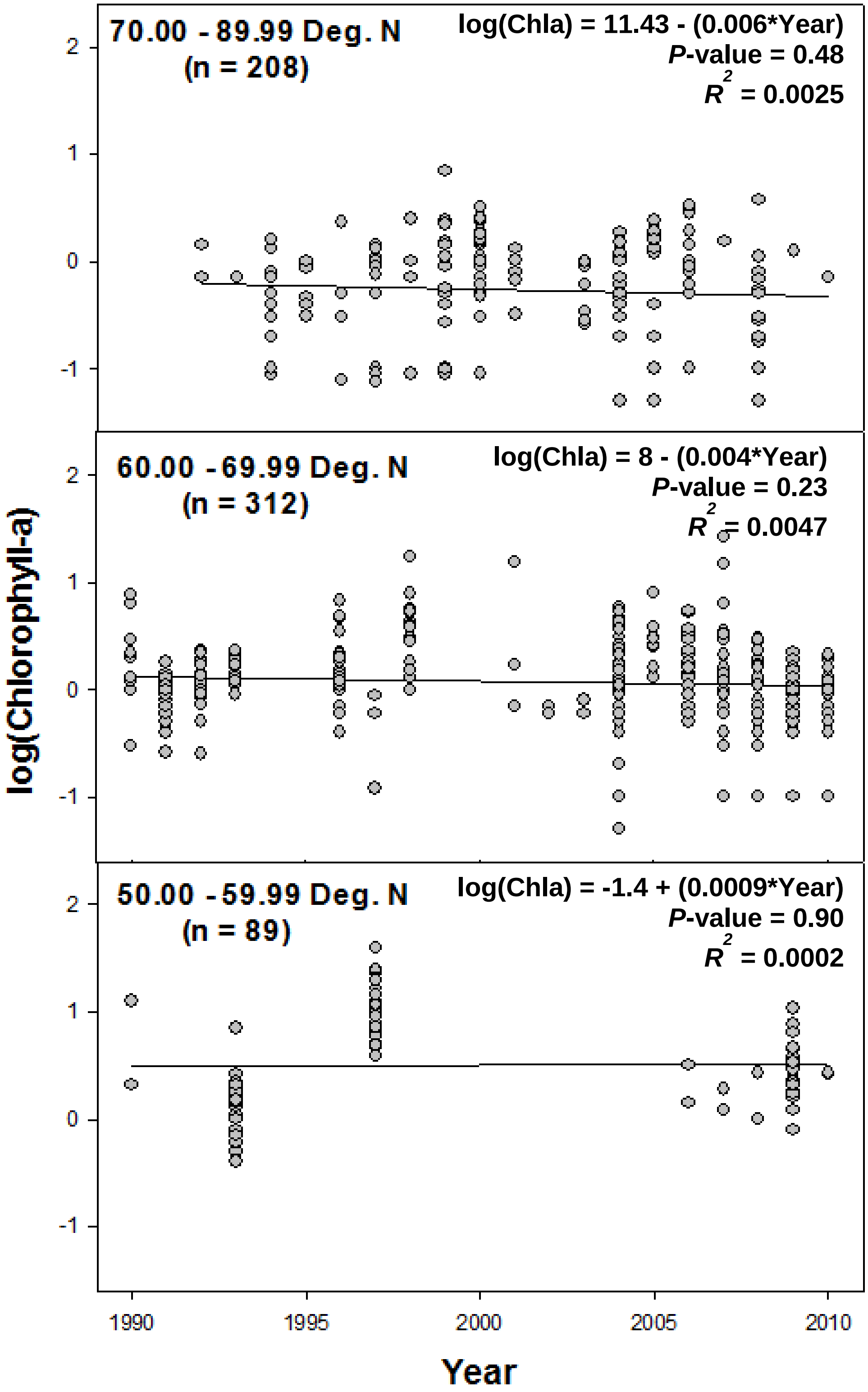

Figure 5-10: Linear trends of averaged *in-situ* observations of open-water Chl-*a* across different latitudinal bands for the period 1990-2010. Chl-*a* values were log-transformed, and were originally units of (µg/L). Number of observations (n) are included under the latitudinal band (Deg. N). Significant *P*-values ($P < 0.05$) are underlined.

5.4 Discussion

The separation of lakes into latitudinal bands was crucial to generating more refined hypotheses pertaining to the observed temporal trends in Chl-*a*, as well as key environmental drivers (TP, TN, and DOC). Firstly, it is important to note that the observed decrease in the average open-water chlorophyll-*a* concentrations of Arctic lakes (60.00-89.99 Degrees North) (Figure 5-2) were driven primarily by data collected from lake ecosystems situated within the lower-Arctic region (60.00-69.99 Degrees North) (Figure 5-6). Further latitudinal separation of limnological data demonstrated that average open-water chlorophyll-*a* concentrations of lake ecosystems situated within the higher-Arctic region (70.00-89.99 Degrees North) did not change significantly over the observed time period. Similar to the 60.00-69.99 Degrees North latitudinal band, the subarctic 50.00-59.99 Degrees North latitudinal band demonstrated a significant decrease over the observed time period. Perhaps similar mechanisms caused these concomitant depressions in Chl-*a* across both lower-Arctic latitudinal bands (50.00-59.99 and 60.00-69.99 Degrees North).

Additional complexity emerges when attempting to interpret whether temporal trends in key environmental drivers can explain the observed trends in Chl-*a*, in each latitudinal band. The importance of these latitude-based partitions becomes clear when analyzing the dissolved nutrient components of lakes in each latitudinal band. For example, given that the 50.00-59.99 and 70.00-89.99 Degrees North latitudinal bands display insignificant changes in TP, it is evident that significant decreases in average open-water TP in lakes situated within the 60.00-69.99 Degrees North latitudinal band (Figure 5-7) were the main driver of the observed decrease in Arctic lake TP (Figure 5-3).

It is interesting to note that the significant depression in Total Nitrogen observed in Arctic lakes (Figure 5-4) was driven primarily by lakes situated in the high-Arctic latitudinal band (70.00 – 89.99 Degrees North); low-Arctic and Subarctic regions reflected insignificant changes over time. More importantly, the more detailed trends of TN were masked while all Arctic lake data was grouped together; the separation of limnological data into more detailed subcategories based on latitude was fundamental to improving understanding of temporal trends. Taken together, TN and TP support the notion that not only are ecological responses to climate warming latitude-dependent, but the responses of key environmental drivers are also latitude- dependent.

The importance of latitude is further reflected in the analyses of trends in dissolved organic carbon. Similar to the analysis of TN, trends in Arctic lake DOC were masked until limnological data was separated into latitudinal bands. While Arctic lake dissolved organic carbon displayed no significant change over time (Figure 5-5), further latitude-based portioning revealed that this was a result of grouping effects with higher-Arctic limnological systems (Figure 5-9). Significant, decreasing trends observed in the low-Arctic region (60.00-69.99 Degrees North) actually matched the observed trends in Subarctic lakes, however this commonality was not discernable until the 60.00-69.99 Degrees North latitudinal band was separated from the 70.00-89.99 Degrees North latitudinal band.

All three environmental drivers exhibited distinct and disparate trends depending on the latitudinal band under consideration; this demonstrates the importance of latitude in processes governing the hydro-ecology of limnological ecosystems.

Why is the average level of algal biomass in lake systems of the low-Arctic and Subarctic (60.00-69.99 and 50.00-59.99 Degrees North) decreasing over the period 1965-2010?

There are a number of possible mechanisms that could be contributing to observed patterns of decreased autotrophy; it is important to consider that decreases in the concentration of pelagic Chl-*a* does not imply decreases in overall lake productivity. A weakness of observational and *in-situ* approaches are the multitude of confounding processes influencing the sensitivity of phytoplankton communities to climate signals including chemical processes (eutrophication, reoligotrophication, acid deposition), physical factors (geographic location, lake type), and ecological factors (trophic status). Four possible mechanisms could be responsible for the observed trends, which could be operating in isolation or in combination with each other, and with potential for interactions amongst one another.

Could decreased availability of phosphorous be contributing to declines in algal biomass?

Could the compounding effects of (a) decreased ice-cover duration, (b) reduced stratospheric ozone over Arctic regions, and (c) decreased dissolved organic matter be exposing phytoplankton communities to increased levels of UVR that exceed their current environmental tolerances?

Could changes to the stratification- and mixing-regimes of Arctic limnological systems be supressing pelagic primary production?

Could the majority of autotrophic production be shifting from the pelagic zone to benthic microbial mats in Arctic lakes?

Coupling between Chl-*a* and TP are well-established in limnology (e.g., Sakamoto, 1966; Schindler, 1977; McCauley et al., 1989; Evans et al., 1996; Elser et al., 2007; Schindler et al., 2008). A multitude of studies have reported reductions in algal biomass due to lower concentrations of lake TP (e.g., Edmondson and Lehman, 1981; Wojciechowski et al., 1988; Ruggio et al., 1998; Willén, 2001; Reynolds, 2002). More recently, Jeppesen et al. (2005) analysed 35 long-term limnological studies and concluded that lower concentrations of Chl-*a* were a consequence of reduced concentrations of lake TP. Similarly, an 11-year year study of 18 Danish lakes attributed decreases in Chl-*a* to declining lake concentrations of TP (Jeppesen et al., 2003). As rates of decline in algal biomass exceeded those of zooplankton, higher ratios of zooplankton phytoplankton indicated the potential for enhanced grazing pressure on lake algae (Jeppesen et al., 2003). Could processes associated with, and contributing to, decreases in concentrations of lake TP be causing resultant declines in Chl-*a*?

The relationship between high UVR-exposure and phytoplankton-scarcity in aquatic ecosystems has been discussed in limnology since the 1950s (e.g., Thomasson, 1956) and damage to phytoplankton communities related to increased UVR-exposure has been reported in empirical studies (Rautio and Korhola, 2002; Leavitt et al., 2003; Gareis et al., 2010) as well as experimental investigations (Belzile et al., 2001). Research suggests that earlier break-up and melting of lake-ice cover can increase UVR-exposure by orders of magnitude in the limnological systems of polar latitudes (Rautio and Korhola, 2002; Vincent et al., 2007). More importantly, while prolonged intervals of the open-water season may reduce the constraining influence of light-limitation, any resultant increases in annual rates of primary production in Arctic lakes may be offset by the aforementioned detrimental effects of UVR on phytoplankton communities (Gareis et al., 2010). While modelling exercises undertaken by Vincent et al. (2007) indicate that the loss of protective lake-ice and snow coverage can result in greater increases in UVR-exposure than the depletion of ozone in the stratosphere, it is important to consider that both processes may have contributed to significant increases in the levels of UVR affecting Arctic lentic systems.

Furthermore, the consideration of latitudinal trends in DOC (Figure 5-9) provides further evidence of potential increases in the levels of detrimental UVR affecting lake ecosystems of the lower Arctic regions (50.00-59.99 and 60.00-69.99 Degrees North). Attenuation of UVR in freshwater systems is largely controlled by concentrations of DOC, which is in turn regulated by regional climatic changes (Schindler, 1990; 1992). Williamson et al. (1996) suggest that environmental factors altering lake concentrations of DOC may be more important than stratospheric ozone depletion in regulating the UVR environments of lakes since there is a negative exponential relationship between DOC and UVR.

Although both phytoplankton cells and communities can utilize a range of photoprotection and repair strategies against increases in UVR, changes to Arctic lake algal community structure are likely (Smol, 1988; Vincent et al., 2012). Since phytoplankton photosynthetic capacity, biological composition, and nutrient-utilization are negatively affected by UVR (Harrison and Smith, 2009), perhaps increases in UVR due to the compounding effects of decreased ice-cover thickness and duration, ozone depletion, decreased concentrations of DOC, or a combination of all three, have pushed Arctic phytoplankton outside of their environmental tolerances, causing an overall decrease in pelagic algal biomass (Chl-*a*).

Warming of water column temperatures is likely in many northern lakes as both increased incident sunlight and shorter under-ice seasons contribute to an overall radiative heating of high-latitude lakes (Vincent et al., 2012). As a result of Arctic lakes being comprised of water often at 3.98 ^{o}C or less, even minute changes to lake thermal structure can have profound effects on the mixing and stratification regimes of these limnological systems (Vincent et al., 2012). For example, lakes that are cold monomictic (stratifying only in winter) may shift to dimictic (mixing and stratification during both seasons), and thermokarst ponds may endure shorter spring and fall mixing periods (Laurion et al., 2010). In several of the individual lakes monitored in the Impact of Climate Change on European Lakes (CLIME) study, Nõges et al. (2010) claim the most significant weather-related effect to be the change in the timing of the thermal stratification and extension of the summer growth period. Potential changes to the mixing- and stratification-regimes of lakes can have important implications for the thermodynamics of the surface layer, phytoplankton and zooplankton growth rates, and the exhaustion of oxygen in deep water (Sorvari et al., 2002; Vincent et al., 2012). Consequently, impacts to higher trophic organisms are probable due to increased preservation of contaminants within Arctic ecosystems (Chételat and Amyot, 2009) and the loss of oxythermal habitats for certain coldwater fish species (Jacobson et al., 2010). Stronger stratification and changes to the depth of the mixed-layer were suggested to have contributed to the exhaustion of nutrients in the Arctic Ocean, contributing to observed decreases in primary production (Wassman, 2011; Tremblay et al., 2012). Perhaps changes to mixing and stratification characteristics of Arctic lakes are altering primary production in these lentic systems.

The relatively shallow, oligotrophic, and transparent nature of Arctic limnological systems has facilitated the establishment of microbial mats dominated by cyanobacteria (Vincent and Quesada, 2012). Importantly, these microbial mats support zooplankton densities and biomass in shallow Arctic lakes despite persistent oligotrophy (O’Brien et al., 2004; Rautio and Vincent, 2006). Investigations into benthic algae are limited at the high latitudes of the northern hemisphere and information remains pautious; as a result, little if any temporal or spatial analyses or comparisons have been undertaken (Meltofte, 2013). Given the aforementioned mechanisms and processes that could be contributing to, in isolation or in combination with one another, the decline in open-water pelagic algal biomass, could a greater portion of primary production now be occurring in the microbial mats of the benthic zones?

5.5 Conclusion

This study is a temporally-integrative and spatially-comprehensive analysis of primary production in northern limnological systems over the period 1965-2010, as measured by Chl-*a*. Additionally, temporal trends in total phosphorous, total nitrogen, and dissolved organic carbon were also investigated over similar time periods. In total, data for 1069 lake-years from 727 different lakes was generated. While recent assessments of terrestrial productivity suggest that the Arctic terrestrial biosphere has greened dramatically since 1982 in relation to increases in air temperature and extended growing seasons (Xu et al., 2013), the open-water epilimnetic concentrations of Chl-*a* of lakes in these regions has decreased.

The observed trends in Arctic lake Chl-*a*, TP, TN, and DOC may provide insight in the effects of climate change on more southerly lake ecosystems. Since Arctic regions are experiencing greater rates of warming than the global average, they represent an important opportunity to track and measure changes. Further investigation into potential mechanisms responsible for decreasing concentrations of Chl-*a* in low-Arctic and Subarctic regions since 1960 may provide insight into the factors regulating algal biomass in Arctic lakes, offer direction for future research in Arctic aquatic ecosystems, and allow accurate forecasts concerning the effects of climate change on aquatic ecosystems. More specifically, additional investigations into the effects of changing lake-ice and UVR characteristics, changes to concentrations of allochthonous DOC, mixing- and stratification-regime changes, and benthic productivity are necessary.

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CHAPTER 6: MECHANISTIC MODELLING OF ALGAL BIOMASS IN HIGH-LATITUDE LIMNOLOGICAL SYSTEMS OF THE NORTHERN HEMISPHERE

Abstract

The polar amplification of climate change has rendered Arctic aquatic ecosystems particularly sensitive to shifts in climate. The compounding impacts of accelerated warming and industrial development in northern high latitudes are expected to cause a multitude of geochemical and hydro-ecological impacts to Arctic lakes, with important implications for the trophic structure and ecological interactions of these ecosystems. Nutrient loading, increased influxes of dissolved organic matter, warmer ambient air- and water-temperatures, and changing lake-ice regimes are expected to have considerable consequences on autotrophic productivity. However, there still exists substantial uncertainty in accurately projecting how autotrophic productivity in Arctic lakes will respond. The development and improvement of models capable of robustly predicting the responses of phytoplankton are essential to the stewardship and management of these ecosystems in the future. However, previous studies have overwhelmingly relied on data from temperate lakes, compromising their utility in estimating algal biomass in Arctic systems. With the understanding that factors regulating Arctic lake productivity are different from those in temperate lakes, there has been increasing interest in developing and improving mechanistic models capable of predicting Chl-*a* in Arctic lakes. To date, the limited attempts at developing mechanistic equations capable of predicting algal biomass (Chl-*a*) in Arctic lakes have not incorporated a large enough variety of limnological variables, from large enough datasets of Arctic lakes. The goal of this exercise was to improve predictive modelling of Chl-*a* in Arctic lakes by expanding the size and spatial-representation of the datasets used for analysis, testing the capacity of established parameters to predict Chl-*a* given improved datasets, and testing the capacity of novel parameters to predict Chl-*a*. Novel parameters tested include lake-ice duration, dissolved organic carbon, dissolved inorganic carbon, and conductivity. An additional goal of this exercise was to explore trade-offs between model-fit and model parsimony; given that many monitoring and stewardship initiatives operate under tight financial-, logistical-, and temporal-constraints, it is useful to explore which parameters offer the greatest predictive capacity given expenditures necessary to measure them.

Keywords: Chlorophyll-*a*, Lake-ice duration, Dissolved organic carbon, Dissolved inorganic carbon, conductivity, Predictive modelling; Lake ice cover

6.1 Introduction

Arctic aquatic ecosystems are sensitive to long-term climatic shifts (AMAP, 2011). Considerable research has emerged projecting how perturbations to the physical and geochemical properties of Arctic lakes, resulting from accelerated warming and industrial development in northern latitudes, will impact ecological interactions and trophic structure (Medeiros et al., 2012). Broadly speaking, accelerated warming in polar regions is expected to increase concentrations of nutrients and carbon in Arctic lakes through the compounding effects of (a) permafrost thaw and active-layer deepening, (b) increased precipitation and runoff during summer months, (c) warmer temperatures facilitating nutrient regeneration (Prowse et al., 2006); and (d) increased organic matter decomposition at higher temperatures (Hobbie, 1996; Mack et al., 2004). As the expansive coverage of permafrost underlying Arctic landscape continues to degrade and recede, fluxes of nutrients and ionic species into lakes are likely to increase as subsurface flow and surficial deposit-erosion are enhanced (Hinzman et al., 2005; Schindler and Smol, 2006). The cumulative effects of increased summer precipitation, enhanced nutrient regeneration, and intensifying rates of decomposition of organic matter are likely to supplement influxes of nutrients and ionic species into northern lake systems.

Nutrient-loading can negatively affect lake water quality, as well as overall ecosystem health, through a range of processes (e.g., algal blooms, anoxia-driven fish kills,) (Edmondson, 1991; Schindler et al., 2008). The development and improvement of models capable of robustly predicting the responses of phytoplankton are, therefore, essential to the stewardship and management of these ecosystems moving forward (Attayde and Bozelli, 1999; Flanagan et al., 2003). Several studies have investigated cause-and-effect mechanisms relating to the impact of nutrient additions on algal biomass through the examination of empirical relationships of ecological variables across ecosystems (e.g., Antoniades et al., 2003; Flanagan et al., 2003; Ogbebo et al., 2009a); this has proven to be a valuable approach (Ogbebo et al., 2009a). However, further investigation is necessary to account for substantial local and regional differences in physical (geology, energy), geochemical (nutrient input), and ecological conditions (food-web structure) amongst the numerous Arctic lakes (Flanagan et al., 2003; Medeiros et al., 2012) as well as the heterogeneity with which climate change and variability will impact different regions of the circumpolar Arctic.

Using measured concentrations of chlorophyll-*a* (Chl-*a*) as an estimate of algal (or phytoplankton) biomass is ubiquitous in aquatic ecology (Vollenweider, 1968). Chl-*a* is considered to respond to changes in limnological conditions (Wetzel, 2001) and its integration with measured concentrations of nutrients (e.g., phosphorous and nitrogen) facilitates a relatively robust assessment of the trophic status of a freshwater lake (Mitchell and Prepas, 1990). Additionally, concentrations of Chl-*a* and nutrients are coupled (Westover et al., 2009). For example, the relationships between concentrations of phosphorous, during both spring turnover and the open-water season, to average summer Chl-*a* concentrations are well established (Dillon and Rigler, 1974; Wetzel, 2001).

Beginning in the mid-1960s, the development of empirical models capable of estimating algal biomass in temperate lakes was facilitated by the use of linear regression equations predicting Chl-*a* based on concentrations of epilimnetic total phosphorous (TP) (Sakamoto, 1966; Schindler, 1977; McCauley et al., 1989; Evans et al., 1996; Elser et al., 2007; Schindler et al., 2008). Although corroborated by a wide spectrum of limnological investigations - including empirical studies (Dillon and Riegler, 1974), whole-lake experimental studies (Schindler, 1977), and eutrophication-management (Edmondson, 1991) - the historical reliance of these models on temperate lake data has compromised their utility in estimating algal biomass in Arctic lakes (Flanagan et al., 2003).

Flanagan et al. (2003) determined that for given increases in the concentration of phosphorous, algal biomass of Arctic lakes do not respond as significantly as temperate systems. These conclusions were corroborated by Antoniades et al. (2003), who reported that regression equations generated in analyses of temperate lake data overestimated responses of Chl-*a* to additions of TP in Arctic sites; similar difficulties were reported by Ogbebo et al. (2009a). This suggests that factors other than low nutrient-availability (e.g., low energy and temperatures, high surface albedo/low absorbed radiation, extreme discrepancies in day length, prolonged ice cover) (Medeiros et al., 2012) may be constraining primary productivity to the extent that available nutrients are not entirely exploited in Arctic lentic systems (Riegman and Mur, 1986).

Accelerated warming in northern latitudes is expected to enhance algal productivity, growth, and standing stocks as well as accelerate nutrient cycling (Flanagan et al., 2003). With the understanding that the factors regulating Arctic lake productivity are different from those in temperate lakes (Flanagan et al., 2003), there has been increasing interest in the development and improvement of mechanistic models capable of predicting Chl-*a* in Arctic lakes. Furthermore, the compounding effects of intensifying demands for natural resources, as well as improved accessibility of the region, has expanded development in the region. As a result, the need to understand both nutrient and algal responses to climatic and biogeochemical changes has become even greater.

It should be noted that while the integrated analysis of Chl-*a*, nutrients, and organic carbon affords information on the productivity of a given Arctic lake system and its catchment, Chl-*a* may not incorporate all the necessary processes contributing to total aquatic primary productivity; this is especially true in periphyton-dominated shallow, oligotrophic lakes (Westover et al., 2009). Productivity of benthic communities in Arctic lakes may be unaffected by epilimnetic nutrients and controlled by physical factors (e.g., light, temperature), thus, explaining why Bonilla et al. (2005) reported high benthic biomass despite low concentrations of epilimnetic nutrients. Differences in primary production and nutrient availability are related to compounding variation in precipitation, duration of the growing season, catchment vegetation density, and soil development (Pienitz et al., 1997a,b; Ruhland et al., 2003b).

6.1.1 Flanagan et al. (2003)

Analyses undertaken by Flanagan et al. (2003), which examined the relationship between algal biomass and a number of independent variables relating to lake location (latitude, longitude, and altitude), as well lake chemical (total phosphorous (TP), total nitrogen (TN)) and physical (maximum depth) features, was expanded. Flanagan et al. (2003) compiled a dataset comprised of 433 separate lake years' worth of data (representing 269 individual lakes), with approximately 113 of the lake-years being located north of 60^{o}N; an additional "modified dataset" was generated by averaging values from lakes sampled in multiple years of replicate lakes to eliminate potential autocorrelation between years. Identical multiple regression exercises were performed on the modified dataset. Mean epilimnetic summer values of Chl-*a*, as well point observations, were used in the analyses. Residuals of the point observations were examined prior to inclusion in the datasets to ensure they did not influence the results.

First, to assess whether or not Arctic lakes differed from temperate lakes with respect to Chl-*a* – TP relationships, Flanagan et al. (2003) grouped lakes by latitude into Arctic (≥60°N) and temperate (<60°N) sets, and subsequent regression equations were compared by testing for significant differences between intercepts and slopes. Temperate lakes were governed by the equation:

$$\log(\text{chl-}a) = -0.382 + 0.921*\log(\text{TP}) \; (r^2 = 0.28, n = 316, P < 0.05)$$

The Temperate lake equation involved slopes (but not intercepts) that were significantly different ($P < 0.05$; *t* test of regression coefficients) from the equation governing Arctic lakes:

$$\log(\text{chl-}a) = -0.418 + 0.332*\log(\text{TP}) \; (r^2 = 0.07, n = 113, P < 0.05)$$

Since there was a statistically significant difference in slopes but not intercepts, Flanagan et al. (2003) suggested that responses in average algal biomass in temperate lakes are significantly different from Arctic lakes.

Next, to determine which empirical model best predicted the variation in Chl-*a*, Flanagan et al. (2003) performed multiple regression analyses with variable selection by backward elimination on both the full and modified data sets. Higher order terms (i.e., log(TP), $(\log(\text{TP}))^2$) were also included to capture non-linearity in the relationship between Chl-*a* and the independent variables (e.g., latitude, longitude, altitude, maximum depth, TP, TN). Interestingly, after performing multiple regression with variable selection by backward elimination on both the full and modified data sets, the variables remaining in both models were latitude, log(TP), and $(\log(\text{TP}))^2$. The relationship calculated from the full data set was governed by the regression equation:

$$\log(\text{chl-}a) = 0.01 + 2.13\log(\text{TP}) - 0.38(\log(\text{TP}))^2 - 0.03\text{latitude} \; (r^2 = 0.75, n = 433, P < 0.05)$$

The relationship calculated from the modified data set was governed by the equation:

$$\log(\text{chl-}a) = 0.38 + 1.74\log(\text{TP}) - 0.28(\log(\text{TP}))^2 - 0.03\text{latitude} \; (r^2 = 0.7, n = 269, P < 0.05).$$

6.1.2 Objective of Modelling Exercise

While a significant portion of Arctic limnological studies have reported the physical, geochemical, biological, and ice characteristics of lakes, few analyses have integrated these parameters together in the context of predicting algal biomass. Furthermore, previous attempts at developing mechanistic equations capable of predicting algal biomass in Arctic lakes have not incorporated a large enough variety of limnological variables, from a large enough dataset of lakes. The goal of this exercise was to construct the best possible predictive model for Chl-*a* in Arctic lakes by expanding the dataset used for the analyses, testing the capacity of established parameters to predict Chl-*a* given the larger dataset, as well as testing the capacity of novel parameters to predict variation in algal biomass.

Given that many monitoring and stewardship initiatives operate under tight financial-, logistical-, and temporal-constraints, it is useful to know which parameters offer the greatest predictive capacity given associated fiscal and logistical expenditures necessary to measure them; an additional goal of this exercise was to explore the trade-offs between model-fit and model parsimony.

6.2 Variable Selection

A number of studies advocate for the integration of limnological parameters beyond nutrients into studies of algal biomass in Arctic lakes. Antonaides et al. (2003) attribute the absence of a statistically significant relationship between Chl-*a* and TN or TP to the fact that other factors may be controlling phytoplankton production in Arctic lakes. Similarly, a more recent study undertaken by Ogbebo et al. (2009a) reported that the temperate lake-derived P – Chl-*a* relationships were not sufficient to explain the variation in algal standing stocks in 30 lakes situated along a north-south transect on the eastern side of the Mackenzie River and within the Mackenzie River Delta. More importantly, the inclusion of additional limnological parameters - including depth, conductivity, and dissolved organic carbon (DOC) - improved the robustness of the multivariate nutrient-based model being used to predict algal biomass (Ogbebo et al., 2009a). Additionally, one of the most robust factors regulating biological activity in Arctic limnological systems is the pervasive influence of ice and snow that covers lake systems for the majority of the year (upwards of eight months) (Hobbie et al., 1999a). An important consequence of ice and snow covering the lakes is that the growing season is drastically reduced for any aquatic organisms living in these systems (Flanagan et al., 2003). These results suggest that the incorporation of additional biogeochemical parameters such as conductivity and DOC, as well as crucial physical parameters such as ice-cover duration and lake depth into analyses of algal biomass in Arctic lakes is needed (Ogbebo et al., 2009a).

6.2.1 Dissolved Organic/Inorganic Carbon

Although the importance of DOC to lake ecosystems is well-established, and the incorporation of DOC into studies investigating lake ecosystem responses to anthropogenic and natural disturbances is not a novel concept (Williamson et al., 1999), the role of DOC in lentic processes has largely been undervalued at the expense of Chl-*a* - TP relationships. As previously stated, defining lake trophic status based on the Chl-*a* - TP relationship became a central interpretive paradigm in limnological research in light of widespread validation from empirical studies (e.g., Schindler, 1977, 1978; Dillon and Rigler, 1974; Prairie et al., 1989). Recently, however, the significance of DOC to numerous ecosystem processes, especially those pertaining to disturbances, has garnered renewed interest; this has facilitated a subsequent re-emergence of attention to DOC (Williamson et al., 1999) as well as recommendations to incorporate DOC into future analyses of lake autotrophic productivity (Ogbebo et al., 2009a; Medeiros et al., 2012).

For a detailed review of the various functions and mechanisms through which DOC affects lake ecosystem processes, refer to Williamson et al. (1999) and the references therein. Broadly speaking, the significance of DOC to lentic systems can be categorized into four broad partitions: (1) the attenuation of incoming solar radiation and harmful ultraviolet radiation – given even modest levels of DOC (less than 4 mg/L), harmful effects of UV-B are limited to the surface (Schindler et al., 1990); (2) the role of DOC as an indicator of watershed characteristics (e.g., extent and type of vegetation, drainage ratio, slope) and the nature and dynamics of processes affecting a given lake (e.g., drought, forest fire, acidification, climate change); (3) the bio-availability of nutrients as well as the toxicity of contaminants; and (4) interference with the purification of water for consumption (Williamson et al, 1999). As a result, DOC data are important to ecological investigations of northern lake ecosystems (Duff et al., 1999).

Since the harmful effects of ultraviolet B-radiation (UVR-B) increase exponentially with decreases in DOC (Schindler et al., 1990), even minor declines in DOC can impose undesirable consequences on biota (Ogbebo et al., 2009a). Additionally, plankton may endure increased UV-B damage in shallow lakes as they are continually re-exposed due to regular circulation between the shallow and deep layers of the epilimnion; this is caused by wind-induced intermittent mixing in the shallow layers (Duff et al., 1999; Squires and Lesack, 2002; Ogbebo and Ochs, 2008). The multitude of effects and interactions that DOC has on lentic freshwater systems are complex and there is a need to expand paradigms of lake ecosystem analysis to include DOC in addition to nutrient and Chl-*a* data (Williamson, et al. 1999; Ogbebo et al., 2009a; Medeiros et al., 2012); this is especially important given the fact that the quantity and quality of DOC is likely to change moving forward (Ogbebo and Ochs, 2008).

A study undertaken by Medeiros et al. (2012) was designed to improve understanding of limnological relationships across key gradients of the eastern Canadian Arctic by repeating a sample regime encompassing lakes of disparate size, depth, and catchment area characteristics. Medeiros et al. (2012) detected key relationships between organically bound nutrients and Chl-*a* across all regions as evidenced by TN, TP, PON, and DOC all having statistically significant relationships with Chl-*a*. Furthermore, various studies suggest that DOC may play an important role in determining the ratio of autotrophy to heterotrophy in northern limnological systems by influencing competition scenarios between phytoplankton and bacterioplankton (e.g., Currie and Kalff, 1984; Jansson et al., 2000; Blomqvist et al., 2001; Karlsson et al., 2002). Additions of allochthonous DOC have reportedly enhanced secondary production at the expense of algal productivity and biomass (Jansson et al., 1996; Drakare et al., 2001).

6.2.2 Lake Depth

A reoccurring inference in limnological investigations is the importance of lake depth and its relationship to parameters such as TN, TP, Chl-*a*, and DOC (Hamilton et al., 2001; Westover et al., 2009). Lake depth is a landscape-specific parameter that strongly influences localized gradients in nutrients, major ions, and water temperature (Medeiros et al., 2012). Broadly speaking, deeper lakes are more dilute with respect to concentrations of nutrients and ions (Lim et al., 2005). Similarly, empirical investigations have reported inverse relationships between depth and TN in treeline lakes in the central Canadian Arctic (Rühland et al., 2003b), negative correlations between lake depth and TKN, POC, DIC, DOC, and Chl-*a* across the Canadian Arctic Archipelago (Hamilton et al., 2001), as well as negative correlations TP and depth in lakes in the western Canadian Arctic (Pienitz et al., 1997a). Similarly, Westover et al. (2009) reported inverse relationships between lake depth and TN, TP, DOC, and Chl-*a* in lakes located across mainland Nunavut and southeastern Victoria Island. There are several processes that are likely to contribute to reported negative correlations between lake depth and nutrient concentrations in Arctic lakes: (a) effects of dilution - stemming from deeper lakes being more volumous (Westover et al., 2009) and (b) effects of enhanced nutrient cycling in shallower lakes (Alexander et al., 1989). Shallow lakes derive inorganic N from lake sediments, macrophyte production in interstitial waters, and resupply in the water column via wind agitation of the sediments; conversely, deep lakes rely primarily on processes of water column regeneration and allochthonous inputs, generally leading to N-deficiencies (Alexander et al., 1989).

6.2.3 Specific Conductance

Devlin and Finkelstein (2011) cite a number of limnological studies (e.g., Ng and King, 1999; Joynt and Wolfe, 2001; Lim et al., 2001; Antoniades et al., 2004; Keatley et al., 2008a) that identify the specific conductance of lake water as being important to ecological components of lake ecosystems such as the distribution of diatom taxa. To date, few limnological investigations have explored the relationship between lake water conductance and primary production.

6.2.4 Ice Phenology

Freshwater ice covering Arctic lakes for upwards of 8 months of the year exerts a strong influence on phytoplankton growth and lake productivity (AMAP, 2011). As a result, changes to the timing, duration, extent, and overall phenology of lake-ice will have prominent and expansive impacts on the limnology of northern lake ecosystems (Vincent et al., 2008; Mueller et al., 2009; Prowse et al., 2011c). Reductions in ice-cover duration and extent are likely to affect productivity in Arctic lakes through several mechanisms: (a) warmer and longer ice-free seasons will extend the period in which biological activity can take place, facilitating a likely increase in productivity (Douglas and Smol, 1995); (b) decreased ice-cover thickness is likely to augment under-ice oxygen and algal production by increasing the amount of solar radiation able to penetrate through the ice (Prowse and Stephenson; 1986); (c) longer ice-free seasons will lengthen the stratified season of certain lakes contributing to increased mixing depths; and (d) warmer water temperatures will increase reaction rates and the metabolic activity of organisms (Wrona et al., 2006). Rühland et al. (2003) suggest that patterns of ice-cover duration ought to be incorporated into further limnological investigations. As such, average ice-cover duration was included in the analyses of algal biomass in Arctic lakes.

6.3 Dataset Generation

The dataset used for the analyses was generated using three approaches: (1) a wide spectrum of published scientific literature pertaining to northern limnological systems was examined for the inclusion of, most importantly, pelagic chlorophyll-*a* as well as other key limnological (i.e., conductivity, nutrient concentrations, DIC/DOC) and physical (i.e., latitude, longitude, lake depth, ice cover duration) variables; (2) data acquisition requests were sent to a number of researchers and scientists requesting any available *in-situ*, open-water chlorophyll-*a* measurements; and (3) appropriate data, accessible through the World Lake Database (http://wldb.ilec.or.jp/), were added to the database as well. Care was taken to ensure that the data in the re-analyses were different from the data used in the Flanagan et al. (2003) analysis.

Compared to the dataset used for the Flanagan et al. (2003) analyses, the overall number of sources and lake years represented in the dataset, as well as the circumpolar representation of the lakes characterised, were both increased. A total of 51 separate academic sources, one accommodated data request (Westover et al., 2009), and 45 lakes' worth of data from the World Lake Database were used, resulting in 953 lake-years' worth of individual observations (753 different lakes), as well as a mosaic of additional limnological and environmental observations. To encompass as many ice-covered, high latitude lake systems of the northern hemisphere as possible, lakes situated at 50°N or above were included in the analyses; 721 of the observations come from lakes located above 55°N.

The subset dataset, which will be discussed in greater detail in “6.4.3 Subset Model Selection Analysis Using Information Criterion Approaches”, consisted of identical suites of *in-situ* environmental and limnological measurements representing 484 lake-years (472 individual lakes). All observations in the subset dataset were measured in lakes located north of 55°N. It should be noted that the vast majority of the measurements are derived from Canadian lakes, with the rest being from Sweden. As a result, care must be taken not to generalize conclusions from this particular analysis across the circumpolar Arctic.

The various parameters incorporated in the assembled dataset included chlorophyll-*a* (Chl-*a*), latitude, altitude, max depth, mean depth, lake area, total phosphorous (TP), total nitrogen (TN), conductivity (COND), dissolved organic carbon (DOC), dissolved inorganic carbon (DIC), and ice cover duration (ICD). It should be noted that due to the paucity of lakes with appropriate ice-cover data, it was not included in the initial analysis of Chl-*a* predictive modelling. Instead, it will be discussed later in this section. Open-water, unfiltered Chl-*a* measurements were used in the analyses due to two factors: (1) correcting the Chl-*a* measurements of relatively unproductive Arctic lakes for pheophytin commonly renders subsequent concentrations below detection limits; (2) additionally, correcting Chl-*a* measurements for pheophytin only became common analytical practice for limnological investigations in the early 1990s. Mainly unfiltered values of total phosphorous were included in the dataset since they were overwhelmingly more commonly included in limnological investigations, with the exception of 16 Alaskan lakes located in Gates of the Arctic National Park and Preserve (LaPerriere et al., 2003), 30 lakes in the lower Mackenzie River Basin, Northwest Territories (Ogbebo et al., 2009), and 2 lakes located in the Cape Bounty region of Melville Island, Nunavut (Stewart and Lamoreaux, 2010) which reported filtered values of total phosphorous. Similarly, unfiltered measurements of Total Nitrogen were included in the datasets since they were also more common than filtered measurements.

Several variables were log-transformed including Chl-*a*, TP, TN, and DOC. Similar to Flanagan et al. (2003), higher-order terms ($(\log(\text{DOC}))^2$, $(\log(\text{DIC}))^2$, $(\log(\text{TP}))^2$, $(\log(\text{TN}))^2$) were also included to capture non-linearity in their relationship with Chl-*a*.

6.4 Methods

Three separate analyses were undertaken to examine modelling algal biomass in Arctic lakes:

(*1*) A larger, more spatially comprehensive dataset of novel high latitude northern lakes was used to validate Chl-*a* – TP relationships reported by Flanagan et al. (2003) - (*6.4.1 Model Validation*);

(*2*) Results from mixed effects modelling and backward stepwise regression were synthesized to investigate the capability of novel physical and limnological parameters to explain variation in chlorophyll-*a* - (*6.4.2 Mixed Effects Regression Analysis*); and

(*3*) Based on the parameters determined to best explain variation in Chl-*a* via mixed effects modelling and backward stepwise regression exercises, a subset of data comprised of these limnological parameters was used to perform model selection analyses using information criterion approaches - (*6.4.3 Subset Model Selection Analysis Using Information Criterion Approaches*).

6.4.1 Model Validation

Using the larger and more spatially-comprehensive circumpolar dataset, Chl-*a* – TP relationships reported by for Arctic lakes by Flanagan et al. (2003), were validated. While the dataset used in the Flanagan et al. (2003) analysis included 113 separate lake-years' worth of data from limnological systems located north of 60^{o}N, the dataset assembled for this analysis was comprised of 509 lake-years' worth of data from distinctly different limnological systems located north of 60^{o}N. Regression equations were compared by testing for significant differences in slopes and intercepts using *t*-tests of regression coefficients.

6.4.2 Mixed Effects Regression Analysis

Multiple measurements taken in the same lake, either in successive years, or more problematically in the same year, can result in errors of correlation that overtly violate the assumptions of ANOVA and regression models (Seltman, 2013). Generally, one common solution has been the use of classical “repeated measures analysis” (RMA); however, mixed effects models offer a number of advantages over these traditional methods (Seltman, 2013). Mixed models refer to any model that incorporates the use of both “fixed” and “random” effects. Fixed effects contain treatment levels that would be used again if the experiment were to be repeated (Seltman, 2013); these are the various parameters involved in the analyses. Random effects are generally considered to be a random selection from a more comprehensive suite of levels (Seltman, 2013); these are the individual lakes in the analyses.

First, while RMAs completely discard results based on any subject with missing data, mixed models are still capable of analysing the data set (Seltman, 2013); this is important given that the limnological data set is a mosaic of data taken from a number of investigations, each one including a different suite of limnological, physical, and environmental parameters. Second, mixed effects models can evaluate, intentionally and/or unintentionally, irregular spacing of repeated measurements (Seltman, 2013). Third, the analyses of mixed models are ordinarily easier to interpret than RMAs (Seltman, 2013). Last, the extension of mixed models to non-normal results is possible through the use of generalized mixed models (Seltman, 2013).

The first step was to perform a multiple mixed model regression that included all the parameters (transformed and untransformed) of the dataset. The model was governed by the equation:

$$\log(\text{chl-}a) \sim \text{latitude} + \text{altitude} + \text{max depth} + \text{mean depth} + \text{lake area} + \text{conductivity} + \log(\text{TP}) + (\log(\text{TP}))^2 + \log(\text{TN}) + (\log(\text{TN}))^2 + \log(\text{DOC}) + (\log(\text{DOC}))^2 + \log(\text{DIC}) + (\log(\text{DIC}))^2 + \text{ice cover duration}$$

Upon calculating the full mixed model regression, variance inflation factors (VIF) were scrutinized to detect multicollinearity amongst model parameters. Given that any combinations of parameters were determined to be multicollinear with each other (VIF > 10), individual regressions were performed to determine which of the multicollinear parameters explained the most variance in log(Chl-*a*). The parameter that was determined to explain comparatively less variance in log(Chl-*a*) was excluded from further regression analyses.

Once the VIFs of all parameters remaining in the mixed model analyses were <5, backward stepwise regressions were undertaken to determine which parameters did not significantly explain variation in log(Chl-*a*) and, thus, could be removed. This is similar to the method used by Flanagan et al. (2003). Once all remaining variables were significant in explaining variation in log(Chl-*a*), subsequent linear regression model fits were performed, using a multitude of different parameter combinations, to infer which combinations of parameters consistently explained the most variation in algal biomass. Combinations were ranked in order of highest marginal R^2.

6.4.3 Subset Model Selection Analysis Using Information Criterion Approaches

It is important to recognize that R^2 may not be the best representation of the “best model”. In fixed-effects modelling, the coefficient of determination (R^2) describes the proportion of variance described by the model. While marginal-R^2 describes the proportion of variance explained exclusively by fixed factors, conditional-R^2 explains the proportion of variance explained by both fixed and random factors. Although R^2 is dimensionless, and therefore facilitates valid comparisons across different datasets, it should be used with caution in model selection since R^2 favors complex models (Burham and Anderson, 2002). As a result, additional tools and methods should be incorporated in determining which parameters best explain variation in Chl-*a*.

In testing the adequacy of a variety of model combinations, the use of penalized likelihood methods (PLM) are effective in testing relative model fit while penalizing model-complexity by castigating models constructed with greater numbers of predictor variables (Seltman, 2013). Two common PLMs are the Akaike Information Criterion (AIC) and the Bayesian Information Criterion (BIC). The most significant difference between the AIC and BIC is a function of the “target model” each method attempts to select; while the goal of the AIC is to determine the best model given the dataset being scrutinized, BIC calculates the probability that different model options are the “true” model and represent the “real world” in its simplest form. Additionally, while the target model of AIC is specific to the sample size of the dataset, BIC assumes that its true model is independent of sample size. An additional important difference between the two approaches is that BIC stresses model-parsimony more rigorously than AIC by incorporating larger penalties for additional predictor variables.

A crucial element of PLMs is that AIC and BIC values are not comparable between models that used different datasets to perform their calculations. A subset of data with lakes that included data from the parameters determined to explain the most variation in Chl-*a* in the Mixed Effects Regression Analysis (refer to previous section) was compiled. Given a suite of models estimated from the same dataset, the lowest values of AIC and BIC would be the preferred model. However, a difference of less than 2 in AIC or BIC values can be considered insignificant; additionally, comprehensive deductions based on these exercises must be avoided (Seltman, 2013). Burnham and Anderson (2004) suggest that instead of simply “selecting the true model”, researchers should be making inferences into the likely causes of similarities and differences between a range of models, determined to be valuable by AIC and BIC analyses.

AIC and BIC have been demonstrated to be unique and robust in their approaches to model comparison. While they are typically vigorous in their approximations of model adequacy, they are limited in effectiveness given certain conditions (Burnham and Anderson, 2004; Kuha, 2004). As a result, it is recommended that incorporating both criteria into model selection exercises, in addition to other indices of model fit, is useful practice. It is not uncommon for the best model selected by BIC and AIC to be different from one another (Burnham and Anderson, 2004), however much insight for model selection can be gained from investigating when they agree and when they disagree (Kuha, 2004). In particular, while convergence between both criteria can reaffirm the robustness of the model selection, disagreement can provide tentative boundaries for a set of prospective models, guiding further investigation (Kuha, 2004).

Theoretical results suggest that certain conditions favour the use of either AIC or BIC; dissimilar situations contribute to varying conclusions, and ultimately the full complexity of the real world is difficult to encapsulate by either method (Kuha, 2004; Burnham and Anderson, 2004). Since lake productivity is a collection of several effects, operating independently and simultaneously, changing over time, space, and across individuals, the prediction-based AIC might be the most appropriate. However, given the financial constraints affecting research and community-based monitoring programs in northern latitudes, the parsimonious nature of BIC can reveal the fewest number of parameters necessary to explain a significant portion of variability in algal biomass (Chl-*a*). As such, a cautious approach was taken to utilize valuable insights afforded from both methods.

6.5 Results

6.5.1 Model Validation

The TP – Chl-*a* relationship reported by Flanagan et al. (2003) was governed by the equation:

$$\log(\text{Chl-}a) = -0.418 + 0.332*\log(\text{TP}) \quad (r^2 = 0.007,\ n = 113,\ P < 0.05)$$

The relationship reported by the new dataset was governed by the equation:

$$\log(\text{Chl-}a) = -0.415 + 0.480*\log(\text{TP}) \quad (r^2 = 0.116,\ n = 509,\ P < 0.05)$$

Both relationships are displayed in (Figure 6-1). Using *t*-tests of regression coefficients, it was determined that there was not sufficient evidence ($P > 0.05$) to conclude that either the slope or intercept of the new equation differed significantly from those reported by Flanagan et al. (2003).

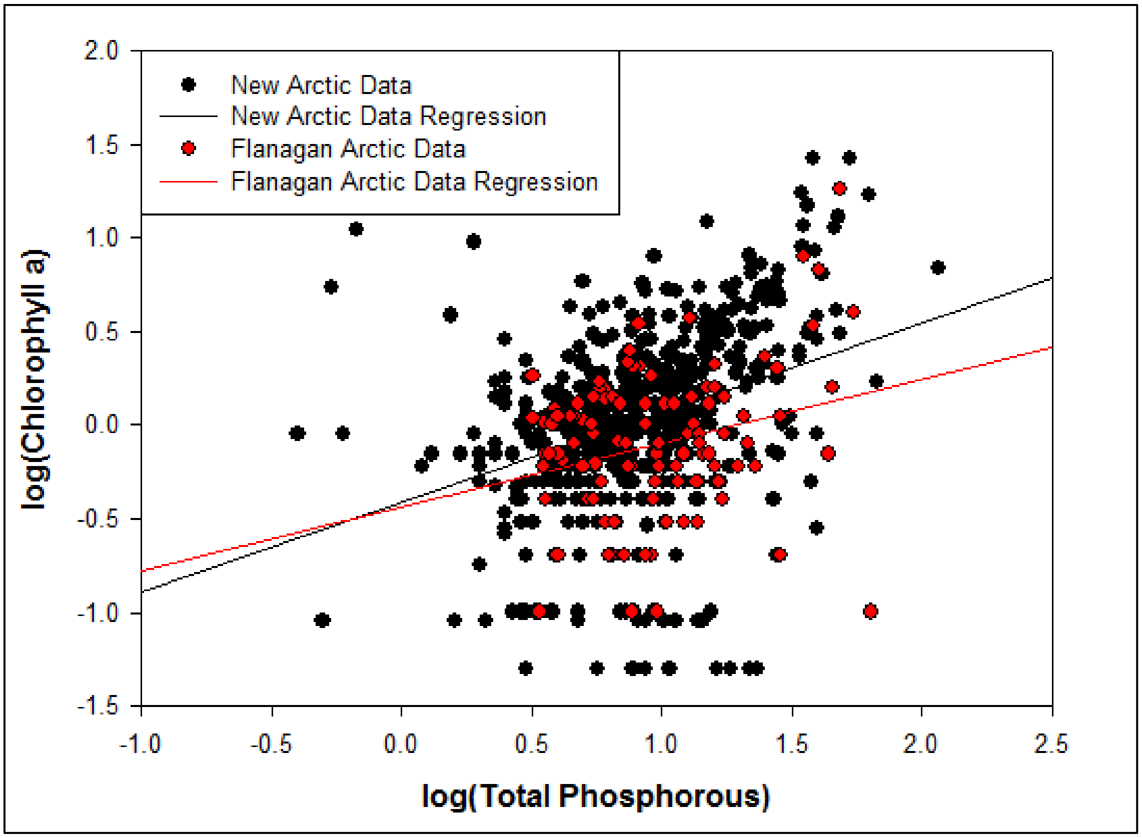

Figure 6-1: The Chl-*a* – TP (chlorophyll *a* – total phosphorous) relationship for Arctic systems. Black circles represent lakes from the new dataset and red circles represent Arctic lakes from the Flanagan et al. (2003) dataset. The difference in slope is not statistically significant.

6.5.2 Mixed Effects Regression Analysis

Upon comparing the VIFs of all model parameters, log(TN) was determined to be multicollinear with $(\log(TN))^2$, and log(DOC) was determined to be multicollinear with $(\log(DOC))^2$. Since log(TN) and $(\log(DOC))^2$ both had weaker predictive capacities than the parameter they shared multicollinearity with, they were eliminated from the model. Exclusion of variables from further analysis was corroborated by the visual analysis of partial regressions with added variable plots. The first parameters eliminated in backwards stepwise regression were ice cover duration, altitude, and max depth. Due to the initial intention of exploring the capacity of ice cover duration to explain variance in algal biomass, ice cover duration will be discussed further in “6.6.1 Lake Ice Duration**”**. Further regressions could not be undertaken with logDIC or $(\log(DIC))^2$ included in the model due to an unsatisfactory ratio of observations to model parameters. When each parameter was tested in single regressions, they did not significantly explain variation in log(Chl-*a*) (P-value < 0.05); as a result, logDIC and $(\log(\mathrm{DIC}))^2$ were removed from further analyses. Specific conductivity was the next variable determined to be insignificant in predicting variation in log(Chl-*a*) and was removed. The parameters remaining in the model were:

$$\log(\text{Chl-}a) \sim \text{latitude} + \log(\mathrm{TP}) + (\log(\mathrm{TP}))^2\ (\log(\mathrm{TN}))^2 + \log(\mathrm{DOC}).$$

Results of the regression analyses are included in Table 6-1. Parameters highlighted in red for a given model indicate that the variable was determined to be insignificant (P-value < 0.05) in explaining variation of log(Chl-*a*).

Table 6-1: Model combinations and results in order of decreasing Conditional-R^2 (Cond. R^2). Marginal-R^2 (Marg. R^2), number of observations (# of Obs.), and number of lakes represented (# of Lakes).

Lat.	**log(TP)**	**log(DOC)**	**$(\log(\mathrm{TP}))^2$**	**$(\log(\mathrm{TN}))^2$**	**Marg. R^2**	**Cond. R^2**	**# of Obs.**	**# of Lakes**
●	●			●	0.415	**0.836**	586	556
●	●	●		●	0.374	**0.821**	483	472
●	●		●	●	0.434	**0.788**	586	556
				●	0.072	**0.755**	593	559
●					0.325	**0.738**	953	795
●				●	0.355	**0.733**	593	559
●			●	●	0.427	**0.730**	586	556
●	●		●		0.556	**0.720**	783	719
	●	●	●	●	0.292	**0.718**	483	472
●		●		●	0.327	**0.701**	483	472
●	●				0.507	**0.697**	783	719
●	●	●	●	●	0.397	**0.696**	483	472
	●				0.277	**0.676**	783	719
●		●	●	●	0.391	**0.582**	483	472
			●		0.19	**0.506**	783	719
●	●	●	●		0.356	**0.465**	542	522
●	●	●			0.328	**0.464**	542	522
●		●	●		0.351	**0.450**	542	522
		●			0.197	**0.424**	544	524
●		●			0.264	**0.396**	544	524

Using Conditional-R^2 as the metric to evaluate model suitability, latitude and dissolved nutrients consistently explain a greater portion of the variance than dissolved organic carbon. However, as discussed in previous section, although R^2 is dimensionless and enables effective comparisons between diverse datasets, it should be used with caution in model selection since R^2 favors complex models (Burham and Anderson, 2002). Additionally, parsimonious models are preferable to management and monitoring initiatives operating under limited financial resources since measuring few crucial parameters is more cost- and time-effective than sampling a multitude of different parameters. As a result, subsequent analyses were undertaken to account for issues arising from disparate datasets used for each model and unduly favoring complex models.

6.5.3 Subset Model Selection Analysis Using Information Criterion Approaches

Results of the AIC subset model selection are included in Table 6-2. Parameters highlighted in red for a given model indicate that the variable was determined to be insignificant ($P\text{-value} < 0.05$) in explaining variation of log(Chl-*a*).

Table 6-2: Model combinations and results in order of increasing AIC (Akaike Information Criterion) values. Number of observations for each model is 483. Number of lakes represented in each model is 472. Conditional-R^2 (Cond. R^2), Marginal-R^2 (Marg. R^2).

Latitude	**log(TP)**	**log(DOC)**	**(log(TP))^2**	**(log(TN))^2**	**Marg. R^2**	**Cond. R^2**	**AIC**
●	●	●	●		0.391	0.671	1360.109
●	●		●	●	0.395	0.700	1361.574
●	●		●		0.380	0.664	1362.444
●		●	●		0.382	0.561	1362.616
●			●	●	0.389	0.555	1362.753
●	●	●	●	●	0.397	0.696	1365.449
●		●	●	●	0.391	0.582	1365.742
●	●			●	0.371	0.824	1369.307
●	●	●			0.364	0.786	1370.904
●			●		0.364	0.499	1370.973
●	●	●		●	0.374	0.821	1372.837
●	●				0.346	0.780	1376.920
●		●		●	0.327	0.701	1403.277
●				●	0.318	0.679	1404.102
●		●			0.290	0.644	1417.699
	●	●	●		0.290	0.703	1420.443
		●	●		0.283	0.622	1420.845
	●	●			0.267	0.794	1426.262
		●	●	●	0.286	0.643	1428.097
	●	●	●	●	0.292	0.718	1428.755
	●	●		●	0.273	0.819	1432.551
	●		●	●	0.255	0.739	1445.506
			●	●	0.244	0.628	1447.967
	●			●	0.230	0.827	1449.889
	●		●		0.224	0.707	1453.972
		●		●	0.228	0.724	1455.562
		●			0.206	0.672	1459.785
	●				0.184	0.784	1466.107
●					0.202	0.654	1466.168
				●	0.166	0.751	1477.096
			●		0.145	0.471	1494.757

Latitude is clearly one of the most important predictors of algal biomass, as are the dissolved nutrient components (e.g., TP and TN). Importantly, results suggest that phosphorous explains more variation in log(Chl-*a*) than nitrogen. Remarkably, including both log(TP) and (log(TP))^2 in the same model does not create multicollinearity, nor does it render one of the phosphorous-based predictor variables redundant to the model; this further emphasizes the important of TP to Arctic lake ecosystems, and advocates for the inclusion of TP measurement in any limnological investigations. In accordance with initial hypotheses, dissolved organic carbon appears to be useful in predicting algal biomass in Arctic lakes.

Results of the BIC subset model selection are included in Table 6-3. Parameters highlighted in red for a given model indicate that the variable was determined to be insignificant (P-value < 0.05) in explaining variation of log(Chl-*a*).

Table 6-3: Model combinations and results in order of increasing BIC (Bayesian Information Criterion) values. Number of observations for each model is 483. Number of lakes represented in each model is 472. Conditional-R^2 (Cond. R^2), Marginal-R^2 (Marg. R^2).

Latitude	**log(TP)**	**log(DOC)**	**$(\log(TP))^2$**	**$(\log(TN))^2$**	**Marg. R^2**	**Cond. R^2**	**BIC**
●	●		●		0.380	0.664	1387.474
●		●	●		0.283	0.561	1387.647
●			●	●	0.389	0.555	1387.784
●	●	●	●		0.391	0.671	1389.297
●	●		●	●	0.395	0.700	1390.761
●			●		0.364	0.499	1391.842
●	●			●	0.371	0.824	1394.337
●		●	●	●	0.391	0.582	1394.929
●	●	●			0.364	0.786	1395.934
●	●				0.346	0.780	1397.789
●	●	●	●	●	0.397	0.696	1398.790
●	●	●		●	0.374	0.821	1402.024
●				●	0.318	0.679	1424.971
●		●		●	0.327	0.701	1428.307
●		●			0.290	0.644	1438.568
		●	●		0.283	0.622	1441.714
	●	●	●		0.290	0.703	1445.473
	●	●			0.267	0.794	1447.131
		●	●	●	0.286	0.643	1453.127
	●	●		●	0.273	0.819	1457.581
	●	●	●	●	0.292	0.718	1457.943
			●	●	0.244	0.628	1468.836
	●		●	●	0.255	0.739	1470.536
	●			●	0.230	0.827	1470.758
	●		●		0.224	0.707	1474.841
		●		●	0.228	0.724	1476.431
		●			0.206	0.672	1476.489
	●				0.184	0.784	1482.810
●					0.202	0.654	1482.871
				●	0.166	0.751	1493.799
			●		0.145	0.471	1511.461

Analogous to the results generated with the AIC, latitude was an essential forecaster of algal biomass, as were the dissolved nutrient components. Similarly, it appears that phosphorous explains more variation in log(Chl-*a*) than nitrogen. Again, the inclusion of both log(TP) and $(\log(TP))^2$ in the same models does not create multicollinearity, nor does it render one of the phosphorous-based predictor variables redundant to the model. Similar to the results of the AIC approach, the addition of dissolved organic carbon appears to be an additional explanatory variable in predicting algal biomass in Arctic lakes using the BIC methodology.

6.6 Discussion

Flanagan et al. (2003) suggested that the factors and processes regulating primary productivity in Arctic lakes differed from those in temperate lakes, and that additional, Arctic-focused limnological investigations were necessary to better understand factors regulating Arctic lake primary productivity. More specifically, temperate lake derived Chl-*a* – TP relationships have been reported to inaccurately predict algal biomass in Arctic lakes; consequently several studies have suggested that additional variables related to Arctic limnology need to be included in analyses of algal biomass. These included dissolved organic/inorganic carbon (Williamson et al., 1999; Ogbebo et al., 2009a; Medeiros et al., 2012), lake depth, specific conductivity (Ogbebo et al., 2009a), and lake-ice duration. In addition to incorporating these parameters (e.g., dissolved organic carbon, dissolved inorganic carbon, conductivity, ice phenology, and lake area), analyses were undertaken on larger, more spatially-comprehensive datasets of Arctic limnological parameters. While these analyses did not produce sufficient evidence to conclude that lake depth, lake area, and conductivity were significant in explaining variation in Chl-*a*, dissolved organic carbon proved to have an important role in explaining variation. The capacity of dissolved inorganic carbon to explain variation in Chl-*a* was inconclusive due to an insufficient number of observations. The role of ice cover duration was also inconclusive, but will be discussed further in section “6.6.1 Ice Cover Duration”.

The five “best” models, as predicted by the AIC and BIC methods, were identical; however, the associated rankings of the five “best” models differed between the AIC and BIC. For both approaches, the five models include two models being comprised of 4 predictor-variables and three models constructed of 3 predictor-variables. Accordingly, since the BIC approach stresses model-parsimony more rigorously, it is unsurprising that the models constructed with only 3 predictor-variables were ranked highest. Alternatively, the AIC approach favoured the 2 models constructed with 4 predictor-variables. It is interesting to note that if the relative rankings of the 3 predictor-variable models and 4 predictor-variable models are considered separately, the AIC and BIC rankings of the two groups are identical.

The five "best" models, as determined by AIC and BIC approaches, were re-ranked based on the sum of each model's respective AIC and BIC values (Table 6-4).

Table 6-4: Model combinations and results in order of increasing sum of AIC and BIC. Number of observations for each model = 483. Number of lakes represented in each model is 472. Conditional-R^2 (Cond. R^2), Marginal-R^2 (Marg. R^2), sum of Akaike Information Criterion Value and Bayesian Information Criterion Value (AIC + BIC).

Latitude	log(TP)	log(DOC)	$(\log(TP))^2$	$(\log(TN))^2$	Marg. R^2	Cond. R^2	AIC + BIC
●	●	●	●		0.391	0.671	2749.406
●	●		●		0.380	0.664	2749.918
●		●	●		0.382	0.561	2750.263
●			●	●	0.389	0.555	2750.537
●	●		●	●	0.395	0.700	2752.335

1. $\log(\text{Chl-}a) = 3.370669 - 0.064444*\text{Latitude} + 0.190506*\log(TP) + 0.161873*\log(DOC) + 0.066848*(\log(TP))^2$

2. $\log(\text{Chl-}a) = 4.069792 - 0.072913*\text{Latitude} + 0.243449*\log(TP) + 0.072511*(\log(TP))^2$

3. $\log(\text{Chl-}a) = 3.49737 - 0.063717*\text{Latitude} + 0.206551*\log(DOC) + 0.095127*(\log(TP))^2$

4. $\log(\text{Chl-}a) = 3.348883 - 0.070479*\text{Latitude} + 0.087485*(\log(TP))^2 + 0.028245(\log(TN))^2$

5. $\log(\text{Chl-}a) = 3.258895 - 0.069802*\text{Latitude} + 0.177412*\log(TP) + 0.062291*(\log(TP))^2 + 0.022682(\log(TN))^2$

According to this metric, the "best" model includes latitude, log(TP), log(DOC), and $(\log(TP))^2$. However, the differences in the sum of AIC and BIC values between all five models are marginal and, thus, additional considerations were included in selecting the most preferable model. For example, the third-ranked model is one of the most parsimonious models and is of considerable use to monitoring initiatives operating under tight logistical and financial constraints. Furthermore, only two actual values need measurement: (1) latitude can be determined relatively easily, either with the use of a handheld GPS device or free computer program such as Google Earth; and (2) TP can be determined analytically, and subsequent transformations to log(TP) and $(\log(TP))^2$ can be accomplished for little additional logistical and/or financial cost. Additionally, the fifth-ranked model has the highest values for both marginal- and conditional-R^2; considering it also includes two transformed iterations of TP values, as well as TN, which is generally included in the same suite of analyses required to determine TP, it should be considered in future monitoring initiatives.

The most robust predictors of algal biomass in northern high latitude limnological systems are latitude and TP. It is interesting to note that TP is such a strong predictor of Chl-*a* that both log(TP) and $(\log(TP))^2$ each explain significant variation in log(Chl-*a*) without being multicollinear with one another. Similarly, Flanagan et al. (2003) reported robust models involving both log(TP) and $(\log(TP))^2$. This suggests that for any stewardship, management, or research initiative, determining the exact latitude of the lake system and measuring TP are valuable pieces of limnological information. Additionally, their establishment offer significant parameter baseline conditions to which future change can be compared.

Results from these analyses suggest that DOC does in fact play a significant role in the regulation and prediction of algal biomass in Arctic lakes. Firstly, it was selected as one of the parameters that significantly explained variance in log(Chl-*a*) in 6.5.2 Mixed Effects Regression Analysis, and secondly, was included in some of the most robust models as predicted by both AIC and BIC procedures in the 6.5.3 Subset Model Selection Analysis Using Information Criterion Approaches. Recently, DOC has been suggested to impose an important influence on the ratio of net autotrophy and heterotrophy in lake systems (Currie and Kalff, 1984; Jansson et al., 2000; Blomqvist et al., 2001; Karlsson et al., 2002). Given the multitude of complex effects that DOC has on freshwater systems, in addition to the fact that the quantity and quality of DOC is likely to change (Ogbebo and Ochs, 2008), there is a need to expand paradigms of lake-ecosystem analysis to include DOC (Williamson, 1999; Ogbebo et al., 2009a; Medeiros et al., 2012). DOC needs to be more regularly measured and recorded in limnological investigations and stewardships initiatives.

As stated previously, it is important to evade proposing definitive conclusions based on AIC and BIC results; rather than selecting a “best model”, it is more useful to make inferences into the likely causes of parallels and differences between the assortment of models determined to be valuable by AIC and BIC analyses (Burnham and Anderson, 2004). It is evident that the inclusion of latitude and total phosphorous, in both log and log-squared form, are of immense value to the mechanistic modelling of algal biomass in Arctic lakes. Similarly, recommendations to integrate dissolved organic carbon into modelling exercises were well-founded.

6.6.1 Lake-Ice Duration

The results of the backwards stepwise regression analyses suggested that there was insufficient empirical field measurements to investigate whether lake-ice duration significantly explained variability in algal biomass, given the other parameters in the model. However, this does not suggest that lake ice duration has no capacity to explain variation in Arctic lake algal biomass. In total, 348 observations of lake ice duration were incorporated into the dataset. When the 348 observations of lake ice duration were tested as the independent variables in a simple linear regression with Chl-*a*, 11.2% of the variation in Chl-*a* was explained by lake-ice duration data; more importantly, the model was significant (*P*-value < 0.001). The inability of lake-ice duration to explain significant variation in Chl-*a* in mixed model analyses is an indictment of the paucity, reliability, and quality of specific lake-ice data characterizing Arctic lakes. Numerous issues related to the collection, derivation, and availability of lake ice duration data may have limited the suitability of this data for the analyses undertaken. Daily, monthly, and/or annual records of lake ice duration are extremely rare for Arctic lakes due to a legacy of sparse empirical datasets on which analyses can be undertaken, limited lakes with long term records, discontinued records (Prowse et al., 2011a, b), and methodological inconsistencies (Beltaos and Prowse, 2009); furthermore, corresponding *in-situ* measurements of Chl-*a*, taken during the open-water but especially the under-ice season, are even more uncommon.

Rather than actual recorded measurements of lake-ice duration corresponding to the years of a given sampling campaign, researchers commonly group lakes into identical, expansive ranges of possible lake-ice regimes. These approaches neglect important temporal and regional differences in climate, ignore lake-specific parameters that cause even particularly proximal lakes to have drastically different lake ice phenologies, and do not meaningfully contribute to establishing a network of reliable lake ice data on which analyses can be undertaken. For example: Rühland et al. (2003) stated that 56 lakes in the Central Canadian Arctic Treeline Region have average open-water periods ranging anywhere from 90 to 120 days; Lim et al. (2003) list all 15 Bathurst Island, NU lakes sampled in their investigation as having lake-ice regimes that undergo freezeup in late August/early September and begin to thaw in late June; and Ostrofsky and Rigler (1987) list 49 subarctic lakes as having lake ice regimes that melt in mid-May and freeze sometime after fall turnover in September. It is unsurprising that lake-ice duration, as it was incorporated in this investigation, contributed little additional information, given how homogenous the values were. Additionally, while other limnological investigations offer similarly crude averages for individual ice regimes of individual lakes, most limnological studies do not include any lake-ice information at all. Given the importance of lake-ice in relation to phytoplankton growth and lake productivity (AMAP, 2011), its integration and coupling with studies of algal biomass in Arctic lakes needs to occur with greater urgency. It is evident that increased efforts to accurately record lake-ice data with corresponding hydro-ecological measurements are needed.

6.7 Conclusion and Recommendations

The objective of this exercise was to improve predictive modelling of Chl-*a* in Arctic lakes by increasing the number of observations in datasets used for analyses, testing the capacity of established parameters to predict Chl-*a* given the larger dataset, testing the validity of established nutrient – Chl-*a* relationships, as well as testing the capacity of additional water quality and cryospheric parameters (e.g., dissolved organic carbon, dissolved inorganic carbon, specific conductivity, and lake-ice duration) to predict variation in algal biomass. There was insufficient evidence to conclude that specific conductivity explained a significant portion of variation in Chl-*a*. A lack of reliable data, and corresponding measurements of Chl-*a*, severely limited the attempts at incorporating dissolved inorganic matter and lake-ice duration into this investigation; it is evident that greater efforts to accurately and consistently measure and record these parameters are necessary. Dissolved organic carbon appeared to have an important effect on autotrophic productivity; as such, there is a need to expand paradigms of lake-ecosystem analysis to include DOC in addition to nutrient and Chl-*a* data.

In addition to DOC, future limnological investigations should ensure that measurements of organic carbon are incorporated into the study design to appropriately establish current baseline conditions on which future change can be compared. More specifically, rather than acquiring one instantaneous “snap-shot” of DOC, uninterrupted and recurrent sampling of DOC with corresponding measurements of Chl-*a* would facilitate a more comprehensive understanding of carbon dynamics in Arctic lakes. Similarly, associated measurements of lake-ice duration would facilitate a more comprehensive and integrated analysis of the multitude of processes regulating Arctic autotrophic productivity. Total phosphorous continues to be of considerable importance to phytoplankton dynamics in Arctic lakes; given that both TP and the log-transformation of TP explain significant variation in the algal biomass of Arctic lakes, it is imperative that TP continues to be a consistently measured in limnological investigations.

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CHAPTER 7: CONCLUSION

Recently, intensifying awareness of the capacity of Arctic lakes to reflect changes in climate have instigated demands for the development and implementation of research initiatives designed to track changes to the ecological integrity of these systems (e.g., Lim and Douglas, 2003; Rautio et al., 2011; MacDonald et al., 2012). Changing hydrological and crysospheric parameters (e.g., source and timing of water inputs, evaporation, and ice-cover extent and duration) are interacting simultaneously with changes in biogeochemical cycling and resultant ecological responses (Prowse et al., 2006; Mesquita et al., 2010). Developing the capability to meaningfully identify and predict changes in this environmentally complex region is urgently needed (Schindler and Smol, 2006).

Moreover, given the appropriate collection, integration, analysis, and interpretation of significant physical, biogeochemical, and ecological data, northern lakes can produce essential information pertaining to the vulnerability of water resources to climate change and variability (Williamson et al., 2009). It is important that appropriate baseline data is collected to define reference conditions against which subsequent change can be compared. Monitoring initiatives need to be designed to accommodate and capture the unique conditions that characterize Arctic lake ecosystems, as well as differentiate climate-induced changes from the confounding effects of other anthropogenic forces (e.g., acidification and eutrophication) (Williamson et al., 2009).

The capability of novel scientific equipment to record continuous and uninterrupted limnological and environmental measurements was tested in an upland, Arctic tundra lake. Three main criteria were used to evaluate the effectiveness of the instrumentation: (1) sensor reliability; (2) ability to assess spatial homogeneity in environmental conditions; and (3) measurement validity. Overall, the automated instruments were adequate in recording continuous data over a three-year period; however, the water quality (YSI) sondes installed on the subsurface mooring and homemade mooring components were more effective in recording continuous data, at a multitude of depths, than those mounted to the ALMS buoy. The loss of battery-charge in the water quality sondes continues to be a serious constraint especially during under-ice conditions. Water quality values for several parameters quantified from bulk water samples at various locations around the lake did not significantly differ from one another.

Measurement validation requires further investigation as there were a multitude of sensor malfunctions during the study period and the limited parameters that had corresponding laboratory-derived values were questionable under comparison. Only specific conductivity, turbidity, and pH were tested. The concentration of dissolved oxygen, percent saturation of dissolved oxygen, chlorophyll-*a*, oxidation reduction potential, conductivity, and blue green algae had no corresponding analytical values through which comparisons could be made with buoy- and mooring-sonde data because there were no analogous NLET analytical values.

Water samples, collected during a multitude of open-water and under-ice seasons, were shipped to NLET laboratories for further chemical and nutrient analyses providing a sequence of robust geochemical “snap-shots” of instantaneous conditions in Noell Lake. Analyses of recorded limnological and environmental data generated detailed information pertaining to whole-lake mixing events (August 12th – August 26th, 2012 and June 10th – June 23rd, 2013), the development and persistence of winter and summer stratification (from October 23rd, 2012 until June 10th, 2013), and the development of under-ice hypoxic conditions (onset was March 3rd, 2013). The results of nutrient-limitation experiments undertaken in July of 2013 suggest that autotrophic productivity was constrained by nitrogen, while experiments on heterotrophic productivity were inconclusive.

Relationships amongst, as well as temporal trends in algal biomass, dissolved nutrients, dissolved organic/inorganic carbon, lake-ice thickness and duration, and latitude were evaluated using a newly synthesized dataset of over 700 high latitude lakes of the northern hemisphere. Temporal trends in the average open-water primary productivity of Arctic lake ecosystems did not replicate the increases in primary productivity observed in Arctic terrestrial landscapes. In contrast, open-water productivity (as measured by Chlorophyll-*a*) in Arctic and subarctic lakes have decreased since the 1960s. The separation of lakes into latitudinal bands was crucial to generating more refined hypotheses pertaining to the observed temporal trends in Chl-*a*, as well as key environmental drivers (TP, TN, and DOC). The significant decreases in Arctic lake Chl-*a* were driven by trends in the low-Arctic region (60.00-69.99 Degrees North). Furthermore, TP, TN, and DOC each reflected disparate temporal trends across the various latitudinal bands. Several hypotheses could explain these observations; they are not mutually exclusive and there may be others.

A series of analyses related to linear regressions and mixed-effects modelling of primary productivity (as determined by chlorophyll-*a*) were undertaken in to further understanding into factors regulating autotrophic productivity in Arctic lakes. Linear relationships reported by Flanagan et al. (2003) were validated using a larger, more spatially-comprehensive dataset comprised of novel lakes; the size of the dataset used for analyses was increased from 113 lake years' worth of data to 509 lake years' worth of data, all of which are located north of 60^{o}N. The capacity of novel limnological variables (lake depth, conductivity, dissolved organic carbon, dissolved inorganic carbon, lake-ice duration) to explain variability in Chl-*a* given the new dataset were evaluated. Dissolved nutrients (especially total phosphorous) and latitude continue to be fundamental to explaining variation in Chl-*a*; dissolved organic carbon proved to be use in explaining trends in algal biomass as well. More detailed ice-cover duration and thickness data are needed to appropriately assess the ability of ice phenology to affect autotrophic production in Arctic lake ecosystems.

7.1 Recommendations for future research

There is an urgent need to improve available datasets of northern hemisphere, high-latitude lake-ice thickness and duration. The widespread adoption of consistent methodologies for tracking changes to like-ice phenology are essential across the circumpolar Arctic moving forward; there is also a need for a greater number of detailed re-constructions of historical lake phenologies, especially in high latitude regions of the northern hemisphere. Investigations into the effect of increased allochthonous dissolved organic carbon on the ratio of autotrophic to heterotrophic productivity in Arctic lakes are necessary. Given increased understanding into the importance of dissolved nutrients and dissolved organic carbon, as well as the suggested importance of dissolved inorganic carbon and lake-ice duration and thickness to the algal biomass of Arctic lakes, there is a need to increase simultaneous, *in-situ* measurements of these parameters moving forward; future automated and non-automated buoy and mooring initiatives should focus on the continuous collection of these parameters. Similar to the analysis of primary productivity undertaken in Chapter 5, it would be useful to analyze temporal trends in Arctic heterotrophic/secondary productivity. Greater investigations into the effect of increased exposure of UVR, due to decreases in lake-ice and snow extent as well as reductions in the concentration of Arctic ozone, on phytoplankton are needed.

7.2 References

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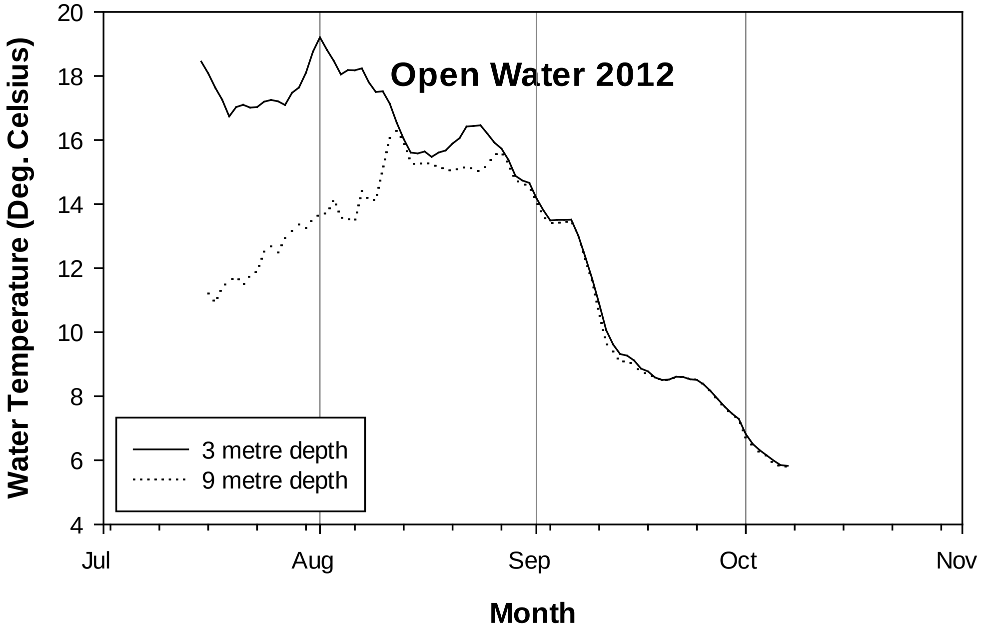
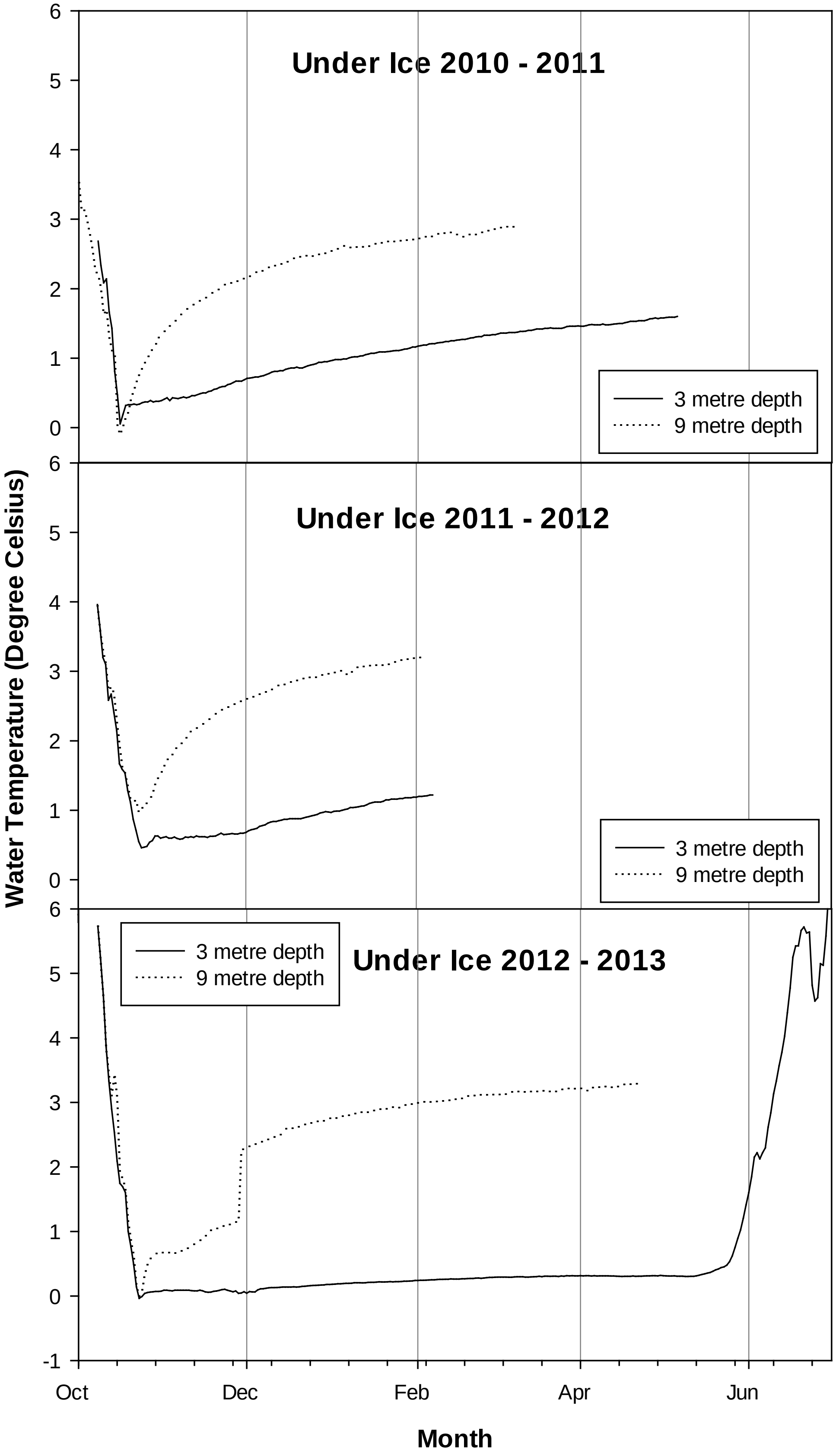
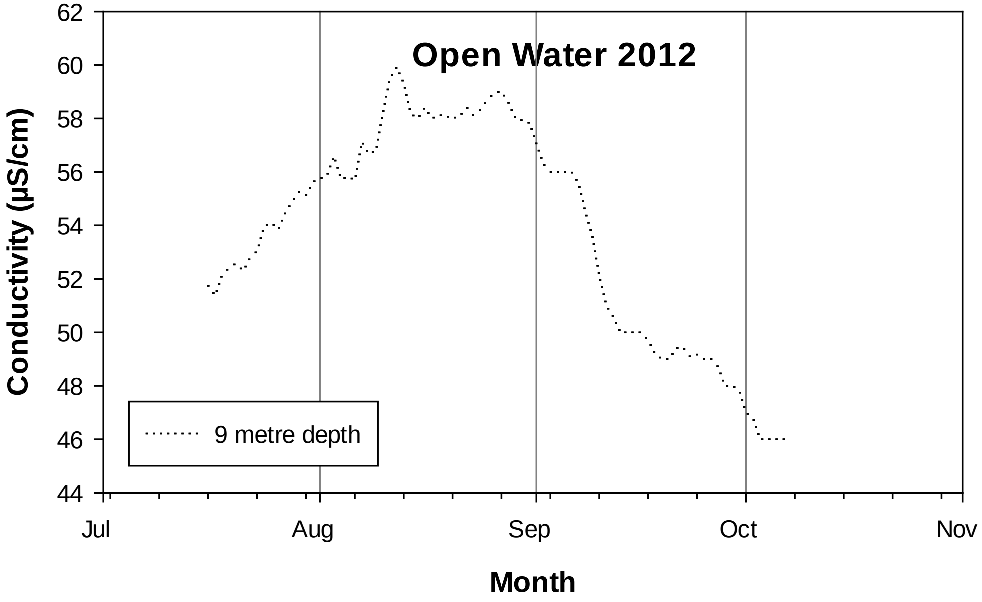
Appendix

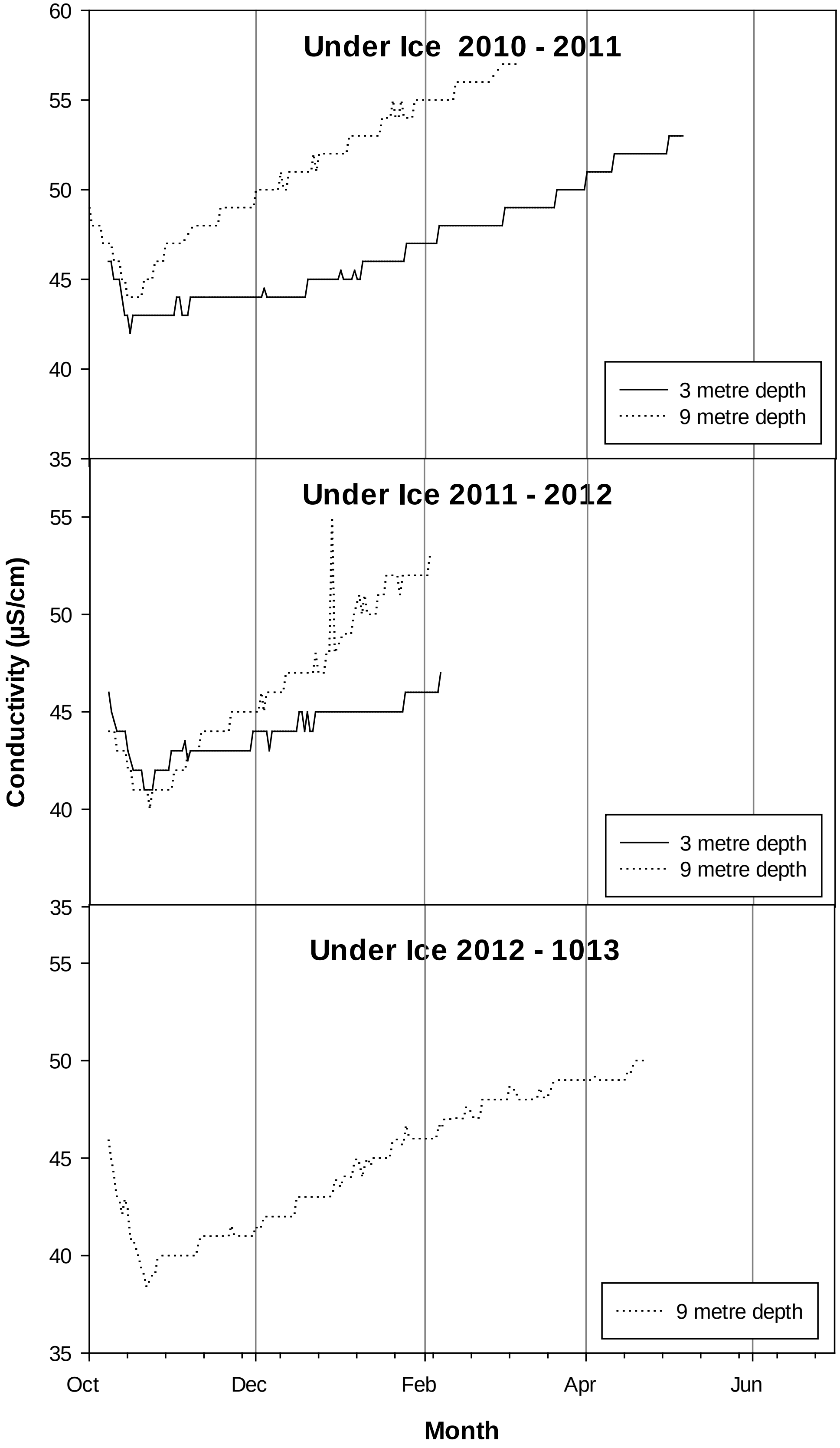
Appendix A: Temporal variation in average daily water temperature (^{o}C) throughout consecutive open-water and under-ice seasons.



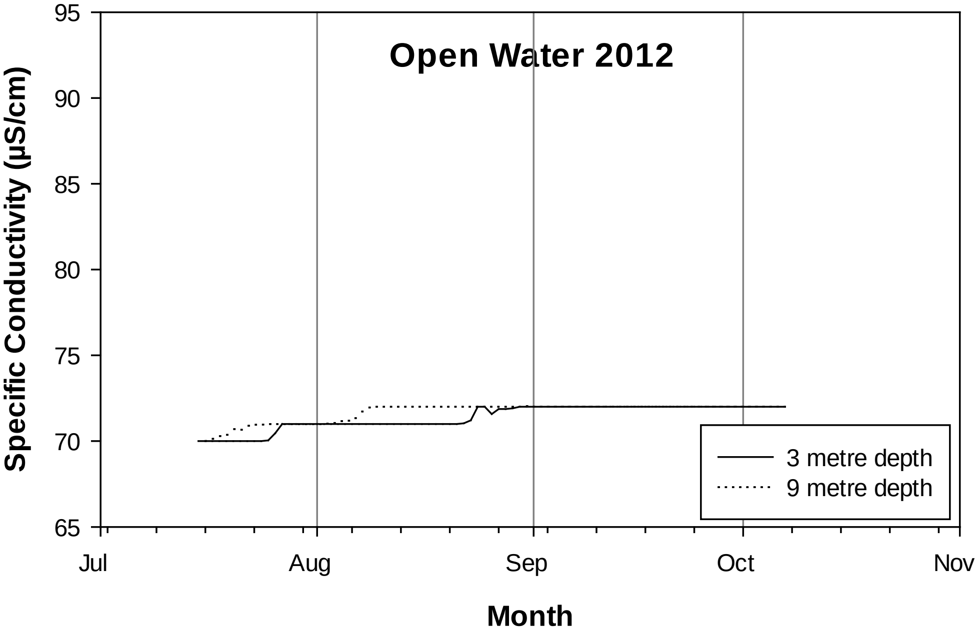

Under Ice 2010 - 2011
Under Ice 2011 - 2012
Under Ice 2012 - 2013
3 metre depth
9 metre depth
Water Temperature (Degree Celsius)
Month
-1
0
1
2
3
4
5
6
Oct
Dec
Feb
Apr
Jun

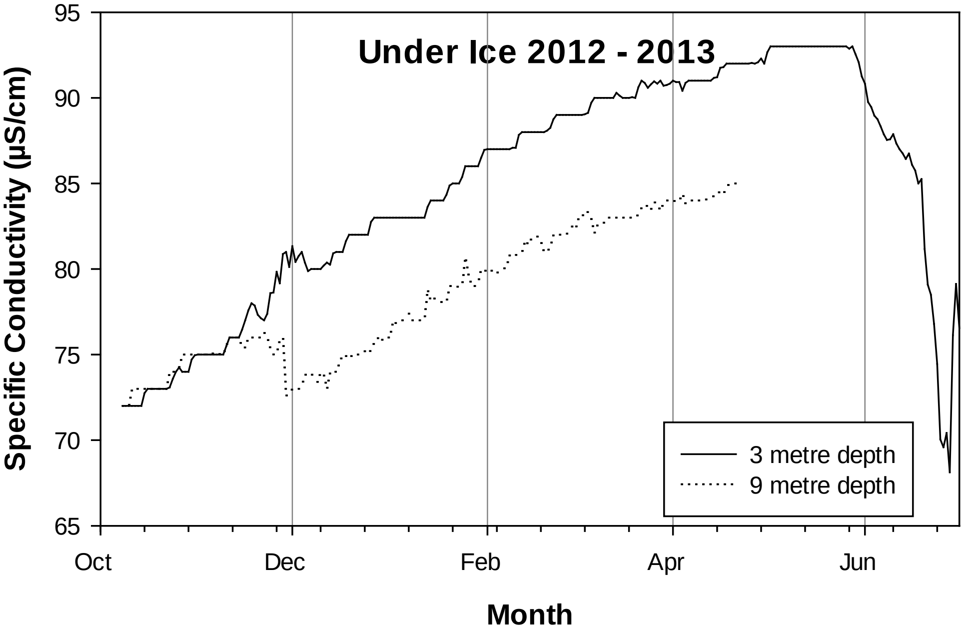
Appendix B: Temporal variation in conductivity (µS/cm) throughout consecutive open-water and under-ice seasons.



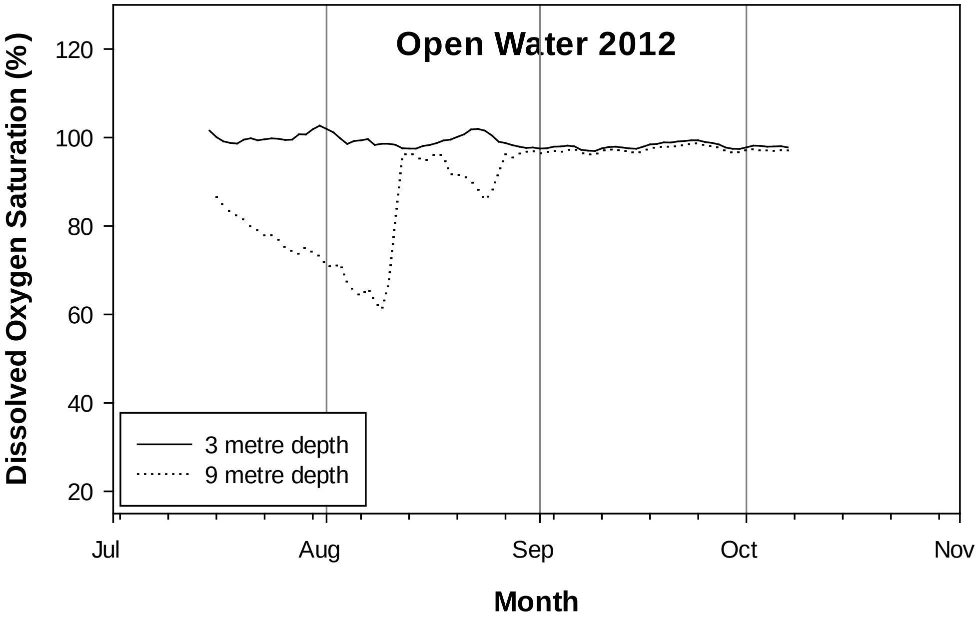

Under Ice 2010 - 2011
Under Ice 2011 - 2012
Under Ice 2012 - 1013
3 metre depth
9 metre depth
Conductivity (µS/cm)
60
55
50
45
40
35
Oct
Dec
Feb
Apr
Jun
Month

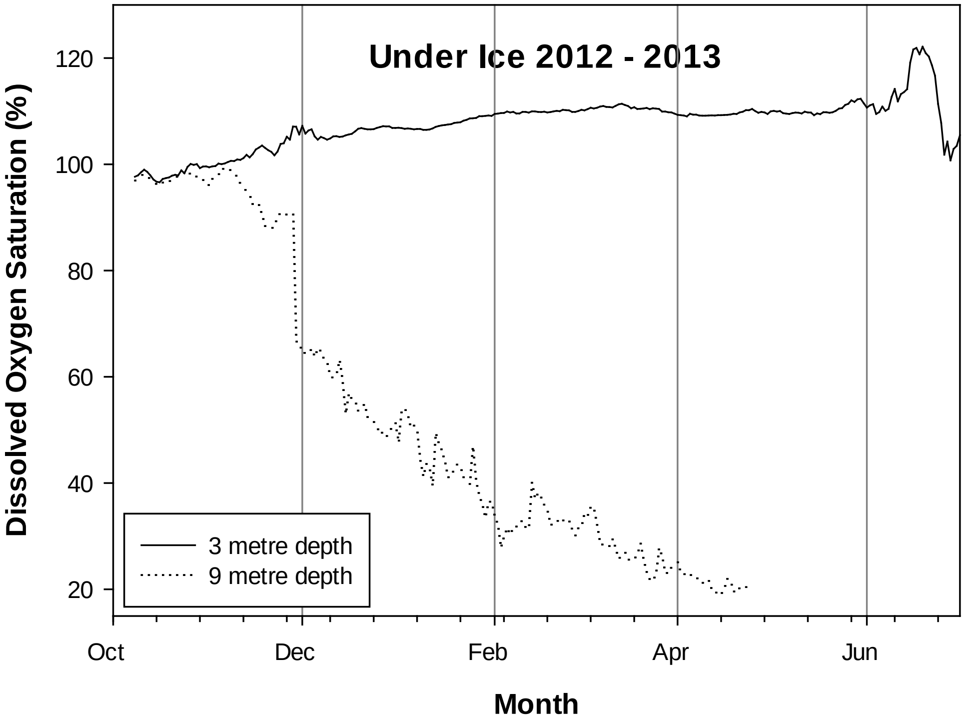
Appendix C: Temporal variation in specific conductance (µS/cm) throughout consecutive open-water and under-ice seasons.



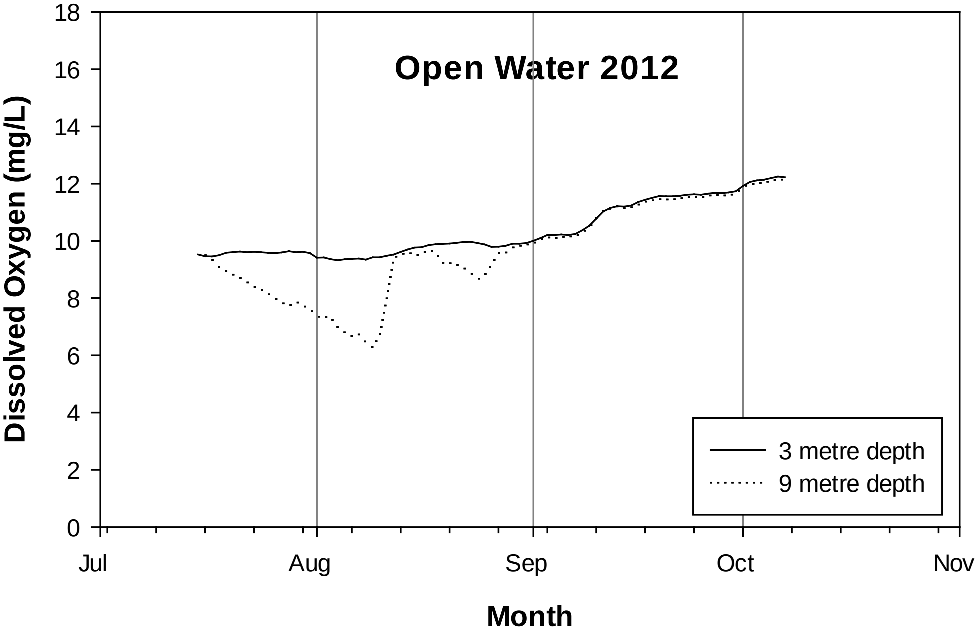


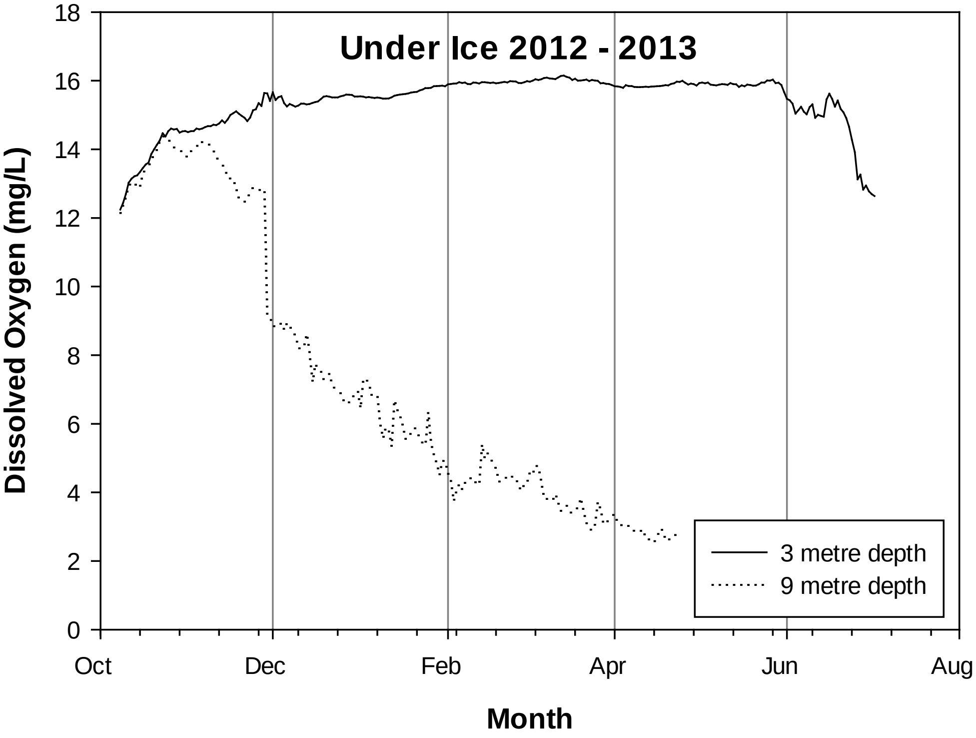
Appendix D: Temporal variation in the percent saturation of dissolved oxygen throughout consecutive open-water and under-ice seasons.



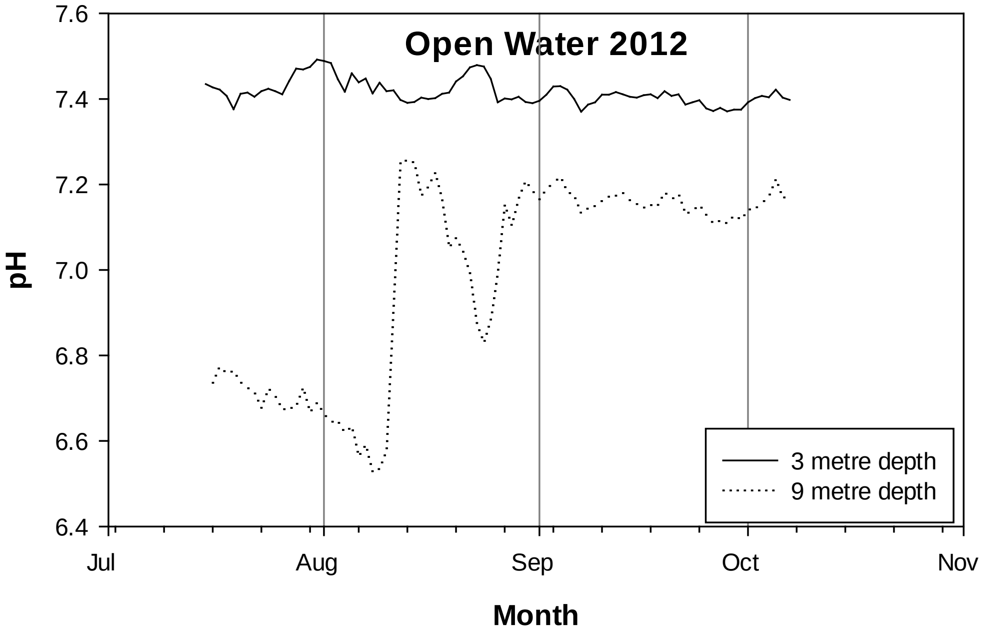


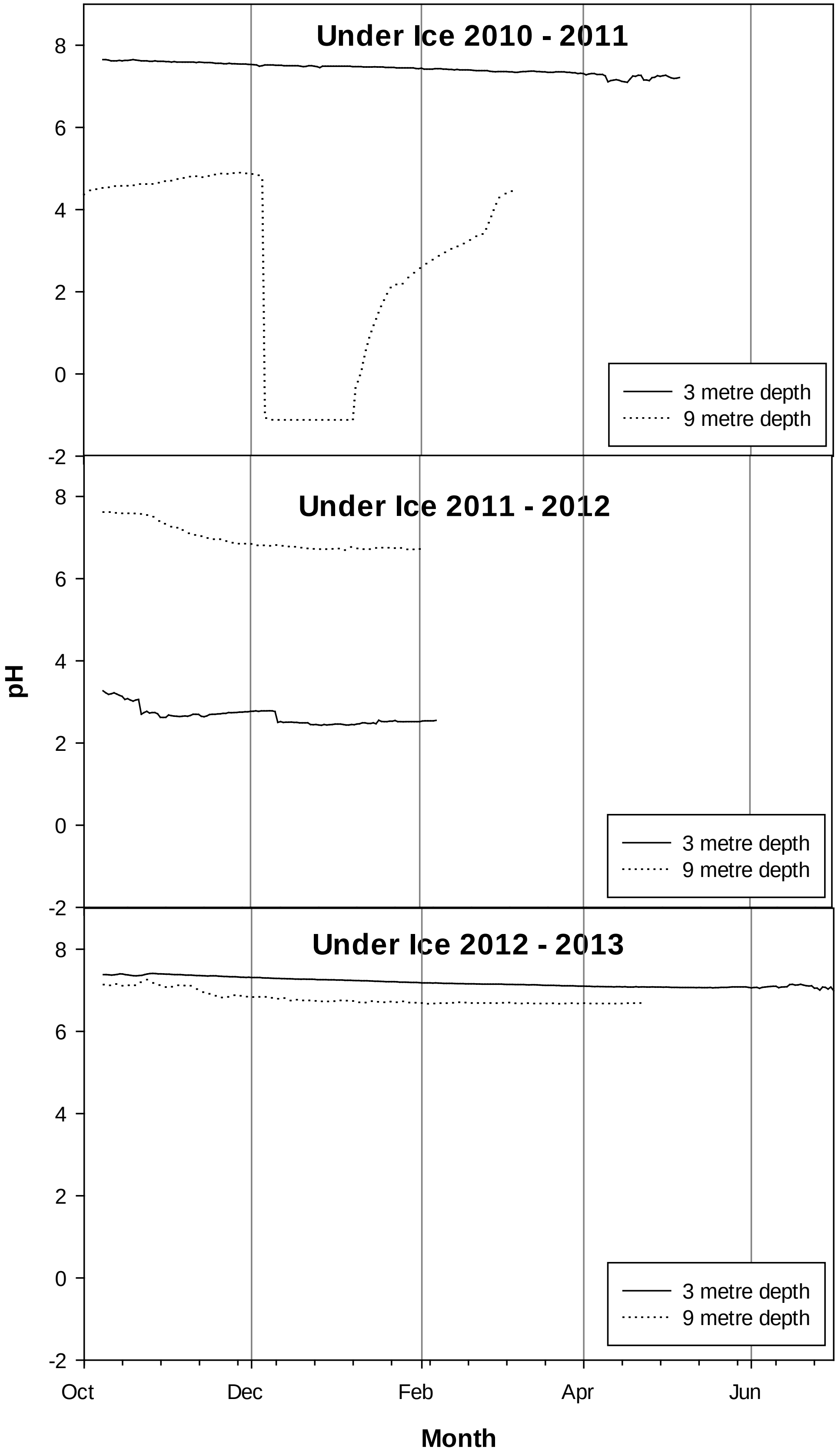
Appendix E: Temporal variation in dissolved oxygen (mg/L) throughout consecutive open-water and under-ice seasons.



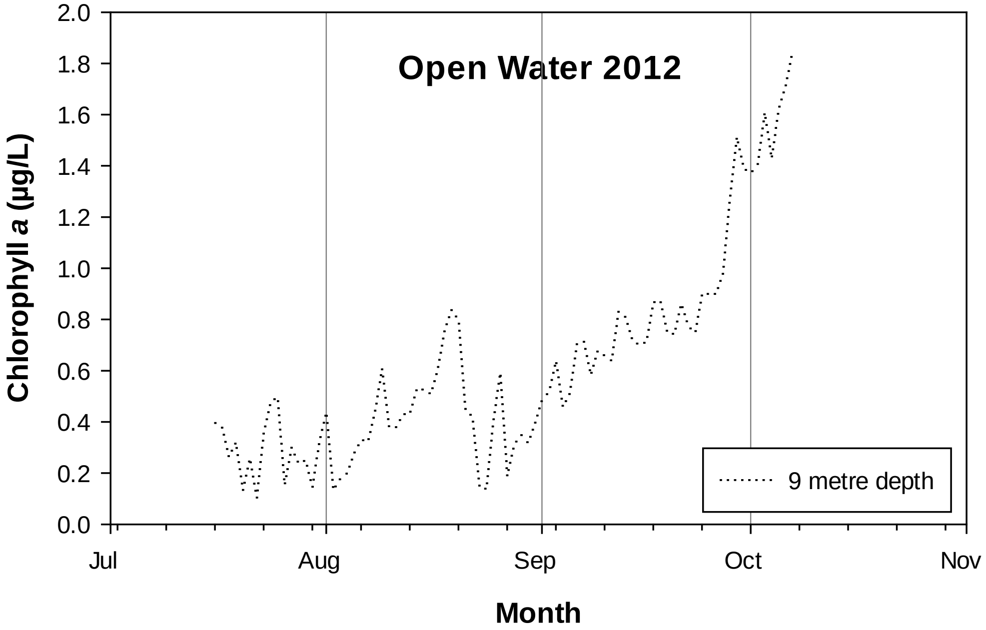


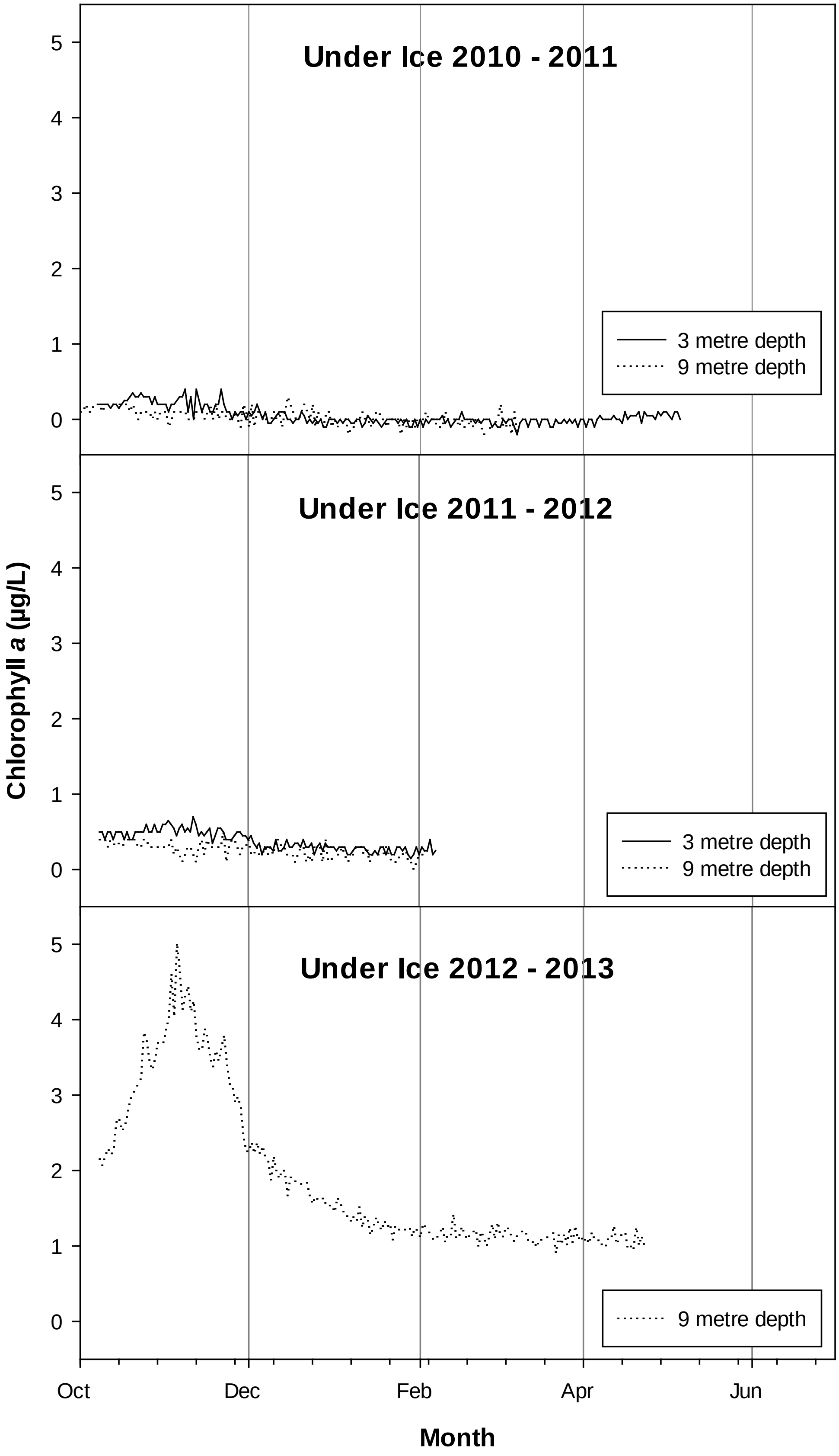
Appendix F: Temporal variation in pH throughout consecutive open-water and under-ice seasons.



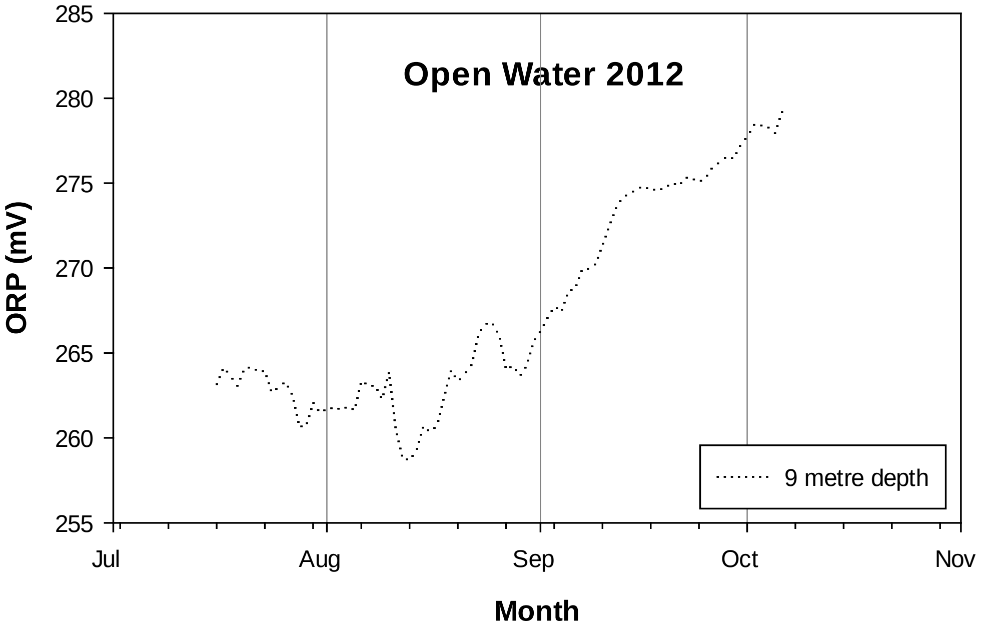

Under Ice 2010 - 2011
Under Ice 2011 - 2012
Under Ice 2012 - 2013
pH
8
6
4
2
0
-2
3 metre depth
9 metre depth
Oct
Dec
Feb
Apr
Jun
Month

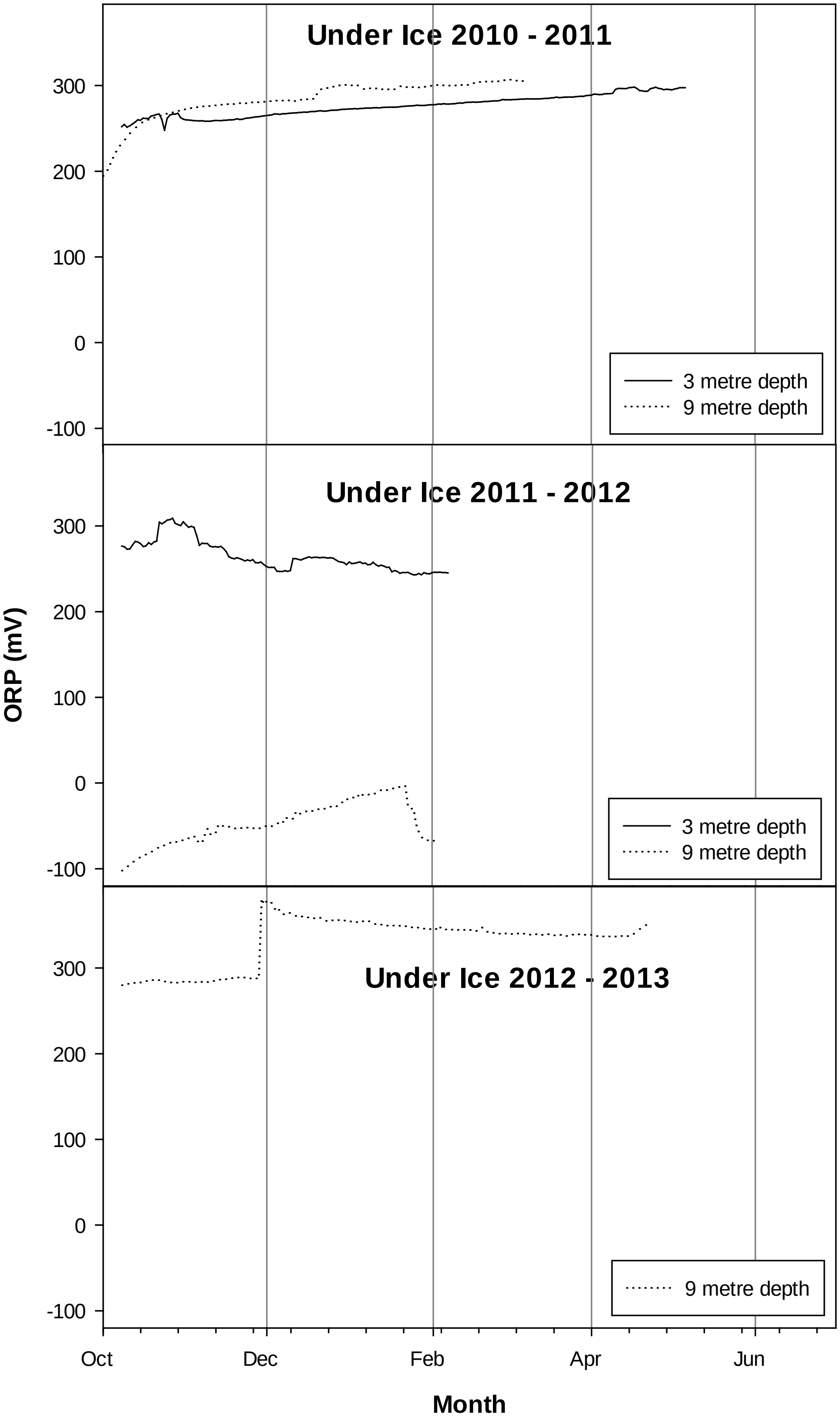
Appendix G: Temporal variation in chlorophyll-*a* (µg/L) throughout consecutive open-water and under-ice seasons.



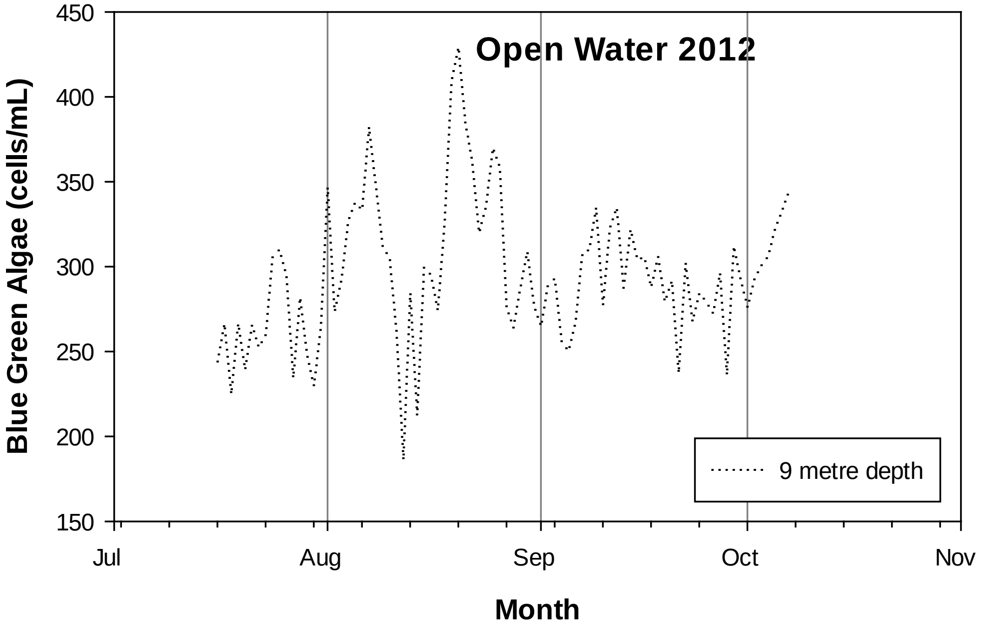

Under Ice 2010 - 2011
Under Ice 2011 - 2012
Under Ice 2012 - 2013
3 metre depth
9 metre depth
Chlorophyll *a* (µg/L)
0
1
2
3
4
5
Oct
Dec
Feb
Apr
Jun
Month

Appendix H: Temporal variation in oxidation-reduction potential (mV) throughout consecutive open-water and under-ice seasons.

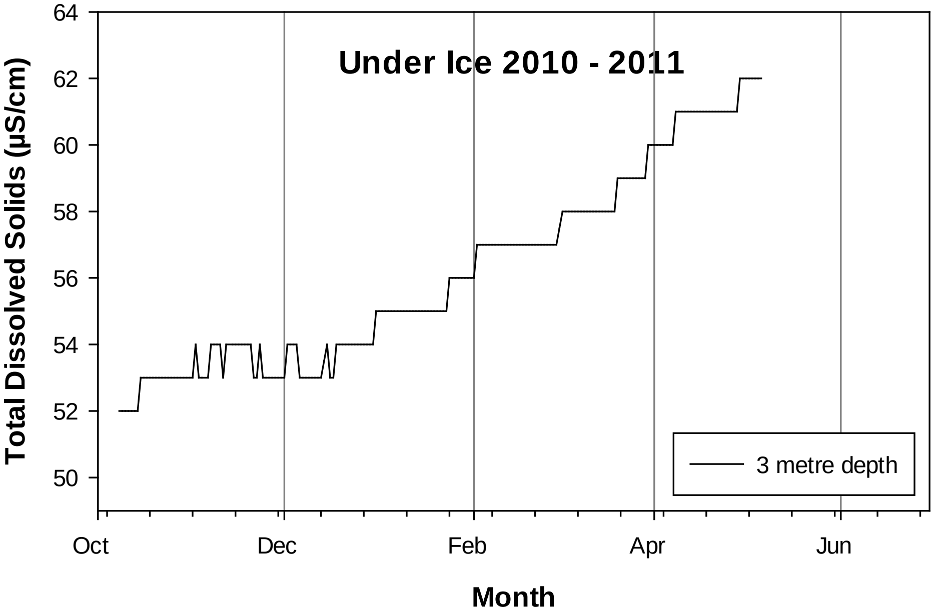


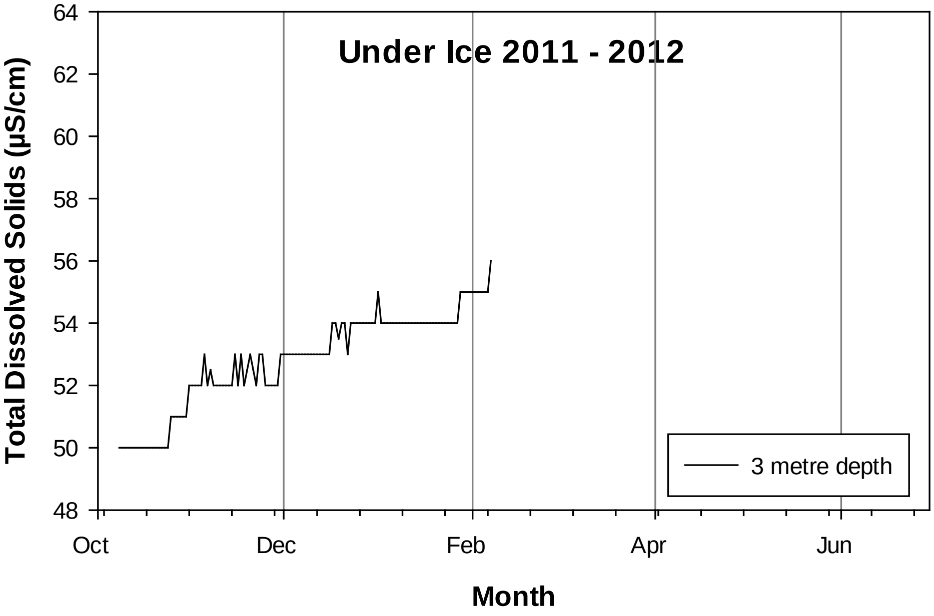

Under Ice 2010 - 2011
Under Ice 2011 - 2012
Under Ice 2012 - 2013
3 metre depth
9 metre depth
ORP (mV)
300
200
100
0
-100
Oct
Dec
Feb
Apr
Jun
Month

Appendix I: Temporal variation in blue green algae (cells/mL) throughout consecutive open-water and under-ice seasons.



Appendix J: Temporal variation in total dissolved solids (µS/cm) throughout consecutive open-water and under-ice seasons.





Appendix K: Chemical and nutrient analyses of water samples removed from various locations and depths of Noell Lake on September 17th, 2011.

	Chemical Analysis																
Sample Location	Spec. Cond.	pH	Alk.	F	Cl	SO_4	Colour	DOC	DIC	Hard.	Ca	Mg	Na	K	NO_2	SiO_2	Turb.
Site 3	74.0	7.38	17.3	0.07	1.77	11.7	4.1	5.4	4.5	26.9	7.21	2.17	2.46	1.07	0.002	0.25	0.30
Site 4	72.4	7.49	17.9	0.07	1.76	11.7	4.3	5.3	4.3	26.6	7.12	2.15	2.48	1.07	0.002	0.25	0.36
Site 13	72.1	7.46	18.0	0.07	1.77	11.7	4.1	5.5	4.3	26.1	6.94	2.15	2.49	1.07	0.002	0.25	0.27
Site 19	72.2	7.45	18.1	0.07	1.77	11.7	4.1	5.4	4.2	26.5	7.04	2.16	2.52	1.06	0.002	0.25	0.30
Site 25	71.8	7.47	18.0	0.07	1.78	11.7	4.2	5.4	4.4	26.6	7.06	2.19	2.49	1.07	0.002	0.24	0.26
Site 27	72.2	7.40	17.3	0.07	1.77	11.7	3.9	5.3	4.5	25.9	6.86	2.13	2.49	1.07	0.002	0.23	26.70
NB 3m	73.3	7.53	16.4	0.07	1.77	11.7	4.2	5.3	4.3	26.5	7.08	2.15	2.51	1.06	0.002	0.23	0.36
NB 4.5m	73.0	7.50	17.8	0.07	1.77	11.7	4.2	5.4	4.3	26.9	7.22	2.16	2.49	1.07	0.003	0.23	0.33
NB 8m	59.7	7.24	12.9	0.06	1.39	8.8	40.7	14.0	3.0	24.0	6.32	1.99	1.60	0.85	0.004	0.01	1.41
Mean	71.19	7.44	17.08	0.069	1.73	11.38	8.20	6.33	4.20	26.22	6.98	2.14	2.39	1.040	0.002	0.216	3.36
Max.	74.00	7.53	18.10	0.070	1.78	11.70	40.70	14.00	4.50	26.90	7.22	2.19	2.52	1.070	0.004	0.25	26.70
Min.	59.70	7.24	12.90	0.060	1.39	8.80	3.90	5.30	3.00	24.00	6.32	1.99	1.60	0.85	0.002	0.01	0.26
S.D.	4.36	0.087	1.66	0.003	0.13	0.96	12.19	2.88	0.46	0.90	0.27	0.058	0.30	0.073	0.001	0.078	8.76

	Nutrient Analysis										
Sample Location	DP	NH_3	NO_3NO_2	OP	POC	PON	TN	TP	TDN	Turb.	Colour
Site 3	0.0020	0.0060	0.001	0.0003	0.13	0.030	0.258	0.0076	0.209	0.4	--
Site 4	0.0025	0.011	0.001	0.0002	0.18	0.035	0.241	0.0051	0.228	0.4	--
Site 13	0.0020	0.0025	0.001	0.0001	0.136	0.035	0.265	0.0062	0.218	0.5	--
Site 19	0.0023	0.0025	0.001	0.0001	0.096	0.036	0.243	0.0050	0.220	0.3	--
Site 25	0.0020	0.0060	0.001	0.0002	0.122	0.029	0.259	0.0044	0.207	0.6	--
Site 27	0.0031	0.0050	0.001	0.0003	6.48	0.757	0.732	0.060	0.208	11.0	--
NB 3m	0.0021	0.0070	0.001	0.0002	0.133	0.021	0.257	0.0058	0.216	0.5	--
NB 4.5m	0.0029	0.0050	0.001	0.0002	0.276	0.036	0.265	0.0067	0.252	0.6	--
NB 8m	0.0052	0.0050	0.001	0.0007	0.271	0.048	0.438	0.012	0.378	1.7	--
Mean	0.0027	0.0056	0.001	0.00026	0.869	0.114	0.329	0.0125	0.237	1.78	--
Max.	0.0052	0.011	0.001	0.00070	6.48	0.757	0.732	0.0596	0.378	11.00	--
Min.	0.0020	0.0025	0.001	0.00010	0.096	0.0210	0.241	0.00440	0.207	0.30	--
S.D.	0.0010	0.0025	0.0	0.00018	2.105	0.241	0.163	0.0178	0.0545	3.48	--

Appendix L: Chemical and nutrient analyses of water samples removed from various locations and depths of Noell Lake on May 13th, 2012.

	Chemical Analysis																
Sample Location	Spec. Cond.	pH	Alk.	F	Cl	SO_4	Colour	DOC	DIC	Hard.	Ca	Mg	Na	K	NO_2	SiO_2	Turb.
Site 3	83.1	7.40	20.1	0.07	1.99	13.1	4.5	6.1	5.1	30.6	8.19	2.46	2.84	1.14	0.001	0.26	--
Site 4	83.5	7.35	19.8	0.08	2.06	13.7	4.2	6.1	5.2	30.6	8.18	2.46	2.82	1.14	0.001	0.31	--
Site 13	95.7	7.39	23.8	0.09	2.40	15.9	4.8	6.8	6.4	35.2	9.42	2.84	3.27	1.33	0.002	0.45	--
Site 19	87.4	7.43	21.6	0.07	1.93	12.4	4.7	6.4	5.4	31.5	8.36	2.57	2.91	1.18	0.001	0.30	--
Site 25	88.1	7.30	21.3	0.08	2.15	14.4	4.4	6.4	5.5	31.8	8.49	2.57	2.94	1.18	0.001	0.29	--
Site 27	88.5	7.27	21.0	0.08	2.17	14.3	4.6	6.6	5.5	32.4	8.62	2.63	2.98	1.21	0.001	0.28	--
NB 3m	87.5	7.22	21.1	0.08	2.09	13.6	4.3	6.5	5.5	31.9	8.50	2.59	2.94	1.20	0.001	0.33	--
NB 4.5m	89.9	7.06	22.6	0.08	2.22	14.4	4.2	5.9	6.7	32.3	8.65	2.61	2.96	1.21	0.002	0.77	--
NB 8m	101.0	7.28	24.8	0.09	2.50	16.5	5.6	7.2	6.3	36.6	9.71	3.00	3.44	1.33	0.002	0.32	--
Mean	89.40	7.30	21.79	0.08	2.17	14.26	4.59	6.44	5.73	32.54	8.68	2.64	3.01	1.21	0.001	0.368	--
Max.	101.0	7.43	24.80	0.09	2.50	16.50	5.60	7.20	6.70	36.60	9.71	3.00	3.44	1.33	0.002	0.770	--
Min.	83.10	7.06	19.80	0.07	1.93	12.40	4.20	5.90	5.10	30.60	8.18	2.46	2.82	1.14	0.001	0.260	--
S.D.	5.70	0.087	1.66	0.007	0.18	1.29	0.43	0.40	0.58	2.037	0.53	0.18	0.21	0.071	0.001	0.160	--

	Nutrient Analysis										
Sample Location	DP	NH_3	NO_3NO_2	OP	POC	PON	TN	TP	TDN	Turb.	Colour
Site 3	0.0037	0.0060	0.0217	0.0016	0.066	0.0050	0.242	0.0055	0.240	0.40	--
Site 4	0.0026	0.016	0.0281	0.0010	0.020	0.0050	0.274	0.0041	0.238	0.10	--
Site 13	0.0023	0.0080	0.0212	0.00070	0.062	0.014	0.264	0.0041	0.252	0.10	--
Site 19	0.0032	0.0080	0.0224	0.0014	0.044	0.010	0.239	0.0052	0.238	0.10	--
Site 25	0.0024	0.017	0.0215	0.00070	0.050	0.0050	0.252	0.0050	0.246	0.10	--
Site 27	0.0021	0.0060	0.0253	0.00050	0.059	0.011	0.274	0.0045	0.261	0.10	--
NB 3m	0.0032	0.0060	0.0275	0.0014	0.053	0.0050	0.241	0.0041	0.240	0.090	--
NB 4.5m	0.0019	0.0060	0.0661	0.00070	0.056	0.0050	0.264	0.0036	0.260	0.20	--
NB 8m	0.0025	0.015	0.0275	0.00080	0.049	0.0050	0.282	0.0041	0.283	0.10	--
Mean	0.0027	0.0098	0.0290	0.00098	0.051	0.0072	0.259	0.0045	0.251	0.14	--
Max.	0.0037	0.017	0.0661	0.0016	0.066	0.014	0.282	0.0052	0.283	0.40	--
Min.	0.0019	0.0060	0.0212	0.00050	0.020	0.0050	0.239	0.0036	0.238	0.090	--
S.D.	0.00059	0.0048	0.0142	0.00039	0.013	0.0035	0.0162	0.00063	0.015	0.10	--

Appendix M: Chemical analyses of water samples removed from various locations and depths of Noell Lake on June 26th, 2012.

	Chemical Analysis																
Sample Location	Spec. Cond.	pH	Alk.	F	Cl	SO_4	Colour	DOC	DIC	Hard.	Ca	Mg	Na	K	NO_2	SiO_2	Turb.
Site 3	69.9	7.12	17.2	0.06	1.77	10.9	0.25	--	--	25.7	6.79	2.13	2.43	1.06	--	0.36	--
Site 4	69.6	7.14	17.1	0.06	1.77	10.9	8.70	--	--	25.7	6.77	2.13	2.43	1.05	--	0.36	--
Site 13	71.8	7.23	17.7	0.07	1.81	11.2	7.50	--	--	26.5	6.98	2.20	2.51	1.07	--	0.36	--
Site 19	70.8	7.21	17.2	0.06	1.78	11.0	8.00	--	--	26.0	6.84	2.16	2.47	1.07	--	0.35	--
Site 25	71.9	7.28	17.5	0.06	1.80	11.2	7.40	--	--	26.6	7.03	2.20	2.51	1.08	--	0.36	--
Site 27	71.7	7.22	17.1	0.06	1.81	11.3	7.50	--	--	26.7	7.04	2.21	2.51	1.08	--	0.35	--
NB 3m	71.0	7.23	16.8	0.07	1.80	11.2	8.10	--	--	26.3	6.94	2.17	2.48	1.07	--	0.36	--
NB 4.5m	70.9	7.18	17.0	0.06	1.80	11.1	8.20	--	--	26.3	6.96	2.18	2.49	1.07	--	0.36	--
NB 8m	72.7	7.11	17.2	0.07	1.81	11.3	7.70	--	--	26.9	7.09	2.22	2.55	1.09	--	0.37	--
Mean	71.14	7.19	17.20	0.063	1.79	11.12	7.04	--	--	26.30	6.94	2.18	2.49	1.07	--	0.36	
Max.	72.70	7.28	17.50	0.070	1.81	11.30	8.70	--	--	26.90	7.09	2.22	2.55	1.09	--	0.37	--
Min.	69.60	7.11	16.80	0.060	1.77	10.90	0.25	--	--	25.70	6.77	2.13	2.43	1.05	--	0.35	--
S.D.	0.99	0.057	0.26	0.005	0.02	0.16	2.58	--	--	0.43	0.11	0.033	0.039	0.01	--	0.006	--

Appendix N: Chemical and nutrient analyses of water samples removed from various locations and depths of Noell Lake on November 19th, 2012.

Sample Location	Chemical Analysis																
	Spec. Cond.	pH	Alk.	F	Cl	SO_4	Colour	DOC	DIC	Hard.	Ca	Mg	Na	K	NO_2	SiO_2	Turb.
Site 3	84.5	7.22	19.6	0.05	2.91	12.7	5.2	--	4.5	29.1	7.59	2.47	3.26	1.15	--	0.24	--
Site 4	76.7	7.21	18.1	0.05	1.89	12.1	4.9	--	4.3	27.6	7.23	2.31	2.67	1.11	--	0.22	--
Site 13	76.7	7.21	18.3	0.05	1.91	12.2	4.9	--	4.3	27.5	7.21	2.31	2.65	1.11	--	0.22	--
Site 19	76.5	7.25	19.0	0.05	1.89	12.2	4.9	--	4.2	27.7	7.29	2.31	2.70	1.11	--	0.22	--
Site 25	82.2	7.23	19.8	0.05	2.04	13.2	5.4	--	4.5	30.6	8.14	2.49	2.88	1.20	--	0.24	--
Site 27	78.0	7.21	19.0	0.05	1.99	12.6	5.1	--	4.3	28.1	7.35	2.37	2.74	1.17	--	0.22	--
NB 3m	78.7	7.23	19.1	0.05	1.96	12.8	5.0	--	4.3	27.8	7.26	2.35	2.70	1.15	--	0.23	--
NB 4.5m	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
NB 8m	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Mean	79.04	7.22	18.99	0.05	2.08	12.54	5.057	--	4.34	28.34	7.44	2.37	2.80	1.14	--	0.23	--
Max.	84.5	7.25	19.80	0.05	2.91	13.20	5.40	--	4.50	30.60	8.14	2.49	3.26	1.20	--	0.24	--
Min.	76.5	7.21	18.11	0.05	1.89	12.10	4.90	--	4.20	27.50	7.21	2.31	2.65	1.11	--	0.22	--
S.D.	3.12	0.015	0.62	0.0	0.37	0.40	0.19	--	0.11	1.13	0.34	0.077	0.22	0.035	--	0.009	--

Sample Location	Nutrient Analysis										
	DP	NH_3	NO_3NO_2	OP	POC	PON	TN	TP	TDN	Turb.	Colour
Site 3	0.0020	0.0025	0.001	0.0001	--	--	0.232	0.0044	0.192	0.4	0.25
Site 4	0.0019	0.0025	0.003	0.0001	--	--	0.214	0.0049	0.214	0.4	0.25
Site 13	0.0019	0.0060	0.003	0.0001	--	--	0.213	0.0044	0.197	0.3	0.25
Site 19	0.0020	0.0060	0.004	0.0002	--	--	0.214	0.0049	0.197	0.3	0.25
Site 25	0.0019	0.0060	0.001	0.0003	--	--	0.232	0.0047	0.227	0.4	0.25
Site 27	0.0025	0.0070	0.003	0.0003	--	--	0.224	0.0051	0.197	0.4	0.25
NB 3m	0.0018	0.0025	0.002	0.0001	--	--	0.212	0.0047	0.193	0.4	0.25
NB 4.5m	--	--	--	--	--	--	--	--	--	--	--
NB 8m	--	--	--	--	--	--	--	--	--	--	--
Mean	0.0020	0.0046	0.0024	0.00017	--	--	0.22	0.0047	0.202	0.37	0.25
Max.	0.0025	0.0070	0.0040	0.00030	--	--	0.23	0.0051	0.227	0.40	0.25
Min.	0.0018	0.0025	0.0010	0.00010	--	--	0.21	0.0044	0.192	0.30	0.25
S.D.	0.00023	0.0020	0.0011	0.000095	--	--	0.0090	0.00026	0.0131	0.049	0.0

Appendix O: Chemical and nutrient analyses of water samples removed from various locations and depths of Noell Lake on July 15th, 2013.

Sample Location	Chemical Analysis																
	Spec. Cond.	pH	Alk.	F	Cl	SO_4	Colour	DOC	DIC	Hard.	Ca	Mg	Na	K	NO_2	SiO_2	Turb.
Site 3	71.2	7.31	17.1	0.07	1.79	11.2	9.1	5.8	4.2	27.1	7.36	2.11	2.49	1.06	0.001	0.58	--
Site 4	70.8	7.30	17.1	0.07	1.80	11.2	8.8	5.6	4.1	26.9	7.30	2.10	2.50	1.06	< 0.001	0.59	--
Site 13	70.6	7.33	16.5	0.07	1.80	11.3	9.2	5.7	4.2	27.0	7.35	2.09	2.47	1.05	< 0.001	0.62	--
Site 19	71.0	7.31	16.3	0.07	1.80	11.3	8.9	5.7	4.1	26.4	7.11	2.11	2.55	1.08	< 0.001	0.59	--
Site 25	70.5	7.34	17.0	0.07	1.80	11.3	9.3	5.7	4.1	26.7	7.19	2.12	2.52	1.08	0.001	0.59	--
Site 27	70.7	7.33	16.9	0.07	1.79	11.2	9.5	5.8	4.1	26.6	7.14	2.14	2.55	1.10	0.001	0.60	--
NB 3m	71.0	7.34	17.2	0.07	1.80	11.3	9.2	5.6	4.1	27.0	7.27	2.14	2.50	1.08	0.001	0.59	--
NB 4.5m	70.8	7.30	16.8	0.07	1.80	11.3	9.3	5.7	4.0	27.0	7.31	2.14	2.47	1.07	< 0.001	0.64	--
NB 8m	71.1	7.33	16.7	0.07	1.79	11.2	9.2	5.7	4.1	27.2	7.40	2.11	2.42	1.04	< 0.001	0.61	--
Mean	70.86	7.32	16.84	0.07	1.80	11.26	9.16	5.7	4.11	26.88	7.27	2.12	2.50	1.07	<0.001	0.60	--
Max.	71.20	7.34	17.20	0.07	1.80	11.30	9.50	5.8	4.20	27.20	7.40	2.14	2.55	1.1	0.001	0.64	--
Min.	70.50	7.30	16.30	0.07	1.79	11.20	8.80	5.6	4.00	26.40	7.11	2.09	2.42	1.04	<0.001	0.58	--
S.D.	0.23	0.016	0.30	0.0	0.01	0.053	0.21	0.071	0.060	0.26	0.10	0.019	0.041	0.02	--	0.019	--

Sample Location	Nutrient Analysis									
	DP	NH_3	NO_3NO_2	OP	POC	PON	TN	TP	TDN	Turb.	Colour
Site 3	0.0023	0.0049	0.001	0.0003	0.20	0.031	0.245	0.0071	0.204	1.3	8.0
Site 4	0.0022	0.0060	0.001	0.0002	0.15	0.026	0.284	0.0073	0.242	1.5	8.0
Site 13	0.0026	0.010	0.001	0.0003	0.22	0.035	0.253	0.0078	0.204	2.0	8.0
Site 19	0.0024	0.0060	0.001	0.0002	0.23	0.043	0.241	0.0098	0.206	1.9	8.0
Site 25	0.0021	0.0049	0.001	0.0002	0.17	0.031	0.275	0.0066	0.219	1.6	8.0
Site 27	0.0027	0.031	0.001	0.0003	0.17	0.025	0.240	0.012	0.199	1.6	8.0
NB 3m	0.0020	0.0080	0.001	0.0002	0.16	0.024	0.232	0.0091	0.196	1.6	8.0
NB 4.5m	0.0022	0.020	0.001	0.0002	0.19	0.031	0.241	0.010	0.201	1.9	8.0
NB 8m	0.0022	0.025	0.001	0.0002	0.21	0.030	0.246	0.0075	0.237	1.9	8.0
Mean	0.0023	0.0129	0.001	0.00023	0.190	0.0307	0.251	0.00861	0.212	1.7	8.0
Max.	0.0027	0.0310	0.001	0.0003	0.229	0.0430	0.284	0.0123	0.242	2.0	8.0
Min.	0.0020	0.00490	0.001	0.0002	0.153	0.0240	0.232	0.00660	0.196	1.3	8.0
S.D.	0.00023	0.00987	0.0	0.00005	0.0285	0.00581	0.0174	0.00184	0.0169	0.235	0.0