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Boreal Subarctic Lake Water Quality, Zooplankton Communities, and Benthic Macroinvertebrate Communities in Areas Impacted by Wildfire

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Boreal Subarctic Lake Water Quality, Zooplankton Communities, and Benthic Macroinvertebrate Communities in Areas Impacted by Wildfire.

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Abstract

Wildfires are a natural phenomenon in the boreal forest, but recent studies suggest that there will be increases in the frequency and severity of fires in this region over the next century. When wildfires occur in lake catchments, they may affect water quality and invertebrate communities living in associated lakes. Invertebrates, including zooplankton and benthic invertebrates, play important roles in aquatic food webs, acting as conduits of energy from primary producers to larger organisms such as fish and water birds. Therefore, it is important to understand how these groups will respond to changes caused by fires. Unfortunately, few studies have been conducted in the northern boreal forest to examine how wildfires might impact lakes in this region. For my thesis, I collected data on water quality, macrophytes, zooplankton, and benthic invertebrates from 20 lakes in the Sahtú Settlement Area of the Northwest Territories, which has been impacted by recent fires. Nine of these lakes were affected by fires in their catchments 4-5 years prior to data collection, while eleven were not. My results showed that surprisingly few water quality variables were related to burn history, but lake colour and phenols were higher in lakes affected by burns, and pH and dissolved oxygen were lower in those lakes. Remote sensing data suggest that macrophyte biomass could have increased in lakes affected by burns following the 2014 fires, and macrophyte biomass was an important predictor of community composition for both zooplankton and benthic macroinvertebrates. In addition, burn history was an important predictor of zooplankton richness, and benthic macroinvertebrate richness and abundance. However, my analyses also revealed that natural variability in catchment and lake properties was more important for explaining differences in invertebrate communities among lakes than the effects of wildfires. Overall, my results suggest that shifts in the lake water quality and biology are still detectable 4-5 years after burns in their catchments, suggesting that increased frequency and severity of burns in the north could have long-term impacts on boreal lakes.

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Chapter 1: Literature Review and Introduction

Thomas J. Pretty

Structure of the Thesis

For the first chapter of my thesis, I conducted a literature review that examined how aquatic ecosystems in the subarctic boreal forest are affected by wildfires in their catchments. Primary literature sources were used to develop a better understanding of how fires can impact the hydrology, water quality, and food webs of lake ecosystems. Using this information, I developed the study objectives and methods central to my thesis.

In chapter 2, I summarized the study I conducted to examine how wildfires effect lakes in the Sahtú Settlement Area. In this chapter, additional authors contributed to the overall conclusions. Charles-Matthew Chanyi assisted with identifying zooplankton as part of his honours thesis project. Catherine Kuhn provided expertise in analyzing remote sensing imagery over the study area. She provided short-wave infrared (SWIR) surface reflectance measurements obtained from the National Aeronautics and Space Agency’s (NASA) Landsat-8 satellite for each lake. Derek Gray supervised the production of the study and greatly helped with overall editing and developing a structured analysis. This chapter is in manuscript form and is being prepared for submission to the Canadian Journal of Fisheries and Aquatic Sciences.

Finally, in chapter 3, I concluded my thesis by providing an overview of my findings. I also included some additional analysis and discussion that is speculative in nature but could be considered in planning future studies. I also discussed how my research further contributes to the field and finished my thesis by describing how my research was integrative.

Literature Review

Increased frequency and severity of wildfires has been recognized to be associated with climate change (Kasischke et al. 2010; Williams et al. 2019). The increased incidence of wildfire could have a significant impact on freshwater resources, as alterations from wildfire to water catchments can temporarily reduce evapotranspiration (Bond-Lamberty et al. 2009), increase snow accumulation (Pomeroy et al. 1999), and cause snowmelt to occur earlier in the spring (Seibert et al. 2010; Gleason et al. 2019). These cumulative effects can increase runoff, erosion, organic material input, and trace metal levels in water bodies. Despite these generalities, there is a large amount of variability among studies in terms of detecting the effects of wildfire on lakes, and few studies have been conducted at high latitudes. Canada’s boreal forest is home to millions of lakes (Badiou et al. 2011), and fire is a natural occurrence in this region, but northern boreal forest lakes have been infrequently studied post-fire.

Frequency and Severity of Wildfires

Several studies have identified an increase in the frequency and severity of wildfires in the boreal region (Flannigan et al. 2005, 2009; Veraverbeke et al. 2017; Hanes et al. 2019). Technologies for remote sensing of wildfires within Canada started to gain usage in the late 1950s, and records of wildfires (>200 ha) since 1959 are considered to be accurate (Stocks et al. 2003). This suggests that analyses showing increases in the frequency of wildfires within the Canadian boreal forest are unlikely to be related to issues with data availability. Instead, increases in temperature (especially during the fire season) and more frequent lightning ignitions, seem to be driving the increasing number of fires (Hanes et al. 2019). Climate dynamics influence fuel moisture, lightning ignitions, and fire growth through the wind, and predicted changes in these drivers are projected to increase the area burned relative to the year 2015 within the Alaskan and Canadian boreal forest by 150-350 % by the year 2050 (Yue et al. 2015). From 1959 to 1999 alone, the frequency of large fires (>1,000 km^2) had doubled (Kasischke and Turetsky 2006). Positive feedback loops between increasing temperatures and precipitation result in northward expansion of forest growth as permafrost thawing occurs (which increases fuel) and increased storm events that lead to lightning ignitions (Veraverbeke et al. 2017). For example, from 1975 to 2015, there have been significant increases in lightning ignitions within the Northwest Territories (NWT). The year 2014 was a particularly destructive wildfire year within the NWT and it coincided with the most lightning ignitions recorded since 1975 (Veraverbeke et al. 2017). These combined conditions reduced the fire cycle (time required to reburn an area of equal consideration) to less than 100 years in parts of the western boreal forest (Zhang and Chen 2007).

Wildfire regimes in the Canadian Boreal Forest can be described according to wildfire frequency, burned area size, continuity, type (e.g. crown vs. surface fire), seasonality, intensity, and severity (Stocks et al. 2003; Boulanger et al. 2012; Hanes et al. 2019). Intensity is a measurement of the energy output from fire, while severity is the above ground and below ground organic matter consumption from fire (Keeley 2009). Burn severity is related to the type of vegetation community and the associated fuel load (Whitman et al. 2018). For example, wetland areas tend to burn in small variable patches at a low severity (being mainly soil surface to crown burned) (Whitman et al. 2018). The diversity and density of the boreal forest stand structure are major factors that predict the severity of wildfires (Miquelajauregui et al. 2016). Large, continuous wildfires tend to burn at moderate to high severity, allowing organic material to be fully consumed on the ground and deep into the soil.

Runoff

Modification of watershed characteristics can change hydrological patterns, influencing lakes and rivers (Robinne et al. 2020). wildfires are a concern in this regard, as they can alter vegetation and soil properties (Shu-ren 2003; Lavoie and Mack 2012). A meta-analysis showed a consistent relationship where lakes in burned watersheds experienced a larger volume of runoff for 1.5 years after wildfires (Vieira et al. 2015). There are several reasons why wildfires may impact runoff to boreal lakes. One possibility is that wildfire-induced changes affect the storage of precipitation (Pomeroy et al. 1999). During the summer months, clear-cut areas consistently have higher moisture content in the soil as a response to decreased evapotranspiration (Pomeroy et al. 1999). This moisture content in the soil freezes during the winter months and is supplemented by increased snowpack due to increases in redistribution of snow and subsequent sublimation with less canopy cover. As a result, during the spring thaw, runoff is greatly increased in comparison with forested areas (Shu-ren 2003).

There are several examples of this phenomenon. In Swedish and Siberian boreal forests, greater runoff in catchments affected by severe burns was attributed to earlier snowmelt and lower infiltration and evapotranspiration (Lebedeva et al. 2014; Pimentel and Arheimer 2018). A burned plot in the sub-alpine forest of Crowsnest Pass, Alberta, showed greater snow accumulation and more rapid snowmelt in comparison with a control plot (Burles and Boon 2011). Increased snow depth and earlier snowmelt were also observed in wildfire burned plots in the Scotty Creek basin, west of Yellowknife, NWT (Mathieu 2018). The earlier snowmelt can persist for upwards of 15 years post-fire and expose the soil to further runoff (Gleason et al. 2019). However, the effects of wildfire on water storage can vary dependant on the structure of the basin and its retention coefficient, as studies in Tulita, NWT, and Norman Wells, NWT showed no differences in pre and post-fire snowpack depths due to increased wind transporting snow away from the sites (Kershaw 2001).

Although some studies show increases in the runoff to lakes following wildfires, there are reasons to expect that the presence of permafrost in the subarctic peatlands may instead reduce overall input to lakes after wildfires. Approximately 80% of the global boreal forest is within the circumpolar permafrost zone (Helbig et al. 2016). The relative thickness of the active layer (top layer of thawed soil) and the permafrost is important since frozen soils produce higher runoff because of low infiltration and evapotranspiration (Pomeroy et al. 1999). This is commonly alluded to as Horton flow where the soil is waterlogged (Beven 2004). During the first five years post-fire, permafrost thawing within peatlands can occur, producing a deeper active layer (Gibson et al. 2018). This thickening of the active layer may allow more infiltration into the soil, expanding thermokarst bogs instead of creating runoff into established water bodies. Therefore, perturbation caused by a wildfire within humic peat areas can result in isolated bogs that may become lakes through time (Kokelj et al. 2007; Torre Jorgenson et al. 2013). In contrast, peatland areas outside of these depressions within hummocks experience rapid water loss as runoff is directed to thermokarst bogs instead of running across the landscape (Thompson and Waddington 2013). This unsaturated area of peat may then act as a dried sponge over a long period and soak up precipitation or snowmelt instead of allowing runoff. Degradation of permafrost after wildfires may also lead to erosion and thaw slumping (Kokelj et al. 2005), which can modify the properties of runoff, affecting lake water quality for >30 years (Kokelj et al. 2005).

The rapid regrowth of vegetation following wildfire may also affect the amount of runoff reaching lakes over time. Severe burns in Yukon and Alaska led to increased seed germination of vascular plants and moss cover three years post-fire (Johnstone and Chapin 2006). Within one year post-fire, re-sprouted shrubs and herbaceous vascular vegetation dominated the floor cover in central Manitoba (Goulden et al. 2011). In the absence of competition, vigorous tree growth is possible (Liu et al. 2011), which requires a greater water supply. Such rapid regrowth of broadleaf vegetation will impact evapotranspiration rates and subsequent runoff, providing a possible reason that recovery times of runoff to reference baseline conditions may vary dependent on vegetation type and increased canopy cover that reduce soil evaporation (Bond-Lamberty et al. 2009).

In conclusion, many different factors influence how long runoff to aquatic systems is affected within the boreal forest after a wildfire. Severe burning can result in a deeper active layer in locations with permafrost, leading to drier surface soil, to less snowpack and increased porosity and infiltration of precipitation and snowmelt into the soil (Bring et al. 2016). A lack of competition and increased nutrients can result in increased shrub growth and rapid recovery of evapotranspiration (Bring et al. 2016). How long increased runoff will occur after wildfire is dependant on these terrestrial factors. However, studies show an initial increase in nutrient loading followed by long term decreases as the terrestrial catchments recover after wildfire (Robinne et al. 2020).

Water Quality

There are very few studies of lakes in the boreal forest post-fire, but the few studies available, along with stream studies, may allow us to set expectations for water quality impacts. Wildfire within the southern boreal forest has been associated with significant changes in a suite of interrelated physical and chemical variables. Previous studies show increases in concentrations of nutrients, including total nitrogen (TN; Bayley et al. 1992; Garcia and Carignan 1999, 2005; Lamontagne et al. 2000; McEachern et al. 2000; Kelly et al. 2006; but see Lewis et al. 2014 for a counterexample) and total phosphorus (TP; (Bayley et al. 1992; Garcia and Carignan 1999, 2005; Lamontagne et al. 2000; McEachern et al. 2000; Kelly et al. 2006; Wagner et al. 2014; but see Allen et al. 2005; Lewis et al. 2014 for counterexamples) in boreal aquatic systems after wildfires. Increases to phosphorous may persist for over a year within streams impacted by wildfire (Stednick et al. 1982). Increases in TN, and TP, can in turn increase primary production (Garcia and Carignan 1999, 2005; Kelly et al. 2006; Silins et al. 2014).

Within regions that contain permafrost, the thickening and potential weathering of the active layer of permafrost after wildfires could increase the amount of DOC and particulate organic carbon (POC) entering nearby water bodies (Worrall et al. 2017). Many studies have noted post-fire increases in dissolved organic carbon (DOC) and total organic carbon (TOC) in boreal lakes and streams (McEachern et al. 2000; Scrimgeour et al. 2001; Allen et al. 2005; Burd et al. 2018). However, the response of DOC levels to wildfires can be variable (Carignan et al. 2000). For example, two to three years post-fire, streams located west of Yellowknife in the Scotty Creek basin showed no differences in DOC, suggesting that wildfire does not have long-lasting effects on water quality within this region (Burd et al. 2018; Tank et al. 2018). Any increase in organic acids associated with allochthonous acidic DOC compounds may lead to decreases in alkalinity and pH in boreal wildfire affected lakes (McEachern et al. 2000).

Wildfires may also increase levels of total suspended solids (TSS) in lakes. Increases in TSS from runoff may then lead to a decrease in dissolved oxygen (DO) levels (Tecle and Neary 2015). For instance, burned streams in southern Alaska were observed to have increased surface water TSS three years post-fire (Betts and Jones 2009). Increased runoff and sedimentation can also lead to increased light attenuation with lake colour turning yellowish-brown, which may affect productivity (Carignan et al. 2000; McEachern et al. 2000; Scrimgeour et al. 2001).

Methylmercury

The increased runoff from a watershed can increase trace metal contamination within northern water bodies, including toxic chemicals such as methyl-mercury (MeHg) (Garcia and Carignan 2005; Kelly et al. 2006). There are strong associations between carbon and MeHg in surface water (Stoken et al. 2016). With large areas of carbon burned, post-fire studies show even stronger relationships (Moreno et al. 2016; Giesler et al. 2017) suggesting that runoff with increases in carbon into water bodies will subsequently increase MeHg as carbon and mercury bond together.

Unlike inorganic mercury (Hg), organic MeHg binds to proteins and can pass through biological membranes, increasing bioaccumulation and biomagnification that can reach hazardous levels in upper-level predators such as fish (Bigham et al. 2017), aquatic invertebrate-eating bats (Little et al. 2015), and birds (Lavoie and Campbell 2018). Within boreal forests, Hg primarily resides within soils and is enriched through atmospheric deposition from anthropogenic activity (Smith-Downey et al. 2010; Amos et al. 2015). Within water bodies, Hg is deposited through atmospheric deposition in the form of precipitation, internal production, or water catchment runoff (Rudd 1995). When exposed to acidic conditions and precipitation, Hg can methylate in the soils of the water catchment by microorganisms and runoff into water bodies (Matilainen et al. 2001; Bigham et al. 2017). Typically, however, MeHg production occurs internally within lakes and wetlands, as the microbial bacteria that methylate Hg require wet conditions. Several key abiotic factors are thought to increase methylation rates, including increases of sulphur (S), low oxygen levels, and acidic or low pH conditions (Kampalath et al. 2013; Bigham et al. 2017; Paranjape and Hall 2017). Smoke plumes from boreal forests have elevated SO_4 and may be deposited into water catchment areas or water bodies themselves, increasing sulphur availability (Saarnio et al. 2010).

Most studies show dramatic increases in MeHg in higher trophic levels following a wildfire (Garcia and Carignan 2005; Kelly et al. 2006; Moreno et al. 2016). There are exceptions, however; five burned lakes were compared to five reference lakes in the boreal plains in Alberta and did not find elevated MeHg within zooplankton two years after the wildfire (Allen et al. 2005). This was primarily attributed to increased lake productivity which diluted mercury at the base of the food web through processes of biodilution associated with phytoplankton uptake (Allen et al. 2005). It should be noted that although no study has been conducted on macrophytes post-fire and mercury accumulation within the boreal forest, mercury uptake in macrophytes, and especially epiphytes, may be stimulated by increases in temperature and lower dissolved oxygen (Hamelin et al. 2015). Lakes located in southern, Quebec also failed to show an increase in zooplankton MeHg post-fire (Garcia and Carignan 1999). Both studies were not located within areas with significant permafrost (isolated and sporadic permafrost locations), which is considered to be a large reservoir for Hg (Korosi et al. 2015; Schuster et al. 2018). Increases in MeHg in subsistence fish are of great concern for northern communities. E.g. after the release of water into South Indian Lake in northern Manitoba (discontinuous permafrost zone) from the hydroelectric damn, thawing of the surrounding permafrost eroded sediment into the lake (Hecky et al. 1984). MeHg concentrations in fish subsequently increased to unsafe levels and the local indigenous who depended on fish from the lake as part of their main diet developed Minamata disease (Hecky et al. 1984). Therefore, it is important to understand the factors that influence MeHg levels in northern lakes so that appropriate information can be included in risk models (e.g. Workman 2019).

Primary Producers

Increases in productivity of the phytoplankton have been observed in some lakes post-fire. Lakes within Quebec experienced an increase in phytoplankton abundance post-fire, however, the species richness declined, and cyanobacteria became dominant (McEachern et al. 2002). Chlorophyll-*a* levels, a proxy for phytoplankton abundance, typically increases in response to the increase in nutrients experienced within boreal lake systems after wildfire (McEachern et al. 2000; Scrimgeour et al. 2001; Garcia and Carignan 2005; Kelly et al. 2006; Silins et al. 2014). However, this is not always the case (Lewis et al. 2014). Unfortunately, I could find no direct studies that have tested the effects of wildfire on macrophyte assemblages in boreal lakes.

Levels of $\delta^{13}C$ can be used as indicators of carbon sources following wildfires can be detected using stable isotope analyses. Allochthonous carbon input from terrestrial vegetation has a lower $\delta^{13}C$ signature, and thus decreased $\delta^{13}C$ indicates more carbon input from terrestrial runoff (Bade et al. 2004). Baseline sphaerid clams found in a burned lake in Northern Alberta had depleted $\delta^{13}C$ levels which were attributed to increased pelagic primary production and less benthic primary production (Kelly et al. 2006). However, wildfire-affected streams in the rocky mountains of northern Alberta had enriched $\delta^{13}C$ in benthic macroinvertebrates compared to reference streams (Silins et al. 2014). This shift suggested higher autochthonous algal primary production post-fire and subsequent cascading effects on the food web.

Zooplankton

Due to their fast generation time, ease of collection, sensitivity to environmental changes, and variety of feeding strategies, zooplankton communities are regarded as excellent bioindicators for short to long-term exposure to stressors (Ovaskainen et al. 2019). They are an important component of lake food webs, as they transfer energy from primary producers (e.g. algae and phytoplankton) to higher trophic consumers such as fish. Zooplankton can be classified as herbivores, carnivores, or omnivores (Barnett et al. 2007), but most exhibit omnivorous behavior provided their prey items are within edible size (Sommer and Sommer 2006).

After wildfires, the biomass and composition of zooplankton may change. Increased nutrients within lakes in the southern Canada boreal forest lakes and central Quebec lakes post-fire initially resulted in higher phytoplankton production within a year (Carignan et al. 2000; Patoine et al. 2000; Kelly et al. 2006). As a result of these increased food levels, the biomass and abundance of zooplankton increased (Patoine et al. 2000; Kelly et al. 2006). However, previous studies show that taxonomic richness and diversity of zooplankton communities appear to be resilient to the short-term influx of nutrients (Patoine 2002; Jalal et al. 2005). This suggests that zooplankton communities may be resilient to the effects of wildfire (Jalal et al. 2005).

Benthic Macroinvertebrates

Benthic invertebrates are also regarded as excellent bioindicators of short to long-term stressors, including changes in nutrients and exposure to harmful chemicals (Li et al. 2010). Benthic invertebrates also play an important role in the food web, as they link organic matter mainly from sediment and other benthic sources (e.g. detritus) to upper trophic levels such as fish. They are commonly used as the bioindicators of wildfire disturbance in boreal water bodies (Scrimgeour et al. 2001; Lewis et al. 2014; Silins et al. 2014). As a measurement of change in benthic invertebrate communities, feeding preference classed into functional feeding groups (FFG) is commonly used (Merritt et al. 2008). A generalized overview of these FFGs categorizes chewers of vascular macrophytes and wood detritus as shredders, suspension feeders of particulate organic material (POM) as filter-collectors, sediment feeders as gathering-collectors, organic or mineral scrapers of attached matter as scrapers, and carnivores of whole animal parts as predators (Merritt et al. 2008). A subcategory of predators where the organisms ingest tissue fluids can be categorized as parasites.

An alternative approach to analysis looks at sensitive groups of species. Differences in sensitive species of Ephemeroptera, Plecoptera, and Trichoptera (EPT) has been widely used as a measurement to assess biological shifts and stressors within benthic macroinvertebrate communities. Streams in northern Alberta, for instance, shifted in response to wildfire disturbances from more sensitive invertebrates (EPT) to more hardy generalists (Dipterans; measured in EPT/D ratio) (Silins et al. 2014).

Increases in primary production with increased nutrient input can have major effects on benthic invertebrate abundance and trophic compositions (Clarke et al. 1997), however, after wildfire, the response may vary. Within discontinuous permafrost areas with peat soils of the Yukon Flats region (Alaska, USA), amphipod abundance and predator benthic invertebrate abundances increased after wildfires, however, overall abundance did not increase (Lewis et al. 2014). Lewis et al. (2014) argued that the increase in amphipod abundance occurred because amphipods tend to be generalist feeders, not specialized, and therefore were able to adapt to the shifting environment through cannibalistic/predatory behaviours in conjunction with their normal feeding behaviour. Higher biomass and abundance of benthic invertebrates were observed in Alberta streams affected by wildfires up to five years post-fire (Silins et al. 2014) while four lakes within the northern Alberta subarctic region had elevated biomass of aquatic invertebrates that persisted for 15-20 years post-fire (Scrimgeour et al. 2001). Finally, streams in northern Alberta also had higher abundances, biomass, and species richness (Silins et al. 2014). Overall, these studies suggest that benthic invertebrate abundance will increase in most cases and that increases in biomass and richness are also possible.

Study Objectives

Wildfires represent a major natural disturbance in the boreal region, and its frequency is increasing in response to climate change. In 2014, severe megafires spread throughout the Canadian Northwest Territories (NWT), burning a landscape rich in lakes and ponds. Lakes found in burned areas often experience large increases in nutrients and metals following wildfires which can have negative impacts on aquatic communities, including sport and subsistence fish.

The overall goal of my thesis work is to describe lake ecosystems within the subarctic through observing the aquatic invertebrates and water quality within lakes impacted by wildfire in the Sahtú Settlement Area (SSA) of the NWT. To achieve this goal, I will use the reference condition approach (RCA) to pursue the two objectives listed below. I have also included my hypotheses under each objective.

1. Determine if water quality in northern boreal forest lakes is related to the history of wildfire in their catchments.

Hypothesis 1: Lakes affected by burns in their catchments would show water quality changes described in past studies, including pH levels, light penetration, and elevated nutrients, productivity, and metals (Garcia and Carignan 1999, 2005; Carignan et al. 2000; Lamontagne et al. 2000; McEachern et al. 2000; Scrimgeour et al. 2001; Allen et al. 2005; Kelly et al. 2006; Silins et al. 2014).

2. Examine the relationship between invertebrate communities and water quality variables in a group of lakes to determine if water quality variables affected by wildfires might be associated (predictor variable correlated with wildfire) with the structure of invertebrate communities.

   Hypothesis 2: the structure of benthic/macroinvertebrate and zooplankton communities (richness, abundance, functional feeding groups) would show differences associated with changes in water quality such as increased productivity and food availability and decreased pH levels, and light penetration (Scrimgeour et al. 2001; McEachern et al. 2002; Jalal et al. 2005; Lewis et al. 2014; Musetta-Lambert et al. 2019).

Site Location

The Sahtú Settlement Area (SSA) extends approximately 150 km southward from the start of the Mackenzie River Delta to just south of Great Bear Lake (Mochnacz et al. 2007). There are five established communities, Colville Lake, Déline, Fort Good Hope, Tulita, and Norman Wells. Norman Wells, Tulita, and Fort Good Hope are located along the Mackenzie River and thus are affected by any changes to the river system. The Mackenzie River traverses 520 km north across the SSA, where it receives freshwater input from tributaries ranging in size from ephemeral streams to large rivers. Tributaries along the east side of the Mackenzie River in the SSA originate from the Franklin Mountains, while those on the west side originate from the Mackenzie Mountains. This region has experienced the greatest warming in Canada during the last 50 years, with the mean annual temperature increasing by >1.5°C, resulting in possible increases in the frequency of wildfires related to drought (Woo et al. 2007).

The area surrounding Norman Wells lies within the Norman Range Low Subarctic (LS) ecoregion at an average elevation of 300 m above sea level (Government of Northwest Territories 2009) within the Taiga plans. Vegetation consists of a mixture of trembling aspen (*Populus tremuloides*), Alaska paper birch (*Betula neoalaskana*), white spruce (*Picea glauca*) and black spruce (*Picea mariana*) with large quantities of shrubs such as Labrador tea (*Rhododendron groenlandicum*) and dominated by lichen (*Cladina spp*.) (Gibson et al. 2018). Organic and mineral-rich turbic crysol soil is the primary soil type within this region (Soil Classification Working Group 1998). Permafrost is found in 48-75% of the moss/peat rich soil surface (discontinuous permafrost) within the study area (Golder Associates 2015). Different surficial geological zones occurred in the study area of the organic-rich: till blanket, till veneer, and fine-grained (glacio)lacustrine, and surficial geological zones with exposed bedrock of colluvial rubble (Golder Associates 2015).

Located within the SSA are thousands of small water bodies, including lakes, ponds, fens, and bogs (Golder Associates 2015). Shallow lakes are important indicators for a variety of environmental stressors as they may be more sensitive to change than deep lakes (Smol 2016). As a result, they are considered to be gauges of stress in high-latitude aquatic ecosystems (Rouse et al. 1997; Smol 2016). Therefore, small lake habitats represent the ideal system in which to study the extent to which wildfires affect water quality and aquatic invertebrates in discontinuous permafrost and humic peat soil sub-arctic lakes in the SSA.

The Reference Condition Approach

The reference condition approach (RCA), is one of the most commonly used and effective techniques for biomonitoring and detecting impacts on ecosystems and habitat (Reynoldson et al. 1997, 2001; Reece and Richardson 1999). Ideally, within laboratory experiments, variables are controlled except for the independent variable of interest. In natural environments, not every variable can be controlled, requiring a different approach. Most RCA predictive models are formulated using benthic invertebrate data and the associated environmental (physical, chemical, and biological) data. Typically, many reference habitat sites are chosen to capture natural variability and identify confounding variables that may be affecting the invertebrates. In this regard, reference sites should be as similar as possible to the test sites to minimize differences in invertebrate communities that are associated with confounding variables. A better approach would be to not call it a “reference condition” but instead the “best reference condition” possible (Reynoldson et al. 1997). The reference condition itself is representative of a group of sites that are minimally impacted by physical, chemical, or biological differences. After creating the predictive model, if the test sites differ from the reference sites, it can be concluded that the test site has been impacted (Reece and Richardson 1999).

References

Allen, E.W., Prepas, E.E., Gabos, S., Strachan, W.M., and Zhang, W. 2005. Methyl mercury concentrations in macroinvertebrates and fish from burned and undisturbed lakes on the

Boreal Plain. Can. J. Fish. Aquat. Sci. **62**(9): 1963–1977. doi:10.1139/f05-103.

Amos, H.M., Sonke, J.E., Obrist, D., Robins, N., Hagan, N., Horowitz, H.M., Mason, R.P., Witt, M., Hedgecock, I.M., Corbitt, E.S., and Sunderland, E.M. 2015. Observational and modeling constraints on global anthropogenic enrichment of mercury. Environ. Sci. Technol. **49**(7): 4036–4047. doi:10.1021/es5058665.

Bade, D.L., Carpenter, S.R., Cole, J.J., Hanson, P.C., and Hesslein, R.H. 2004. Controls of d13C-DIC in lakes : Geochemistry , lake metabolism , and morphometry Controls of d13 C-DIC in lakes : Geochemistry , lake metabolism , and morphometry. Limnol. Oceanogr. **49**(4): 1160–1172. doi:10.4319/lo.2004.49.4.1160.

Badiou, P., Jacobs, J., Kerr, J., Manseau, M., Orions, G., Pimm, S., Raven, P., Root, T., Roulet, N., Schaefer, J., Schindler, D., Strittholt, J., Turner, N., and Weaver, A. 2011. A Forest of Blue: Canada's Boreal. Seattle, WA.

Barnett, A.J., Finlay, K., and Beisner, B.E. 2007. Functional diversity of crustacean zooplankton communities: Towards a trait-based classification. Freshw. Biol. **52**(5): 796–813. doi:10.1111/j.1365-2427.2007.01733.x.

Bayley, S.E., Schindler, D.W., Beaty, K.G., Parker, B.R., and Stainton, M.P. 1992. Effects of multiple fires on nutrient yields from streams draining boreal forest and fen watersheds: nitrogen and phosphorus. Can. J. Fish. Aquat. Sci. **49**(3): 584–596. doi:10.1139/f92-068.

Betts, E.F., and Jones, J.B. 2009. Impact of Wildfire on Stream Nutrient Chemistry and Ecosystem Metabolism in Boreal Forest Catchments of Interior Alaska. Arctic, Antarct. Alp. Res. **41**(4): 407–417. doi:10.1657/1938-4246-41.4.407.

Beven, K. 2004. Robert E. Horton's perceptual model of infiltration processes. Hydrol. Process. **18**(17): 3447–3460. doi:10.1002/hyp.5740.

Bigham, G.N., Murray, K.J., Masue-slowey, Y., and Henry, E.A. 2017. Biogeochemical Controls on Methylmercury in Soils and Sediments : Implications for Site Management. **13**(2): 249–263. doi:10.1002/ieam.1822.

Bond-Lamberty, B., Peckham, S.D., Gower, S.T., and Ewers, B.E. 2009. Effects of fire on regional evapotranspiration in the central Canadian boreal forest. Glob. Chang. Biol. **15**(5): 1242–1254. doi:10.1111/j.1365-2486.2008.01776.x.

Boulanger, Y., Gauthier, S., Burton, P.J., and Vaillancourt, M.A. 2012. An alternative fire regime zonation for Canada. Int. J. Wildl. Fire **21**(8): 1052–1064. doi:10.1071/WF11073.

Bring, A., Fedorova, I., Dibike, Y., Hinzman, L., Mård, J., Mernild, S.H., Prowse, T., Semenova, O., Stuefer, S.L., and Woo, M.K. 2016. Arctic terrestrial hydrology: A synthesis of processes, regional effects, and research challenges. J. Geophys. Res. G Biogeosciences **121**(3): 621–649. doi:10.1002/2015JG003131.

Burd, K., Tank, S.E., Dion, N., Quinton, W.L., Spence, C., Tanentzap, A.J., and Olefeldt, D. 2018. Seasonal shifts in export of DOC and nutrients from burned and unburned peatland-rich catchments, Northwest Territories, Canada. Hydrol. Earth Syst. Sci. Discuss.: 1–32. doi:10.5194/hess-2018-253.

Burles, K., and Boon, S. 2011. Snowmelt energy balance in a burned forest plot, Crowsnest Pass, Alberta, Canada. Hydrol. Process. **25**(19): 3012–3029. doi:10.1002/hyp.8067.

Carignan, R., D'Arcy, P., and Lamontagne, S. 2000. Comparative impacts of fire and forest harvesting on water quality in Boreal Shield lakes. Can. J. Fish. Aquat. Sci. **57**(SUPPL. 2): 105–117. doi:10.1139/f00-125.

Clarke, K.D., Knoechel, R., and Ryan, P.M. 1997. Influence of trophic role and life-cycle duration on timing and magnitude of benthic macroinvertebrate response to whole-lake enrichment. Can. J. Fish. Aquat. Sci. **54**(1): 89–95. doi:10.1139/cjfas-54-1-89.

Flannigan, M., Stocks, B., Turetsky, M., and Wotton, M. 2009. Impacts of climate change on fire activity and fire management in the circumboreal forest. Glob. Chang. Biol. **15**(3): 549–560. doi:10.1111/j.1365-2486.2008.01660.x.

Flannigan, M.D., Logan, K.A., Amiro, B.D., Skinner, W.R., and Stocks, B.J. 2005. Future area burned in Canada. Clim. Change **72**(1–2): 1–16. doi:10.1007/s10584-005-5935-y.

Garcia, E., and Carignan, R. 1999. Impact of wildfire and clear-cutting in the boreal forest on methyl mercury in zooplankton. Can. J. Fish. Aquat. Sci. **56**(2): 339–345. doi:10.1139/f98-164.

Garcia, E., and Carignan, R. 2005. Mercury concentrations in fish from forest harvesting and fire-impacted Canadian Boreal lakes compared using stable isotopes of nitrogen. Environ. Toxicol. Chem. **24**(3): 685–693. doi:10.1897/04-065R.1.

Gibson, C.M., Chasmer, L.E., Thompson, D.K., Quinton, W.L., Flannigan, M.D., and Olefeldt, D. 2018. Wildfire as a major driver of recent permafrost thaw in boreal peatlands. Nat. Commun. **9**: 1–9. doi:10.1038/s41467-018-05457-1.

Giesler, R., Clemmensen, K.E., Wardle, D.A., Klaminder, J., and Bindler, R. 2017. Boreal forests sequester large amounts of mercury over millennial time scales in the absence of wildfire. Environ. Sci. Technol. **51**(5): 2621–2627. doi:10.1021/acs.est.6b06369.

Gleason, K.E., McConnell, J.R., Arienzo, M.M., Chellman, N., and Calvin, W.M. 2019. Four-fold increase in solar forcing on snow in western U.S. burned forests since 1999. Nat. Commun. **10**(1): 1–8. Springer US. doi:10.1038/s41467-019-09935-y.

Golder Associates. 2015. Central Mackenzie surface water and groundwater baseline assessment. Available from https://mvlwb.com/registry/?f[]=region:Sahtu.

Goulden, M.L., Mcmillan, A.M.S., Winston, G.C., Rocha, A. V., Manies, K.L., Harden, J.W., and Bond-Lamberty, B.P. 2011. Patterns of NPP, GPP, respiration, and NEP during boreal forest succession. Glob. Chang. Biol. **17**(2): 855–871. doi:10.1111/j.1365-2486.2010.02274.x.

Government of Northwest Territories. 2009. Ecological regions of the Northwest Territories: Taiga plains.

Hamelin, S., Planas, D., and Amyot, M. 2015. Spatio-temporal variations in biomass and mercury concentrations of epiphytic biofilms and their host in a large river wetland (Lake St. Pierre, Qc, Canada). Environ. Pollut. **197**: 221–230. doi:10.1016/j.envpol.2014.11.007.

Hanes, C.C., Wang, X., Jain, P., Parisien, M.A., Little, J.M., and Flannigan, M.D. 2019. Fire-regime changes in canada over the last half century. Can. J. For. Res. **49**(3): 256–269. doi:10.1139/cjfr-2018-0293.

Hecky, R.E., Newbury, R.W., Bodaly, R.A., Patalas, K., and Rosenberg, D.M. 1984. Environmental impact prediction and assessment: the Southern Indian Lake experience. Can. J. Fish. Aquat. Sci. **41**(4): 720–732. doi:10.1139/f84-084.

Helbig, M., Pappas, C., and Sonnentag, O. 2016. Permafrost thaw and wildfire: Equally important drivers of boreal tree cover changes in the Taiga Plains, Canada. Geophys. Res. Lett. **43**(4): 1598–1606. doi:10.1002/2015GL067193.

Jalal, W., Pinel-Alloul, B., and Methot, G. 2005. Mid-term study of the ecological impacts of forest fires and timber harvesting on zooplankton communities in lakes of the boreal ecozone. Rev. Des Sci. L’eau **18**: 221–248.

Johnstone, J.F., and Chapin, F.S. 2006. Effects of soil burn severity on post-fire tree recruitment in boreal forest. Ecosystems **9**(1): 14–31. doi:10.1007/s10021-004-0042-x.

Kampalath, R.A., Lin, C.C., and Jay, J.A. 2013. Influences of zero-valent sulfur on mercury methylation in bacterial cocultures. Water. Air. Soil Pollut. **224**(2). doi:10.1007/s11270-012-1399-7.

Kasischke, E.S., and Turetsky, M.R. 2006. Recent changes in the fire regime across the North American boreal region - Spatial and temporal patterns of burning across Canada and Alaska. Geophys. Res. Lett. **33**(9). doi:10.1029/2006GL025677.

Kasischke, E.S., Verbyla, D.L., Rupp, T.S., McGuire, A.D., Murphy, K.A., Jandt, R., Barnes, J.L., Hoy, E.E., Duffy, P.A., Calef, M., and Turetsky, M.R. 2010. Alaska’s changing fire regime — implications for the vulnerability of its boreal forests. Can. J. For. Res. **40**(7): 1313–1324. doi:10.1139/X10-098.

Keeley, J.E. 2009. Fire intensity, fire severity and burn severity: A brief review and suggested usage. Int. J. Wildl. Fire **18**(1): 116–126. doi:10.1071/WF07049.

Kelly, E.N., Schindler, D.W., St. Louis, V.L., Donald, D.B., and Vladicka, K.E. 2006. Forest fire increases mercury accumulation by fishes via food web restructuring and increased mercury inputs. Proc. Natl. Acad. Sci. **103**(51): 19380–19385. doi:10.1073/pnas.0609798104.

Kershaw, G.P. 2001. Snowpack characteristics following wildfire on a simulated transport corridor and adjacent subarctic forest, Tulita, N.W.T., Canada. Arctic, Antarct. Alp. Res. **33**(2): 131–139. doi:10.2307/1552213.

Kokelj, S. V., Burn, C.R., and Tarnocai, C. 2007. The structure and dynamics of earth hummocks in the subarctic forest near Inuvik, Northwest Territories, Canada. Arctic, Antarct. Alp. Res. **39**(1): 99–109. doi:10.1657/1523-0430(2007)39[99:TSADOE]2.0.CO;2.

Kokelj, S. V., Jenkins, R.E., Milburn, D., Burn, C.R., and Snow, N. 2005. The influence of thermokarst disturbance on the water quality of small upland lakes, Mackenzie Delta region, Northwest Territories, Canada. Permafr. Periglac. Process. **16**(4): 343–353. doi:10.1002/ppp.536.

Korosi, J.B., Mcdonald, J., Coleman, K.A., Palmer, M.J., Smol, J.P., Simpson, M.J., and Blais, J.M. 2015. Long-term changes in organic matter and mercury transport to lakes in the sporadic discontinuous permafrost zone related to peat subsidence. Limnol. Oceanogr. **60**(5): 1550–1561. doi:10.1002/lno.10116.

Lamontagne, S., Carignan, R., D'Arcy, P., Prairie, Y.T., and Pare, D. 2000. Element export in runoff from eastern Canadian Boreal Shield drainage basins following forest harvesting and wildfires. Can. J. Fish. Aquat. Sci. **57**: 118–128. doi:10.1139/f00-108.

Lavoie, M., and Mack, M.C. 2012. Spatial heterogeneity of understory vegetation and soil in an Alaskan upland boreal forest fire chronosequence. Biogeochemistry **107**(1–3): 227–239.

doi:10.1007/s10533-010-9547-x.

Lavoie, R.A., and Campbell, L.M. 2018. Mercury concentrations in double-crested cormorant chicks across Canada. Arch. Environ. Contam. Toxicol. **75**(1): 111–120. doi:10.1007/s00244-018-0533-y.

Lebedeva, L., Semenova, O., and Volkova, N. 2014. Assessment of wildfire impact on hydrological extremes in eastern Siberia. IAHS-AISH Proc. Reports **363**(October): 90–95.

Lewis, T.L., Lindberg, M.S., Schmutz, J.A., and Bertram, M.R. 2014. Multi-trophic resilience of boreal lake ecosystems to forest fires. **95**(5): 1253–1263.

Li, L., Zheng, B., and Liu, L. 2010. Biomonitoring and bioindicators used for river ecosystems: Definitions, approaches and trends. Procedia Environ. Sci. **2**: 1510–1524. doi:10.1016/j.proenv.2010.10.164.

Little, M.E., Burgess, N.M., Broders, H.G., and Campbell, L.M. 2015. Mercury in little brown bat (Myotis lucifugus) maternity colonies and its correlation with freshwater acidity in Nova Scotia, Canada. Environ. Sci. Technol. **49**(4): 2059–2065. doi:10.1021/es5050375.

Liu, S., Bond-Lamberty, B., Hicke, J.A., Vargas, R., Zhao, S., Chen, J., Edburg, S.L., Hu, Y., Liu, J., McGuire, A.D., Xiao, J., Keane, R., Yuan, W., Tang, J., Luo, Y., Potter, C., and Oeding, J. 2011. Simulating the impacts of disturbances on forest carbon cycling in North America: Processes, data, models, and challenges. J. Geophys. Res. Biogeosciences **116**(4): 1–22. doi:10.1029/2010JG001585.

Mathieu, E. 2018. The effects of fire on snow accumulation , snowmelt and ground thaw on a peat plateau in subarctic Canada.

Matilainen, T., Verta, M., Korhonen, H., and Niemi, M. 2001. Behavir of mercury in soil profiles: Impact of increased precipitation, acidity, and fertilization on mercury methylation. Water, Air, Soil Pollutiom **125**: 105–119.

McEachern, P., Prepas, E.E., Gibson, J.J., and Dinsmore, W.P. 2000. Forest fire induced impacts on phosphorus, nitrogen, and chlorophyll a concentrations in boreal subarctic lakes of northern Alberta. Can. J. Fish. Aquat. Sci. **57**(S2): 73–81. doi:10.1139/cjfas-57-S2-73.

McEachern, P., Prepas, E.E., and Planas, D. 2002. Phytoplankton in boreal subarctic lakes following enhanced phosphorus loading from forest fire: Impacts on species richness, nitrogen and light limitation. Lake Reserv. Manag. **18**(2): 138–148. doi:10.1080/07438140209354144.

Merritt, R.W., Cummins, K.W., and Berg, M.B. 2008. An introduction to the aquatic insects of North America. *In* 4th edition. Kendall Hunt Pub Co, Dubuque, Iowa.

Miquelajauregui, Y., Cumming, S.G., and Gauthier, S. 2016. Modelling variable fire severity in boreal forests: Effects of fire intensity and stand structure. PLoS One **11**(2). doi:10.1371/journal.pone.0150073.

Mochnacz, N.J., Reist, J.D., and Rt, M. 2007. Biological and habitat data for fish collected during stream surveys in the Sahtu Settlement Region , Northwest Territories , 2006. Can. Data Rep. Fish Aquat. Sci. (1189): 1–40.

Moreno, C.E., Fjeld, E., and Lydersen, E. 2016. The effects of wildfire on mercury and stable isotopes (δ15N, δ13C) in water and biota of small boreal, acidic lakes in southern Norway. Environ. Monit. Assess. **188**(3): 1–23. doi:10.1007/s10661-016-5148-z.

Musetta-Lambert, J., Kreutzweiser, D., and Sibley, P. 2019. Influence of wildfire and harvesting on aquatic and terrestrial invertebrate drift patterns in boreal headwater streams. Hydrobiologia **834**(1): 27–45. Springer International Publishing. doi:10.1007/s10750-019-3907-x.

Ovaskainen, O., Weigel, B., Potyutko, O., and Buyvolov, Y. 2019. Long-term shifts in water quality show scale-dependent bioindicator responses across Russia – Insights from 40 year-long bioindicator monitoring program. Ecol. Indic. **98**(2019): 476–482. Elsevier. doi:10.1016/j.ecolind.2018.11.027.

Paranjape, A.R., and Hall, B.D. 2017. Recent advances in the study of mercury methylation in aquatic systems. Facets **2**(1): 85–119. doi:10.1139/facets-2016-0027.

Patoine, A. 2002. Influence of catchment deforestationby logging and natural forest fires on crustacean community size structure in lakes of the Eastern Boreal Canadian forest. J. Plankton Res. **24**(6): 601–616. doi:10.1093/plankt/24.6.601.

Patoine, A., Pinel-Alloul, B., Prepas, E.E., and Carignan, R. 2000. Do logging and forest fires influence zooplankton biomass in Canadian Boreal Shield Lakes? Can. J. Fish. Aquat. Sci. **57**: 155–164. doi:10.1139/f00-105.

Pimentel, R., and Arheimer, B. 2018. Wildfire impact on Boreal hydrology: empirical study of the Västmanland fire 2014 (Sweden). Hydrol. Earth Syst. Sci. Discuss. (August): 1–26. doi:10.5194/hess-2018-387.

Pomeroy, J.W., Granger, R., Pietroniro, A., Toth, B., and Hedstrom, N. 1999. Classification of the boreal forest for hydrological processes. Ninth Int. Boreal For. Res. Assoc. Conf.: 49–

59.

Reece, P.F., and Richardson, J.S. 1999. Biomonitoring with the reference condition approach for the detection of aquatic ecosystems at risk. Proc. a Conf. Biol. Manag. species Habitats Risk: 549–552.

Reynoldson, T.B., Norris, R.H., Resh, V.H., Day, K.E., and Rosenberg, D.M. 1997. The reference condition: a comparison of multimetric and multivariate approaches to assess water-quality impairment using benthic macroinvertebrates. J. North Am. Benthol. Soc. **16**(4): 833–852. doi:10.2307/1468175.

Reynoldson, T.B., Rosenberg, D.M., and Resh, V.H. 2001. Comparison of models predicting invertebrate assemblages for biomonitoring in the Fraser River catchment, British Columbia. Can. J. Fish. Aquat. Sci. **58**(7): 1395–1410. doi:10.1139/f01-075.

Robinne, F.-N., Hallema, D.W., Bladon, K.D., and Buttle, J.M. 2020. Wildfire impacts on hydrologic ecosystem services in North American high-latitude forests: A scoping review. J. Hydrol. **581**: 1–33. doi:10.1016/j.jhydrol.2019.124360.

Rouse, W.R., Douglas, M.S.V., Hecky, R.E., Hershey, A.E., Kling, G.W., Lesack, L., Marsh, P., Mcdonald, M., Nicholson, B.J., Roulet, N.T., and Smol, J.P. 1997. Effects of climate change on the freshwaters of arctic and subarctic North America. Hydrol. Process. **11**(8): 873–902. doi:10.1002/(SICI)1099-1085(19970630)11:8<873::AID-HYP510>3.0.CO;2-6.

Rudd, J.W.M. 1995. Sources of methyl mercury to freshwater ecosystems: A review. Water, Air, Soil Pollut. **80**(1–4): 697–713. doi:10.1007/BF01189722.

Saarnio, K., Aurela, M., Timonen, H., Saarikoski, S., Teinilä, K., Mäkelä, T., Sofiev, M., Koskinen, J., Aalto, P.P., Kulmala, M., Kukkonen, J., and Hillamo, R. 2010. Chemical composition of fine particles in fresh smoke plumes from boreal wild-land fires in Europe. Sci. Total Environ. **408**(12): 2527–2542. doi:10.1016/j.scitotenv.2010.03.010.

Schuster, P.F., Schaefer, K.M., Aiken, G.R., Antweiler, R.C., Dewild, J.F., Gryziec, J.D., Gusmeroli, A., Hugelius, G., Jafarov, E., Krabbenhoft, D.P., Liu, L., Herman-Mercer, N., Mu, C., Roth, D.A., Schaefer, T., Striegl, R.G., Wickland, K.P., and Zhang, T. 2018. Permafrost stores a globally significant amount of mercury. Geophys. Res. Lett. **45**(3): 1463–1471. doi:10.1002/2017GL075571.

Scrimgeour, G.J., Tonn, W.M., Paszkowski, C.A., and Goater, C. 2001. Benthic macroinvertebrate biomass and wildfires: Evidence for enrichment of boreal subarctic lakes. Freshw. Biol. **46**(3): 367–378. doi:10.1046/j.1365-2427.2001.00682.x.

Seibert, J., McDonnell, J.J., and Woodsmith, R.D. 2010. Effects of wildfire on catchment runoff response: A modelling approach to detect changes in snow-dominated forested catchments. Hydrol. Res. **41**(5): 378–390. doi:10.2166/nh.2010.036.

Shu-ren, Y. 2003. Effects of fire disturbance on forest hydrology. J. For. Res. **14**(4): 331–334. doi:10.1007/bf02857863.

Silins, U., Bladon, K.D., Kelly, E.N., Esch, E., Spence, J.R., Stone, M., Emelko, M.B., Boon, S., Wagner, M.J., Williams, C.H.S., and Tichkowsky, I. 2014. Five-year legacy of wildfire and salvage logging impacts on nutrient runoff and aquatic plant, invertebrate, and fish productivity. Ecohydrology **7**(6): 1508–1523. doi:10.1002/eco.1474.

Smith-Downey, N. V, Sunderland, E.M., and Jacob, D.J. 2010. Anthropogenic impacts on global storage and emissions of mercury from terrestrial soils: Insights from a new global model. J. Geophys. Res. **115**(G03008): 1–11. doi:10.1029/2009JG001124.

Smol, J.P. 2016. Arctic and Sub-Arctic shallow lakes in a multiple-stressor world: a paleoecological perspective. Hydrobiologia **778**(1): 253–272. Springer International Publishing. doi:10.1007/s10750-015-2543-3.

Soil Classification Working Group. 1998. The Canadian system of soil classification, 3rd ed. Agriculture and Agri-Food Canada Publication 1646.

Sommer, U., and Sommer, F. 2006. Cladocerans versus copepods: The cause of contrasting top-down controls on freshwater and marine phytoplankton. Oecologia **147**(2): 183–194. doi:10.1007/s00442-005-0320-0.

Stednick, J.D., Tripp, L.N., and McDonald, R.J. 1982. Slash burning effects on soil and water chemistry in southeastern Alaska. J. Soil Water Conserv. **37**(2): 126–128.

Stocks, B.J., Mason, J.A., Todd, J.B., Bosch, E.M., Wotton, B.M., Amiro, B.D., Flannigan, M.D., Hirsch, K.G., Logan, K.A., Martell, D.L., and Skinner, W.R. 2003. Large forest fires in Canada, 1959-1997. J. Geophys. Res. D Atmos. **108**(1). doi:10.1029/2001jd000484.

Stoken, O.M., Riscassi, A.L., and Scanlon, T.M. 2016. Association of dissolved mercury with dissolved organic carbon in U.S. rivers and streams: The role of watershed soil organic carbon. Water Resour. Res. **52**: 3040–3051. doi:10.1002/2015WR017849.

Tank, S., Olefeldt, D., Quinton, W., Spence, C., Dion, N., Ackley, C., Burd, K., Hutchins, R., and Mengistu, S. 2018. Fire in the Arctic: The effect of wildfire across diverse aquatic ecosystems of the Northwest Territories. Polar Knowl. Aqhaliat Rep. **1**(1): 31–38.

doi:10.35298/pkc.2018.04.

Tecle, A., and Neary, D. 2015. Water Quality Impacts of Forest Fires. J. Pollut. Eff. Control **03**(03). doi:10.4172/2375-4397.1000140.

Thompson, D.K., and Waddington, J.M. 2013. Wildfire effects on vadose zone hydrology in forested boreal peatland microforms. J. Hydrol. **486**: 48–56. doi:10.1016/j.jhydrol.2013.01.014.

Torre Jorgenson, M., Harden, J., Kanevskiy, M., O'Donnell, J., Wickland, K., Ewing, S., Manies, K., Zhuang, Q., Shur, Y., Striegl, R., and Koch, J. 2013. Reorganization of vegetation, hydrology and soil carbon after permafrost degradation across heterogeneous boreal landscapes. Environ. Res. Lett. **8**(3). doi:10.1088/1748-9326/8/3/035017.

Veraverbeke, S., Rogers, B.M., Goulden, M.L., Jandt, R.R., Miller, C.E., Wiggins, E.B., and Randerson, J.T. 2017. Lightning as a major driver of recent large fire years in North American boreal forests. Nat. Clim. Chang. **7**(7): 529–534. doi:10.1038/nclimate3329.

Vieira, D.C.S., Fernández, C., Vega, J.A., and Keizer, J.J. 2015. Does soil burn severity affect the post-fire runoff and interrill erosion response? A review based on meta-analysis of field rainfall simulation data. J. Hydrol. **523**(2015): 452–464. doi:10.1016/j.jhydrol.2015.01.071.

Wagner, M.J., Bladon, K.D., Silins, U., Williams, C.H.S., Martens, A.M., Boon, S., MacDonald, R.J., Stone, M., Emelko, M.B., and Anderson, A. 2014. Catchment-scale stream temperature response to land disturbance by wildfire governed by surface-subsurface energy exchange and atmospheric controls. J. Hydrol. **517**: 328–338. doi:10.1016/j.jhydrol.2014.05.006.

Whitman, E., Parisien, M.A., Thompson, D.K., Hall, R.J., Skakun, R.S., and Flannigan, M.D. 2018. Variability and drivers of burn severity in the northwestern Canadian boreal forest: Ecosphere **9**(2): 1–26. doi:10.1002/ecs2.2128.

Williams, A.P., Abatzoglou, J.T., Gershunov, A., Guzman-Morales, J., Bishop, D.A., Balch, J.K., and Lettenmaier, D.P. 2019. Observed impacts of anthropogenic climate change on wildfire in California. Earth's Futur. **7**(8): 892–910. doi:10.1029/2019ef001210.

Woo, M.K., Modeste, P., Martz, L., Blondin, J., Kochtubajda, B., Tutcho, D., Gyakum, J., Takazo, A., Spence, C., Tutcho, J., Di Cenzo, P., Kenny, G., Stone, J., Neyelle, I., Baptiste, G., Modeste, M., Kenny, B., and Modeste, W. 2007. Science meets traditional knowledge: Water and climate in the Sahtu (Great Bear Lake) Region, Northwest Territories, Canada. Arctic **60**(1): 37–46. doi:10.14430/arctic263.

Workman, P. 2019. NWT mercury predictors in lakes. Available from http://www.geomatics.gov.nt.ca/mercury_predictors_in_fish.aspx [accessed 29 October 2019].

Worrall, F., Moody, C.S., Clay, G.D., Burt, T.P., and Rose, R. 2017. The flux of organic matter through a peatland ecosystem: The role of cellulose, lignin, and their control of the ecosystem oxidation state. J. Geophys. Res. Biogeosciences **122**(7): 1655–1671. doi:10.1002/2016JG003697.

Yue, X., Mickley, L.J., Logan, J.A., Hudman, R.C., Val Martin, M., and Yantosca, R.M. 2015. Impact of 2050 climate change on North American wildfire: consequences for ozone air quality. Atmos. Chem. Phys. Discuss. **15**(9): 13867–13921. doi:10.5194/acpd-15-13867-2015.

Zhang, Q., and Chen, W. 2007. Fire cycle of the Canada’s boreal region and its potential response to global change. J. For. Res. **18**(1): 55–61. doi:10.1007/s11676-007-0010-3.

Chapter 2: Field Study

How do wildfires affect water quality and invertebrates in subarctic lakes?

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Abstract (175 words)

Fires are a natural phenomenon in the boreal forest, but their frequency is expected to increase over the coming century. Wildfire may affect water quality and invertebrates in lakes, but there have been few studies in the northern boreal forest to describe these impacts. We collected data on water quality, macrophytes, and invertebrates from 20 lakes in the Sahtú Settlement Area of the Northwest Territories. Nine lakes were affected by wildfire in their catchments 4-5 years before data collection, while eleven were not. Our results showed that few water quality variables were related to burn history. However, remote sensing and field observations suggest that macrophyte biomass was affected by burns and was a significant predictor of invertebrate community composition. Burn history was an important predictor of the richness and abundance of invertebrates, but natural variability in lake properties was more important for explaining differences among lakes. Our results indicate that differences were detected in lakes affected by fire 4-5 years after the fire, suggesting that fires could have important impacts on subarctic boreal lakes.

Introduction

Climate change is leading to an increase in the frequency of disturbance in northern terrestrial and aquatic ecosystems, including increases in biological pathogens (Weed et al. 2013), changing hydrological regimes (Prowse et al. 2006), thawing of permafrost (Lantz and Kokelj 2008), and an increased frequency of wildfires (Stocks et al. 2003; Kasischke et al. 2010). Wildfires are a natural disturbance in the boreal forest, but recent studies suggest that increases in wildfires in the north may be driven by a positive feedback loop involving increased temperatures and precipitation, the northward expansion of forest, and relationships between surface energy fluxes from forests and increased storm events that lead to lightning ignitions (Veraverbeke et al. 2017). Although human ignitions account for approximately half of wildfires, evidence suggests that lightning ignitions have been trending upwards during the last four decades, while human ignitions are steady or declining (Hanes et al. 2019). As a result of climate-driven changes in fire frequency, the area of Canada's forests affected by wildfire is projected to increase over threefold by the mid century (Yue et al. 2015), and the interval between wildfires is expected to decrease significantly (Zhang and Chen 2007; Flannigan et al. 2009).

Increases in fire frequency and severity in the northern boreal forest could have a significant impact on water quality in lakes and ponds, as alterations to the vegetation in water catchments can reduce evapotranspiration (Bond-Lamberty et al. 2009) and increase snowpack accumulation, influencing runoff to lakes (Pomeroy et al. 1999; Mathieu 2018). Most studies have found that runoff from lake catchments after a wildfire is enriched in allochthonous nutrients, leading to increases in total nitrogen (Bayley et al. 1992; Garcia and Carignan 1999, 2005; Lamontagne et al. 2000; McEachern et al. 2000; Kelly et al. 2006; but see Lewis et al. 2014 for a counterexample), ammonium (Bayley et al. 1992; Allen et al. 2005), total phosphorus (Bayley et al. 1992; Garcia and Carignan 1999, 2005; Lamontagne et al. 2000; McEachern et al. 2000; Kelly et al. 2006; Wagner et al. 2014; but see Allen et al. 2005; Lewis et al. 2014 for counterexamples), and dissolved organic carbon levels (McEachern et al. 2000; Allen et al. 2005; but see Carignan et al. 2000 for a counterexample). Lakes that receive increased inputs of allochthonous carbon from humic substances after wildfires also tend to have lower alkalinity (McEachern et al. 2000; Allen et al. 2005), pH (McEachern et al. 2000), and light penetration (decreased Secchi depth and increased colour; Carignan et al. 2000; McEachern et al. 2000;

Scrimgeour et al. 2001). Trace metals and ions such as calcium (Ca), magnesium (Mg), and sulfate (SO_4) may also increase (Garcia and Carignan 1999).

Increases in nutrient input from the catchment can lead to higher productivity in some systems (Garcia and Carignan 1999, 2005; Kelly et al. 2006; Silins et al. 2014). The increases in primary producers in response to these nutrients can cause short-term increases in zooplankton biomass, particularly within the first year after wildfire (Patoine et al. 2000; Winkler et al. 2009; but see Jalal et al. 2005 for a counterexample). However, previous studies show that taxonomic richness and diversity of zooplankton communities appear to be resilient to the short-term influx of nutrients (Patoine 2002; Jalal et al. 2005). Longer lasting changes have been found for benthic macroinvertebrate groups. Changes in lake productivity, as well as the general effects of disturbance, can affect the relative abundance and richness of benthic macroinvertebrates (Silins et al. 2014; Musetta-Lambert et al. 2019; but see Lewis et al. 2014 for a counterexample). Some studies suggest that opportunistic feeders such as amphipods may increase in lakes disturbed by wildfire causing a concomitant increase in predators for at least 30 years post-fire (Scrimgeour et al. 2001; Lewis et al. 2014; Silins et al. 2014; Musetta-Lambert et al. 2019). Lewis et al. (2014) speculated that amphipod increases were related to their generalist ecology, which allowed them to take advantage of changing water quality conditions and allochthonous food sources.

Most studies that have examined the effects of wildfires on boreal lakes have been conducted within the southern boreal forest, leaving a significant gap in our knowledge for northern water bodies (Robinne et al. 2020). As summarized above, past studies have shown a mixture of short-term impacts on water quality and zooplankton, and longer-term impacts on benthic macroinvertebrate communities as vegetation recovers in burned watersheds. However, the presence of permafrost and the shorter growing seasons in the north may lead to longer-lasting impacts on water quality and zooplankton or potentially more severe impacts on benthic invertebrates.

In this study, we used the reference condition approach (RCA; Reynoldson et al. 1997, 2001; Reece and Richardson 1999) to examine how small lakes in the northern boreal forest responded to wildfires in their catchments. Eleven lakes affected by wildfires in their catchments were compared to nine reference lakes that had not experienced wildfires in their catchments for at least 60 years. Our study had two main objectives: 1) To determine if water quality in northern boreal forest lakes differed dependent on burn history in their catchments; and 2) To determine if differences in water quality related to burn history could explain differences in invertebrate communities among lakes. We hypothesized that lakes affected by burns in their catchments would show water quality changes described in past studies, including lower dissolved oxygen, pH levels, light penetration, and elevated nutrients, productivity, and metals (Garcia and Carignan 1999, 2005; Carignan et al. 2000; Lamontagne et al. 2000; McEachern et al. 2000; Scrimgeour et al. 2001; Allen et al. 2005; Kelly et al. 2006; Silins et al. 2014). We also hypothesized that the structure of benthic macroinvertebrate communities (richness, abundance, functional feeding groups) would be affected by changes in water quality such as increased productivity and food availability, and decreased pH levels, DO levels, and light penetration (Scrimgeour et al. 2001; McEachern et al. 2002; Lewis et al. 2014; Musetta-Lambert et al. 2019).

Methods

Sample Sites

Located within the Taiga plains of the Northwest Territories (NWT), Canada, the Sahtú Settlement Area (SSA) extends approximately 150 km southward from the start of the

Mackenzie River Delta to just south of Great Bear Lake (Mochnacz et al. 2007). Located within the SSA are thousands of small water bodies, including lakes, ponds, fens, and bogs (Golder Associates 2015). The Mackenzie River traverses 520 km north across the SSA, where it receives freshwater input from tributaries ranging in size from ephemeral streams to large rivers.

We consulted the Canadian National Fire Database (Canadian Forest Service 2020) to select eleven reference lakes and nine lakes affected by burns for our study. Most lakes were located northwest of the community of Norman Wells at 65°31'15.376"N 127°24'15.894"W with three reference lakes north of Norman Wells, and one reference lake within the community (lake 8; Error! Reference source not found.). The lakes were selected based on wildfire history and accessibility on foot or via floatplane. We chose nine lakes with >68 % of their catchments burned during the 2014 wildfire, two lakes with <16 % of the catchments burned in a 2015 wildfire, and nine unimpacted reference lakes. Throughout our analyses, we grouped the two lakes with <16% of their catchments burned with the reference lakes due to the small proportion of land affected in their catchments. Four of the reference lakes were sampled in August 2019, while the remainder were sampled during August 2018. An effort was made to select lakes that were comparable with in terms of their surface area, catchment size, and sample timing (all sampling was conducted in mid/end of August) **(Table 1)** between reference lakes and those that didn’t experience burns. Most of our study lakes were in separate catchments, except for two lakes exposed to burns that shared a single catchment (lakes 13 and 14), and another two lakes exposed to burns that shared a catchment (16 and 17; **Error! Reference source not found.**)

Hydrological and geological surveys within the study region show lakes are distributed in three different surficial geological zones of the organic-rich: till blanket, till veneer, and fine-grained (glacio)lacustrine, and surficial geological zones with exposed bedrock of colluvial rubble. Permafrost occurred within 48-78% of the land surface within this region (Discontinous permafrost; Golder Associates 2015). All lake sites were surrounded by high organic humic peat mosses (Golder Associates 2015). Vegetation within the Taiga plains ecoregion consists of a mixture of trembling aspen (*Populus tremuloides*), Alaska paper birch (*Betula neoalaskana*), white spruce (*Picea glauca*) and black spruce (*Picea mariana*) with large quantities of shrubs such as Labrador tea (*Rhododendron groenlandicum*) and dominated by lichen (*Cladina spp*.) (Gibson et al. 2018).

Water Quality

We obtained water quality measurements at three sampling stations, each randomly selected using GIS software (ArcGIS), located ~10 m from the shoreline and at one station at the center of each lake. We took the mean of each variable from these four sites for use in our analyses. To measure dissolved oxygen (DO) and conductivity (~0.5 m depth), we used a YSI-Professional Plus meter equipped with a polarographic DO sensor probe and a conductivity sensor probe. We measured temperature and pH (~0.5 m depth) using an Oakton 150 series pH probe. To measure chlorophyll-*a* levels, we collected surface water (250 mL) at each station, filtered it through a Millipore glass filter (0.45μm), and preserved the algae by freezing (-20℃) for <2 months. In the laboratory, we extracted chlorophyll-*a* from the frozen filter by soaking it in 100% methanol for 24 hr and then measured the amount of chlorophyll-*a* in the methanol using a Turner Designs TD700 fluorometer.

To measure organic matter in lake sediments and shoreline soils, we collected surface sediment samples (<5 cm deep) and mineral soil samples (~3 m from the shoreline; <5 cm deep) at each of the three randomly selected sampling stations on each lake. We estimated percent organic matter in sediment (0.5 g) from each site using loss of ignition (LOI) methods described by Heiri et al. (2001). To measure total organic carbon (TOC), dissolved organic carbon (DOC), total phosphorus (TP), total nitrogen (TN), and trace metal concentrations, we collected a sample (~2 m depth) from the center of each lake using an integrated water sampler constructed from a flexible polypropylene tube (19 mm inner diameter). If the lake was <2 m deep, we lowered the tube carefully to not disturb sediment and made multiple collections to obtain the required volume of water. For trace elements and metals, we filtered water (100 mL) with a Whatman 934-AH glass microfiber filter (0.45 μm) and poured it into a Nalgene bottle (125 mL; 24 hr HCl acid-washed) and preserved it by adding trace metal grade HNO_3 until a pH of 1-2 was reached. We repeated these steps to measure DOC except for using trace metal grade H_2SO_4 instead of HNO_3 as a preservative. For TOC, TN, and TP, we poured water (100 mL; unfiltered) into a third Nalgene bottle (125 mL: 24hr HCl acid-washed) and preserved it by adding trace metal grade H_2SO_4 to a pH of 1-2. Finally, we collected water (1 L; unfiltered) that we used to analyze alkalinity, ammonium (NH_4), colour, nitrite (NO_2), phenols, phosphate (PO_4), sulfate (SO_4), and turbidity immediately upon return to the field base (YSI-9500 photometer). Maximum depth (Z_{max}) was estimated using a hand-held depth finder.

In the laboratory, we analysed DOC, TOC, and TN using a carbon and nitrogen analyser (Shimadzu TOC-LCPH Carbon and Nitrogen Analyzer). Total phosphorus (TP) was analysed using an auto-analyser after digestion (SEAL AA3 Auto-Analyser; US EPA method 600; US EPA method 600 1993). An analysis of a suite of trace metals was conducted using an inductively coupled plasma-optical emission spectroscopy analyser (ICP; Perkin Elmer Optima 8000 Inductively Couple Plasma-Optical Emission Spectroscopy).

Biological

At each of the three randomly selected sample stations on each lake where we measured water quality, we collected samples of littoral zooplankton with an oblique tow (~5 m; 64 µm mesh net) at a standardized depth (~1 m). We identified zooplankton samples to genus using a combination of primary literature (Witty 2004) and an online image-based key (Haney 2013). A minimum of one hundred individual zooplankters was identified from each station (≥300 total per lake) using a Hensen-Stemple pipette and a magnetic stirrer to obtain subsamples (5% subsample size). In lakes with low concentrations of zooplankton, we identified all individuals in the sample. We estimated relative abundance based on the percentage of the sample analyzed per station and then calculated a mean for each lake for use in analyses. To adjust for differences in the number of individuals identified from each lake, we rarified genus richness to 96 individuals using the rarefy function in the "*vegan*" package for R (Oksanen 2017).

We also collected benthic macroinvertebrates (500 µm mesh D-net) with a Travelling-Kick-and-Sweep methodology to disturb the sediment at a uniform depth (OBBN and CCME 9.3 protocols; ~5 cm deep) for near-shore sites in lakes (Jones et al. 2007; CCME 2011). We shortened the length of time each sample was collected to five minutes instead of ten because of time constraints at each lake (hourly cost of floatplane). Samples were kept cold and preserved frozen (-80°C) immediately upon return to the laboratory and identified within a year.

To determine if the total abundance of benthic macroinvertebrate groups differed in lakes affected by wildfires, we identified benthic invertebrates to family using a combination of primary literature (Proctor 2006; Merritt et al. 2008; Thorp and Rogers 2016) and online image-based keys (Parker 2012). Members of Hydrozetidae were excluded from analysis due to their small size and oligochaetes were excluded due to their poor preservation upon freezing. To enumerate the animals, we created sub-samples from each benthic macroinvertebrate sample using a random sampler (Marchant Box), as it gives reliable estimates of abundance, is easily randomized, and is less costly and time-consuming than identifying every organism in the entire sample (OBBN and CABIN protocols; Jones et al. 2007; McDermott et al. 2014). We identified >100 individual benthic macroinvertebrates per station (≥5% needed to be analysed) to assess richness and abundance (≥300 total per lake; OBBN and CCME 9.3 protocols; Jones et al. 2007; CCME 2011). We estimated total abundance based on the percentage of the sample analyzed per station and then calculated a mean for each lake for use in analyses. We rarified family richness to 303 individuals, which was the lowest number identified from a single lake, using the rarefy function in the "*vegan*" package for R (Oksanen 2017). We also examined differences in the abundance of Ephemeroptera, Plecoptera, and Trichoptera (EPT), as these groups have been commonly used as indicator species following wildfire disturbance (Silins et al. 2014; Musetta-Lambert et al. 2019).

GIS-Based Data and Remote Sensing

We observed the response of wildfire on the short-waved infrared (SWIR) spectral colour reflectance bands (SWIR1 and SWIR2) of the surface area of each lake using a combination of Landsat 8 and Landsat 7 satellite imagery (30 m resolution; **see Appendix S1**). SWIR colour reflectance is highly correlated with macrophyte biomass (Khanna et al. 2011; Oyama et al. 2015; Yadav et al. 2017). We calculated the water catchment area for each lake using digital elevation model (DEM) files (2 m resolution; Porter et al. 2018) and GIS software (ArcGIS). We manually cut waterbodies from the catchment polygon shapefiles so only terrestrial areas would be included. We also used ArcGIS to calculate the area of wetland within each water catchment, the perimeter of each lake, the surface area of each lake, the slope % of the catchment, total drainage ratio (DR), and the number of hectares in each catchment that were burned. We used the ratio of the surface area of a lake to the total area of the lake catchment burned as a measure of the potential impact of wildfire on each lake. Hereafter we refer to this ratio as the burned drainage ratio (bDR). Our rationale for using this method to calculate the potential impact of wildfire on each lake was based on evidence that the catchment size to lake surface area ratio (drainage ratio) has a direct relationship to the input of allochthonous material and carbon residence time in lakes after wildfire (Garcia and Carignan 1999; Lamontagne et al. 2000; McEachern et al. 2000; Jalal et al. 2005; Jones et al. 2018).

Statistical Analysis

To identify variables that might have been affected by wildfire, we used Spearman's rank correlation coefficient (r_s) to determine which variables were correlated to burned drainage ratio (bDR). Spearman's was chosen rather than typical Pearson correlations because bDR was not normally distributed and could not be transformed to a normal distribution. We also examined correlations among the variables significantly correlated with bDR to determine if these relationships might have been driven by confounding relationships. Correlations were tested using the cor.test function in R (R Core Team 2019).

To determine the relative importance of individual environmental variables for predicting abundance and richness of benthic macroinvertebrates and zooplankton, as well as EPT abundance, we conducted hierarchical partitioning (Chevan and Sutherland 1991; Mac Nally 2000). The R^2 values produced through multiple regression analyses are often not an accurate measure of the importance of individual variables due to collinearity between the predictor variables (Mac Nally 2000, 2002). Hierarchical partitioning minimizes this problem by performing an exhaustive model search and then averaging the influence of each predictor variable in each model it appears, thereby determining the contribution of each variable independent of others (Mac Nally 2000). Before conducting the hierarchical partitioning, some variables were transformed to improve linearity and normality **(see Appendix Table S1)**. Hierarchical partitioning was carried out using the hier.part package in R (Walsh and Mac Nally 2013). The hier.part function accepts a maximum of 12 variables, so for the benthic macroinvertebrate analysis we selected variables that are commonly found to be associated with the structure of benthic macroinvertebrate communities: chlorophyll-*a*, colour, conductivity, dissolved oxygen, dissolved organic carbon, macrophytes (SWIR2), % organic sediment (LOI), pH, temperature, total phosphorus, and maximum depth (Z_{max}). For the zooplankton analysis, we also included variables known to affect community structure, including total phosphorus, calcium, maximum depth, turbidity, lake surface area, macrophyte biomass (SWIR1), chlorophyll-*a*, temperature, pH, conductivity, and total organic carbon. For both the benthic macroinvertebrate and zooplankton analyses we also included bDR, the ratio of hectares burned in the watershed to lake surface area.

To examine differences in water quality among lakes, we used principal component analysis (PCA). A PCA was selected as the appropriate ordination method after conducting a detrended correspondence analysis (DCA) that indicated gradient lengths were less than three, suggesting linear relationships (Borcard et al. 2011). Before conducting the PCA, variables were standardized so that each had a mean of zero and unit variance, and some variables were transformed to improve linearity and normality **(see Appendix Table S1)**. Lakes within the ordination diagram were coloured according to log (bDR +1) to show the expected impact of wildfires on each site. The PCA was conducted using the prcomp function in R and was plotted using the fviz_pca_biplot function in the factoextra package (Kassambara and Mundt 2019).

To determine which environmental variables were related to differences in the relative abundance of benthic macroinvertebrate taxa among lakes, we conducted redundancy analysis (RDA) for benthic macroinvertebrate taxa and zooplankton genera. Before conducting the RDAs, we removed rare taxa (found in <20 % of lakes) and Hellinger-transformed the data (Legendre and Gallagher 2001). Stepwise variable selection for the RDAs was conducted using the ordistep function in the vegan package for R (Oksanen et al. 2015). To avoid data dredging, we did not include a full list of potential variables for selection in the RDAs. Instead, we chose 19-18 variables that have literature support to suggest that there are established relationships with benthic macroinvertebrate or zooplankton community structure **(see Appendix Table S2)**. Before conducting the RDA, some variables were transformed to improve linearity and normality **(see Appendix Table S1)**. The final model was tested for collinearity using the variance inflation factor and all VIF scores were < 5. The overall significance of each RDA was tested by running a permutation test according to Legendre et al. (2011). As with the PCA figures, lakes on the ordination diagram from the RDAs were coloured according to log (bDR +1).

Results

Physical and water quality variables

Our study lakes varied in their physical and water quality characteristics **(see Appendix Table S3-S8)**, but for most physical characteristics (including maximum depth, catchment slope, surface area, and water catchment area), no statistical difference in *P* values (Mann-U Test) was observed among lakes within burned catchments and reference lakes **(Table *1*)**. Elevation between reference and burned lakes had significantly different *P* values (Mann-U Test) **(Table 1)**.

Our Spearman correlations identified several interesting relationships between the physical characteristics of the lakes, water quality, and bDR (hectares burned/surface area of the lake) **(**Error! Reference source not found.**)**. In terms of those related to burns in lake catchments, colour (r_s= 0.34), dissolved oxygen (r_s= -0.38), macrophytes-SWIR1 (r_s= 0.26), macrophytes-SWIR2 (r_s= 0.54), pH (r_s= -0.45), and phenols (r_s= 0.33) showed significant correlations with bDR **(**Error! Reference source not found.**)**. Only one variable, catchment area, was significantly correlated to total hectares burned within a catchment **(**Error! Reference source not found.**)**.

Remote sensing time-series of short-wave infrared (SWIR) reflectance from wavelengths between 2.08 – 2.35 µm (SWIR1) **(**Error! Reference source not found.**A)** and 1.55 – 1.75 µm (SWIR2) **(**Error! Reference source not found.**B)** showed a difference between reference lakes and burned lakes. Reflectance values from Landsat 7 and 8 over a period from 1999 to 2019 diverged significantly between reference lakes and burned lakes. Scaled SWIR1 reflectance values for reference lakes remained a similar mean value of 300.78 in 2013 (pre-burn) and 292.10 when sampled (2018 and 2019; **Table 1**). SWIR1 reflectance values for burned lakes increased to a mean of 329.77 in 2013 (pre-burn) and 353.91 in 2018 when sampled **(Table 1)**. Mean SWIR2 reflectance values for reference lakes in 2013 (pre-burn) were at 162.37 and did not significantly differentiate to 163.81 when sampled (2018 and 2019; **Table 1**). SWIR2 mean reflectance values for burned lakes increased from 174.63 in 2013 (pre-burn) to 208.59 in 2018 when sampled **(Table 1)**.

Our water quality PCA did not show a clear pattern separating burned lakes from reference lakes **(**Error! Reference source not found.**)**. Lakes were instead separated into two categories by geological conditions: clear, bigger versus smaller, coloured. The first axis of the PCA explained 32.4 % of the variation among lakes and separated larger lakes with high chlorophyll-*a*, pH, conductivity, and calcium from those that were smaller with more colour and higher surface temperatures **(**Error! Reference source not found.**)**. The second axis explained 15.7 % of the variation and separated more turbid lakes at a higher elevation from those at lower elevations with higher total organic carbon, total nitrogen, and aquatic macrophytes-SWIR2.

Zooplankton

A total of 21 different zooplankton genera were identified in our study lakes **(see Appendix Table S9-S10)**. Our RDA for zooplankton genera did not show significant differences between lakes affected by wildfire and reference lakes **(**Error! Reference source not found.**)**. Along the first axis of our RDA, variation in zooplankton communities among lakes was best explained by macrophyte biomass and alkalinity. This axis explained 32.59 % of the variation among lakes. Lakes with higher axis 1 score tended to have more of the cladocerans *Bosmina*, *Alona*, *Alonella*, *Sida*, and *Chydorus* while lakes with lower scores were dominated by the calanoid copepods *Leptodiaptomus* and *Skistodiaptomus*. The second axis explained only 4.40 % of the variation among lakes. Warmer lakes had more immature calanoid copepods, while higher abundances of the cyclopoid copepod *Microcyclops* and the cladoceran *Daphnia* appeared in some lakes affected by burns with higher SWIR2 macrophyte biomass reflectance **(**Error! Reference source not found.**)**.

Results of our hierarchical partitioning analysis showed that the most important variables driving differences in zooplankton richness among our study lakes were macrophyte biomass, and calcium **(**Error! Reference source not found.**A)**. bDR and DOC were equally important. For abundance, differences among lakes were best explained by total phosphorus, temperature, turbidity, and chlorophyll-*a* **(**Error! Reference source not found.**B)**.

Benthic Macroinvertebrates

In total, 62 families of benthic invertebrates were found in our 20 study lakes **(see Appendix Table S11-S18)**. Our RDA of benthic invertebrate families showed that lakes with a higher burned drainage ratio (bDR) had a significantly lower axis 1 score, corresponding with higher numbers of Gammaridae and Hyalellidae **(**Error! Reference source not found.**)**. Along the first axis, differences in communities were significantly related to maximum depth and macrophyte biomass (SWIR2). Shallower lakes with lower calcium concentrations had significantly more Chironomidae, while deep lakes with more calcium housed more Gammaridae and Hyalellidae **(**Error! Reference source not found.**)**. Axis 1 explained 20.99 % of the variation in benthic macroinvertebrate families among lakes. Along axis 2, differences in communities seemed to be driven by temperature. Lakes with higher temperatures tended to have more Caenidae, Leptophlebiidae, Baetidae, and Coenagrionidae, while those with low temperature had more Planorbidae, and Lymnaeidae **(**Error! Reference source not found.**)**. The second RDA axis explained 8.67 % of the variation among lakes.

Results of our hierarchical partitioning analysis showed that the most important variables driving differences in benthic macroinvertebrate richness among our study lakes were the burned drainage ratio (bDR), followed by conductivity and dissolved organic carbon **(**

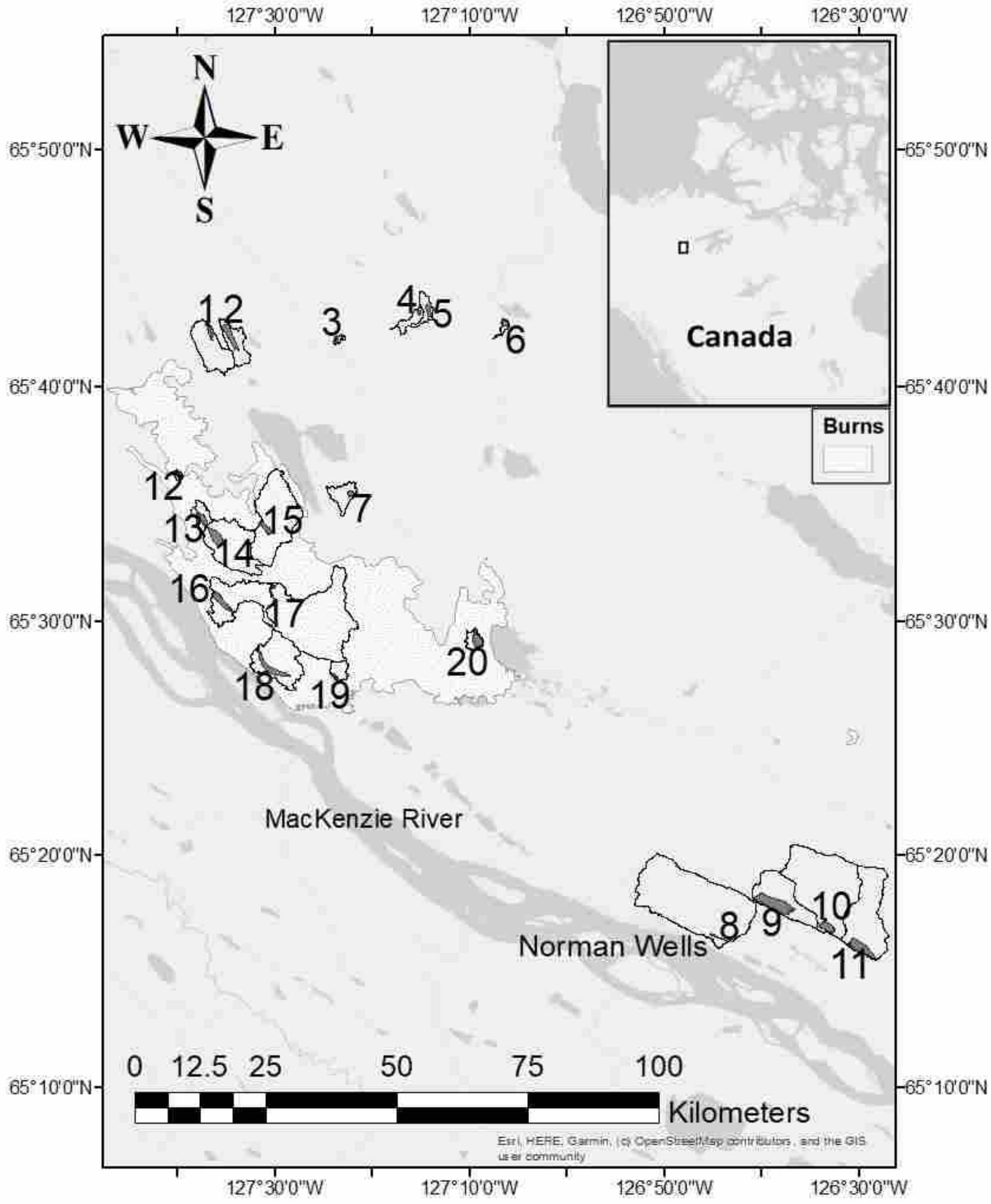


Figure 1: Locations of lakes and respective water catchments (WC) sampled, and the fire-scarred areas from the 2014 and 2015 wildfire years. Numbers represent locations of individual lakes sampled.

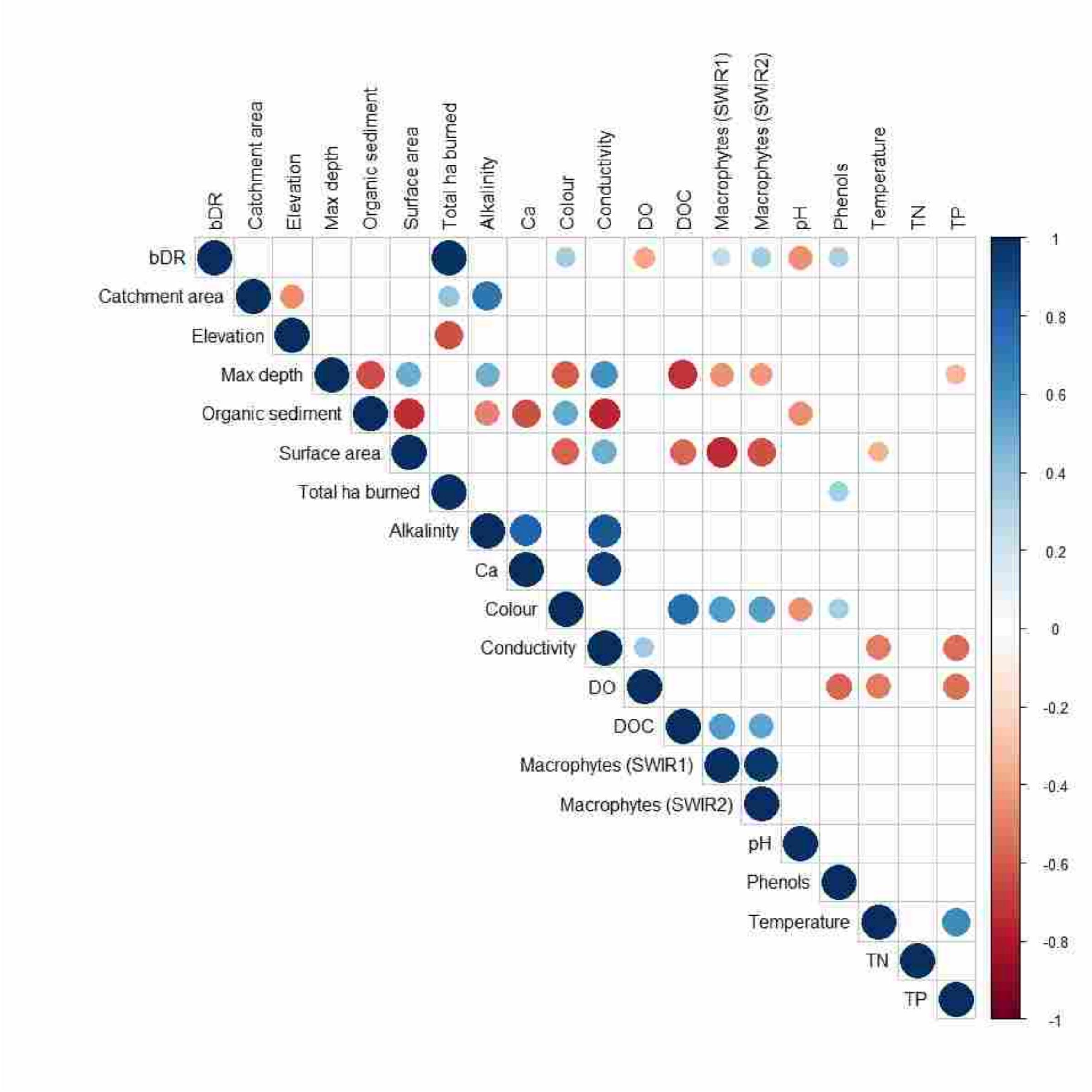


Figure 2: Spearman correlations between the burned drainage ratio (hectares burned/lake surface area), lake physical properties, and water quality variables. Blue is indicative of positive correlations and negative correlations are red. Colour intensity and size of the circle are

proportional to the correlation coefficients. Correlations that were not significant ($p > 0.05$) are indicated by empty cells. Burned drainage ratio = bDR, calcium = Ca, dissolved oxygen = DO, dissolved organic carbon = DOC, total nitrogen = TN, total organic carbon = TOC, and total phosphorous = TP.

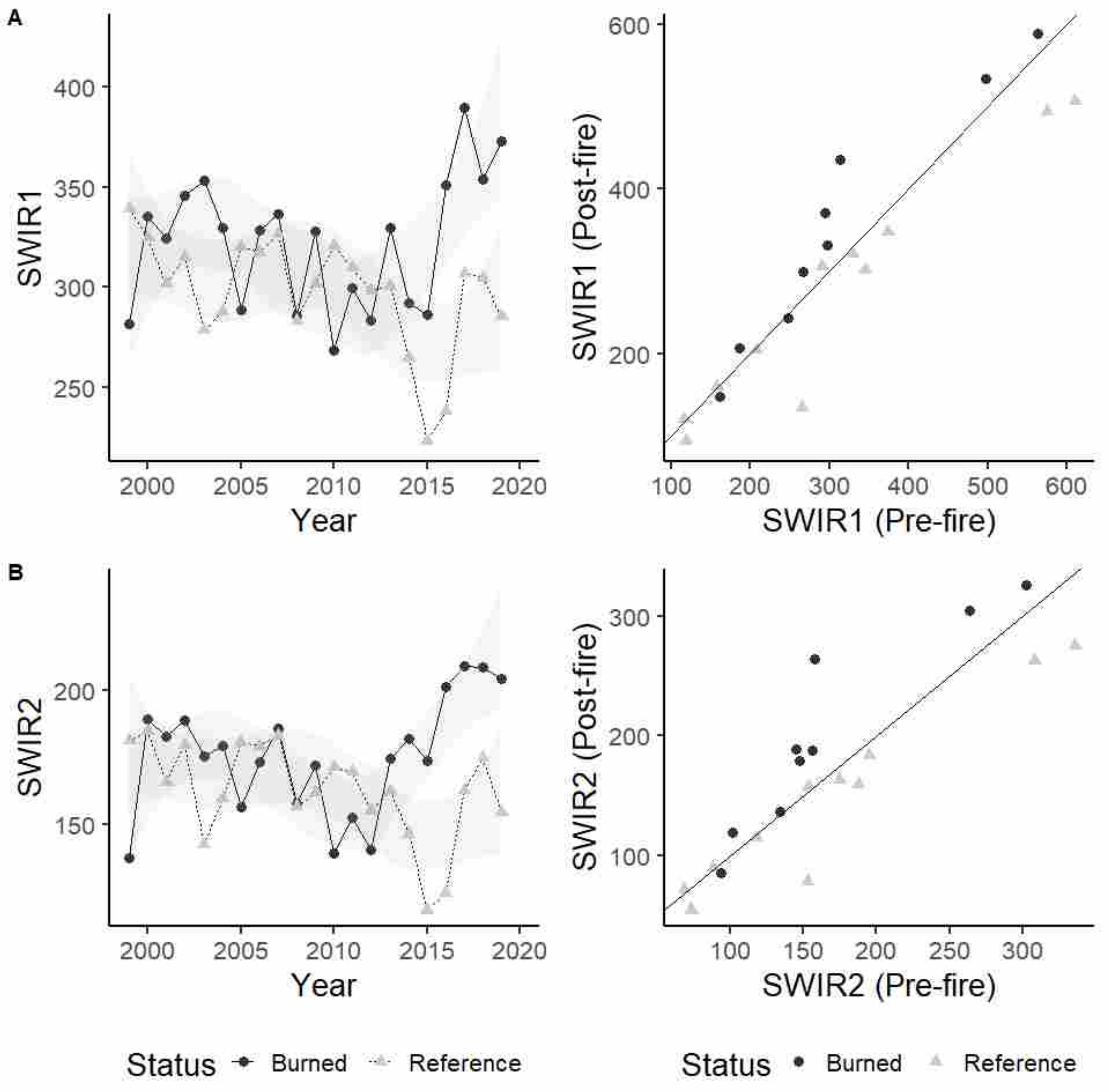


Figure 3: Time-series data of seasonal (June-August) composite medoid atmospheric corrected Landsat 7 and 8 values of mean short-wave infrared imagery SWIR1 and SWIR2 values (left panels). Reference lakes had <16 % of their catchment burned and burned lakes >68 % of their catchment burned. Relationship between mean SWIR values for each lake (9 burned lakes and 11 reference lakes) pre-wildfire (< 2014) and post-wildfire (> 2014) (right panels). The lines in the right panels are 1:1 lines.

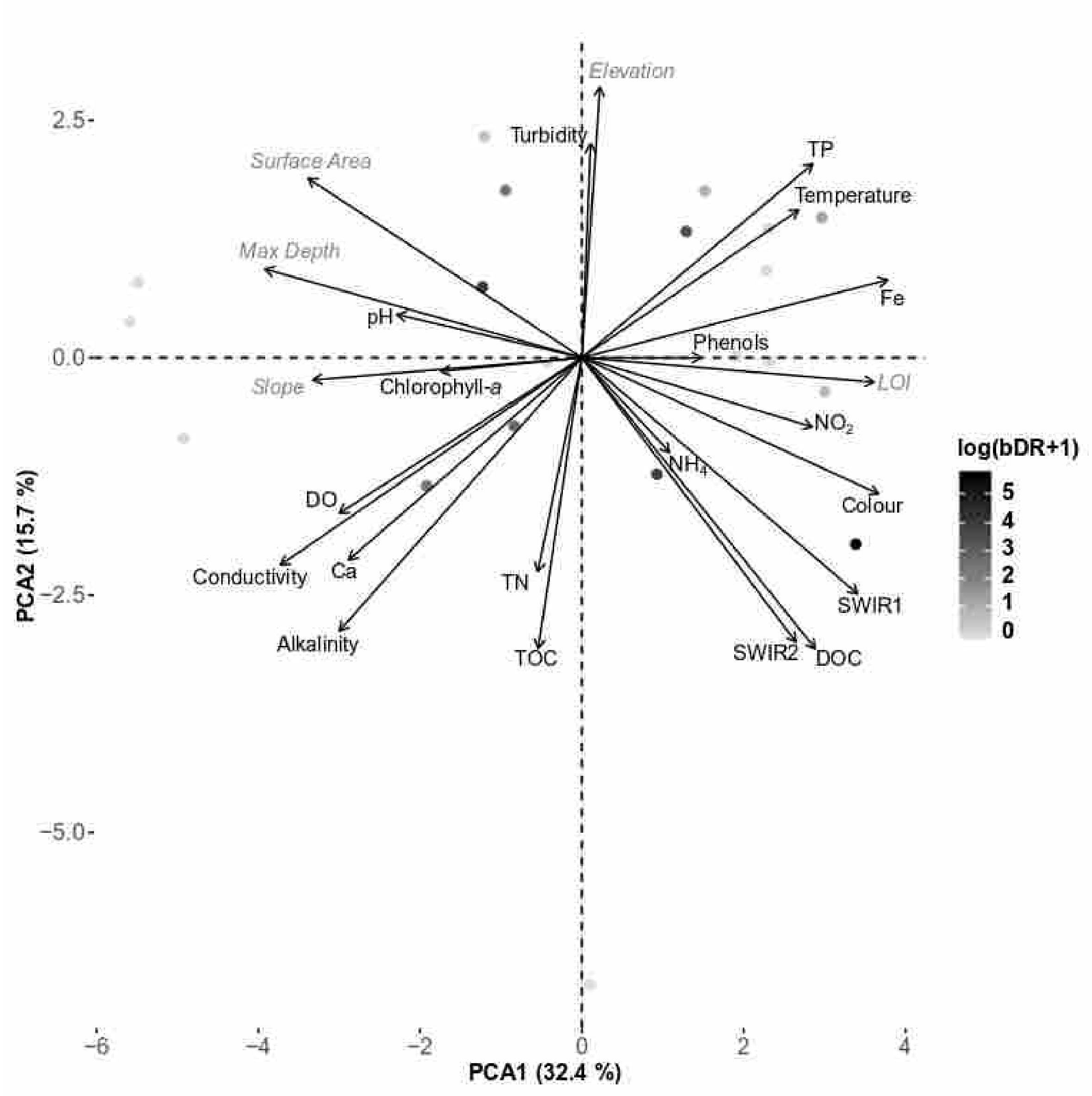


Figure 4: Principal component analysis (PCA) showing water quality variables (black text) and physicochemical variables (gray and italicized text) of lakes sampled in August 2018 and 2019 (points) within the Sahtú Settlement Area (SSA). Darker points represent lakes with a higher log burned drainage ratio +1 (bDR). Calcium = Ca, dissolved oxygen = DO, nitrogen: phosphorous ratio = N/P, total nitrogen = TN, total organic carbon = TOC, macrophyte biomass = SWIR1 and SWIR2, dissolved organic carbon = DOC, ammonium = NH_4, nitrite = NO_2, % organic sediment = LOI, iron = Fe, and total phosphorous = TP.

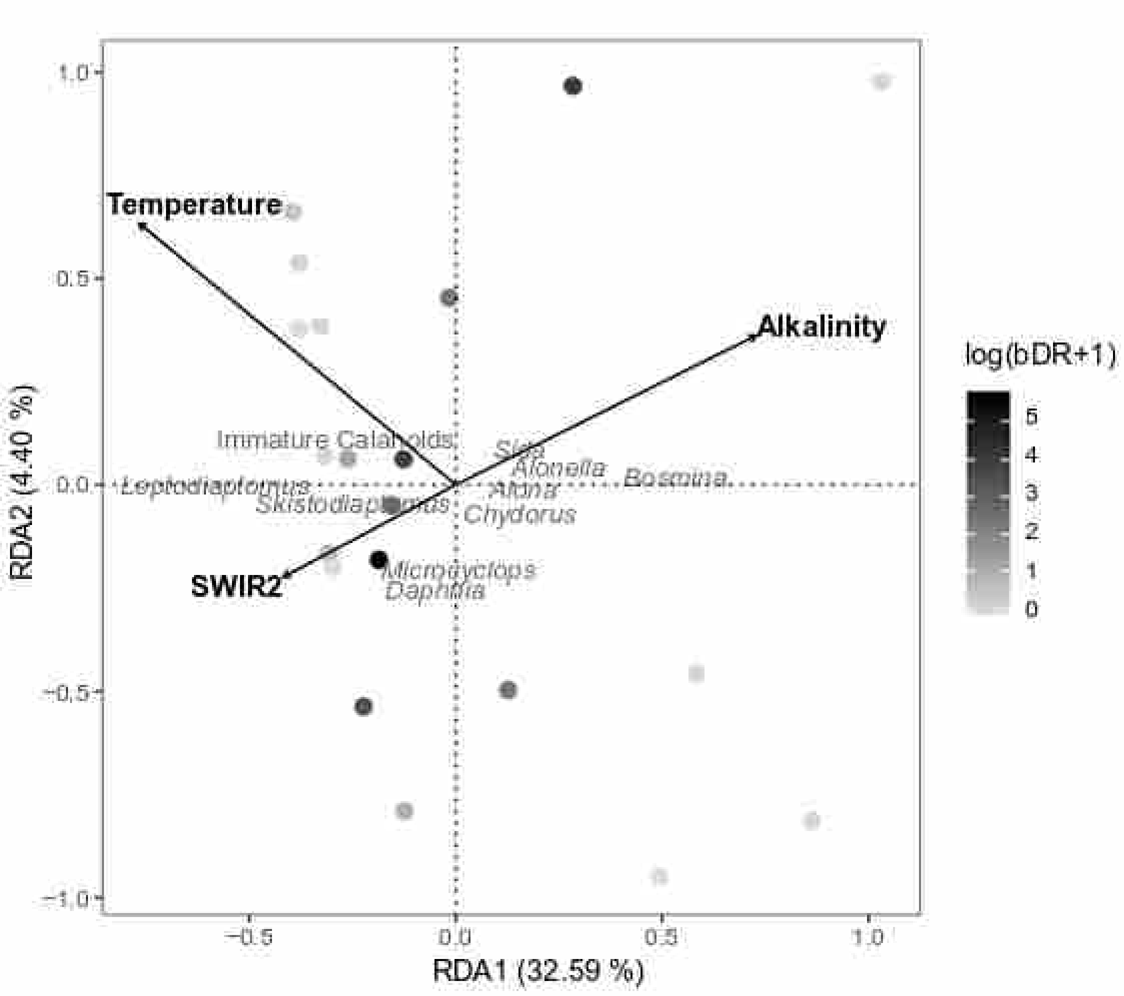


Figure 5: Redundancy analysis (RDA) of zooplankton relative abundances within the Sahtú Settlement Area (SSA). Zooplankton genera labels clustered at the center of the plot were removed for clarity and others were moved slightly to avoid overlap. Darker points represent lakes with a higher log bDR (number of hectares burned / lake surface area). Remotely sensed macrophyte biomass = SWIR2.

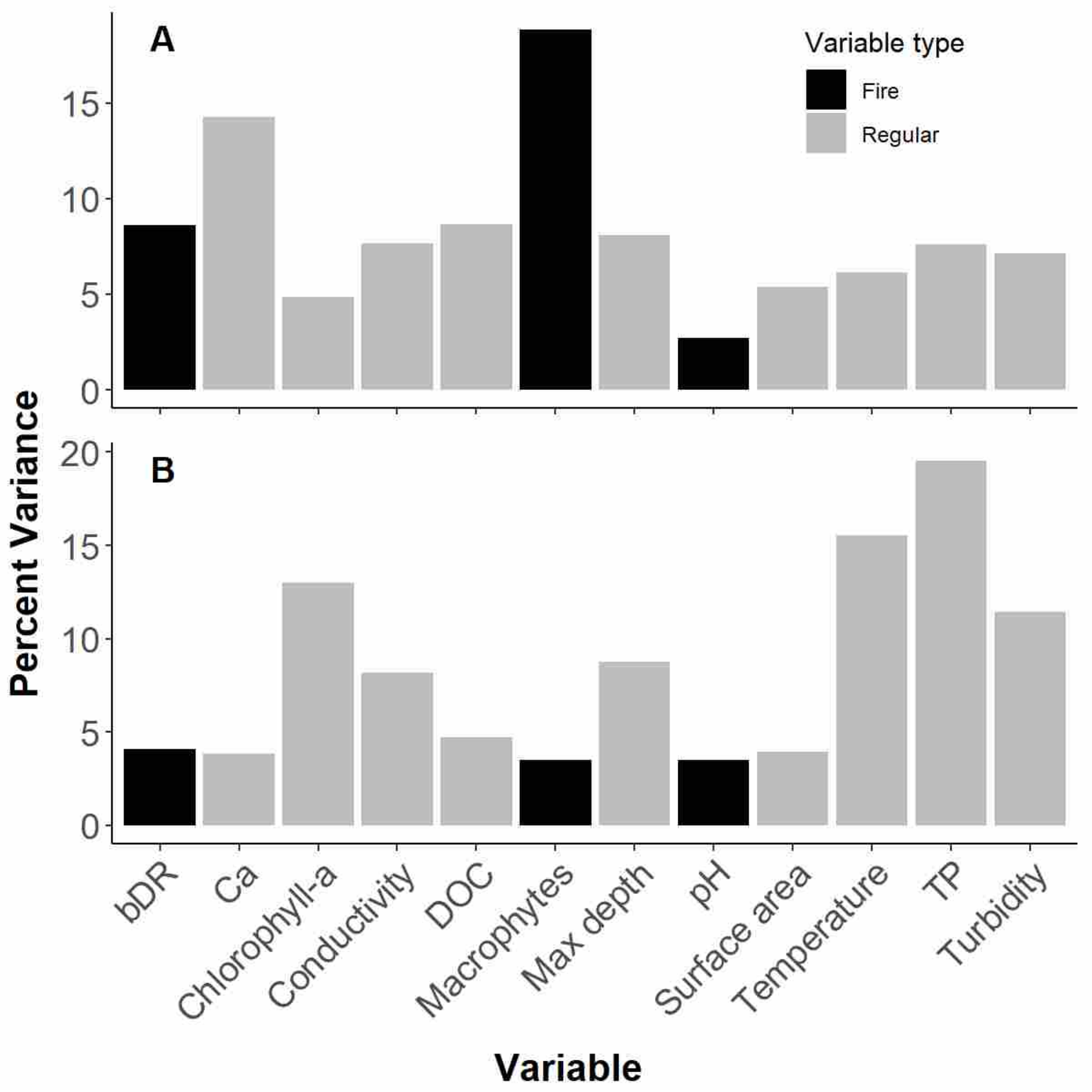


Figure 6: Results of hierarchical partitioning to determine variables most strongly associated with differences in zooplankton richness (A) and abundance (B) among lakes. Variables were identified as “fire” variables if they showed a correlation with bDR (total hectares burned /lake surface area) in Spearman correlations. Calcium = Ca, dissolved organic carbon = DOC, SWIR2 = Macrophytes, and total phosphorous = TP.

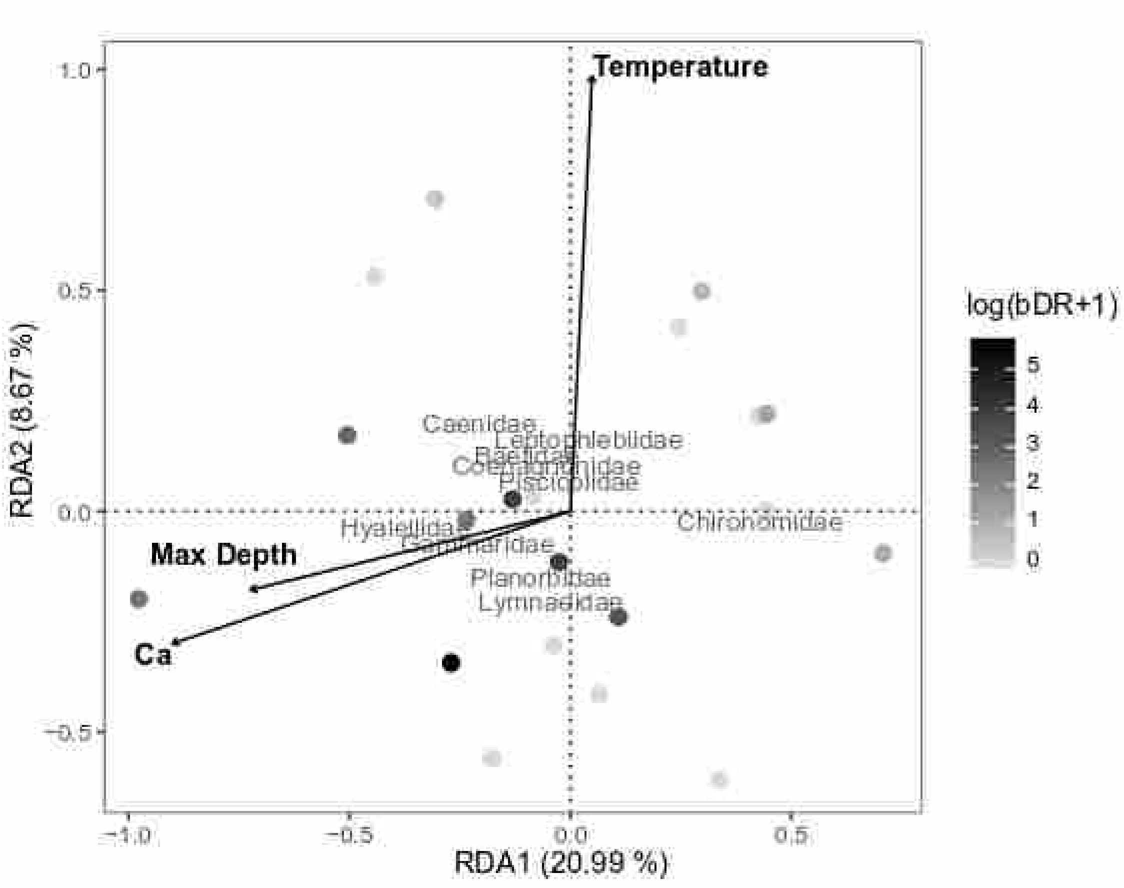


Figure 7: Redundancy analysis (RDA) of benthic invertebrate families found within lakes sampled in the Sahtú Settlement Area (SSA). Benthic macroinvertebrate family labels clustered at the center of the plot were removed for clarity and others were moved slightly to avoid overlap. Darker points represent lakes with a higher log burned drainage ratio +1 (bDR). Calcium = Ca.

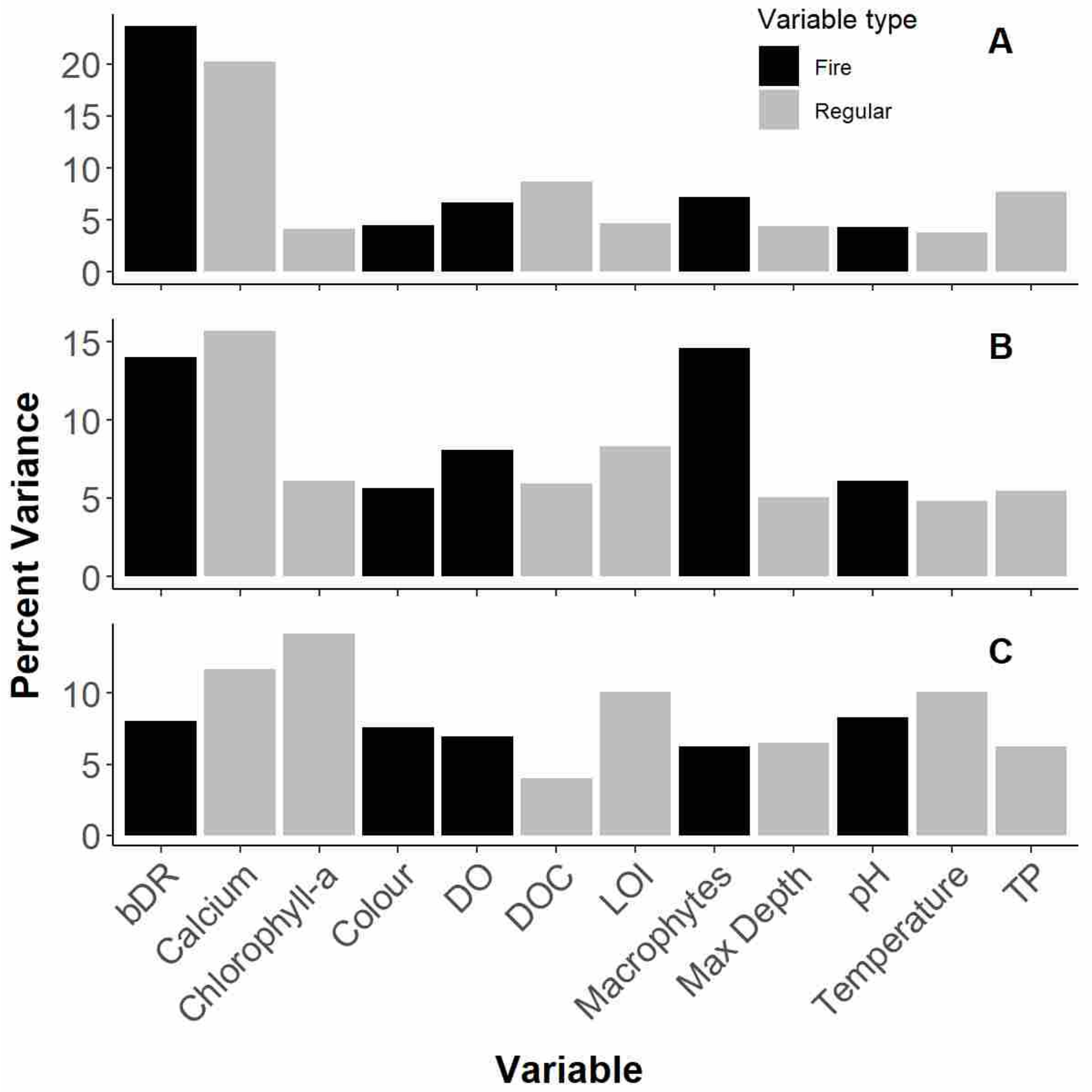


Figure 8: Results of hierarchical partitioning to determine variables most strongly associated with differences in benthic macroinvertebrate taxa richness (A), total abundance (B), and EPT abundance (C) among lakes. Variables were identified as “fire” variables if they showed a correlation with bDR (total hectares burned /lake surface area) in Spearman correlations. Dissolved oxygen = DO, dissolved organic carbon = DOC, % organic sediment = LOI, SWIR2 = Macrophytes, and total phosphorous = TP.

A). The abundance of benthic invertebrates was most related to calcium, macrophyte biomass (SWIR2), and bDR **(**

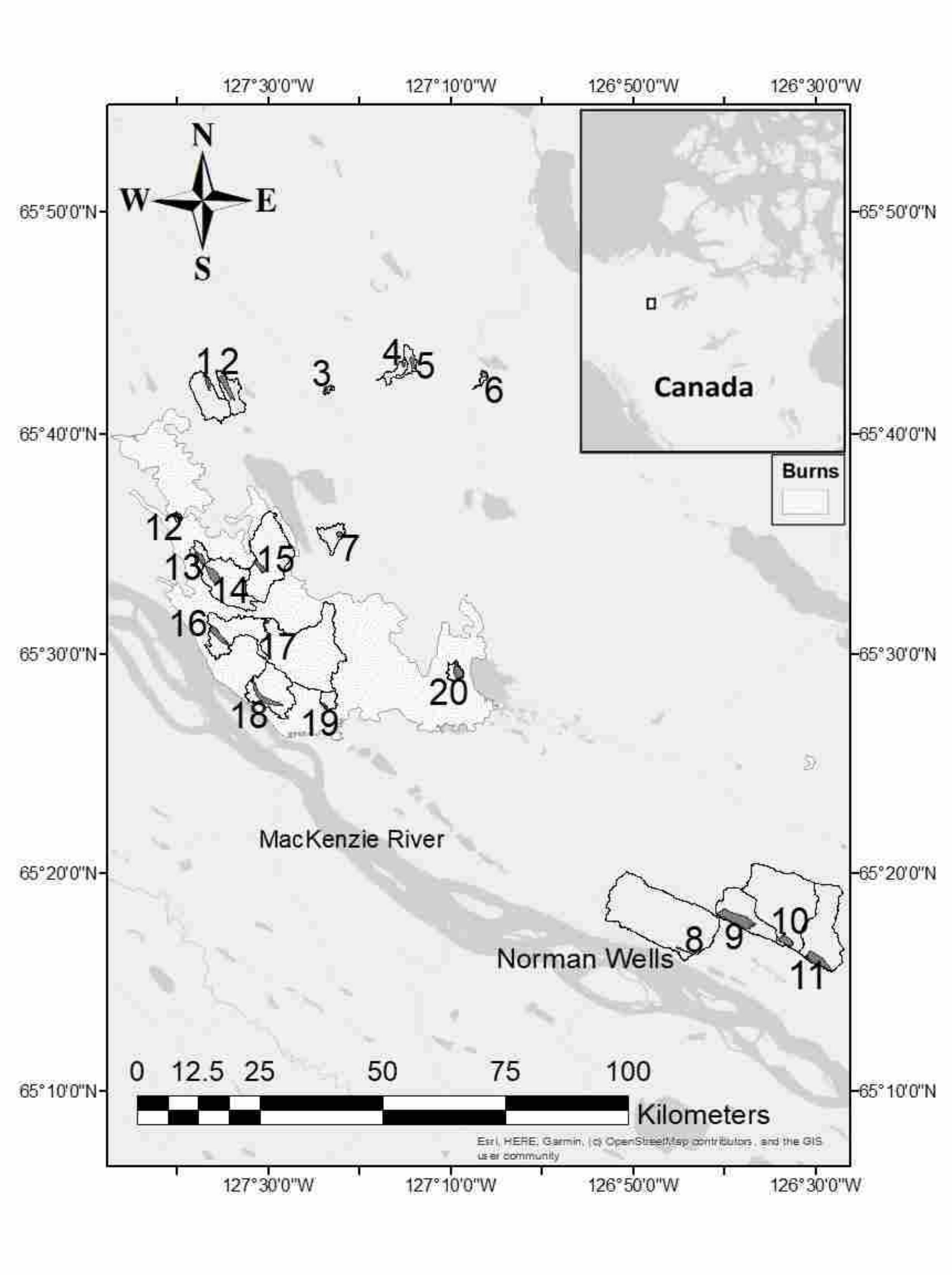


Figure 1: Locations of lakes and respective water catchments (WC) sampled, and the fire-scarred areas from the 2014 and 2015 wildfire years. Numbers represent locations of individual lakes sampled.

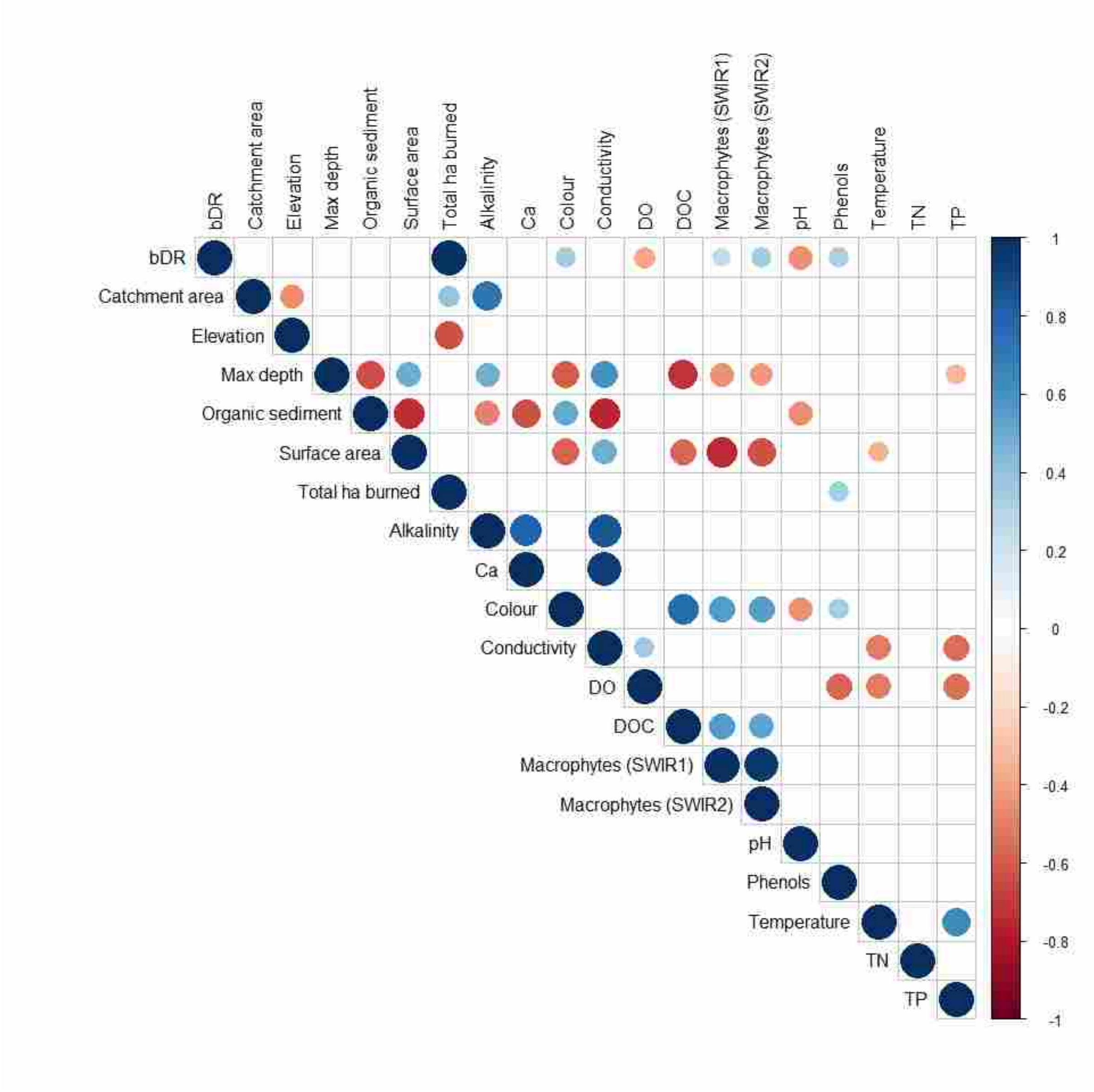


Figure 2: Spearman correlations between the burned drainage ratio (hectares burned/lake surface area), lake physical properties, and water quality variables. Blue is indicative of positive correlations and negative correlations are red. Colour intensity and size of the circle are

proportional to the correlation coefficients. Correlations that were not significant ($p > 0.05$) are indicated by empty cells. Burned drainage ratio = bDR, calcium = Ca, dissolved oxygen = DO, dissolved organic carbon = DOC, total nitrogen = TN, total organic carbon = TOC, and total phosphorous = TP.

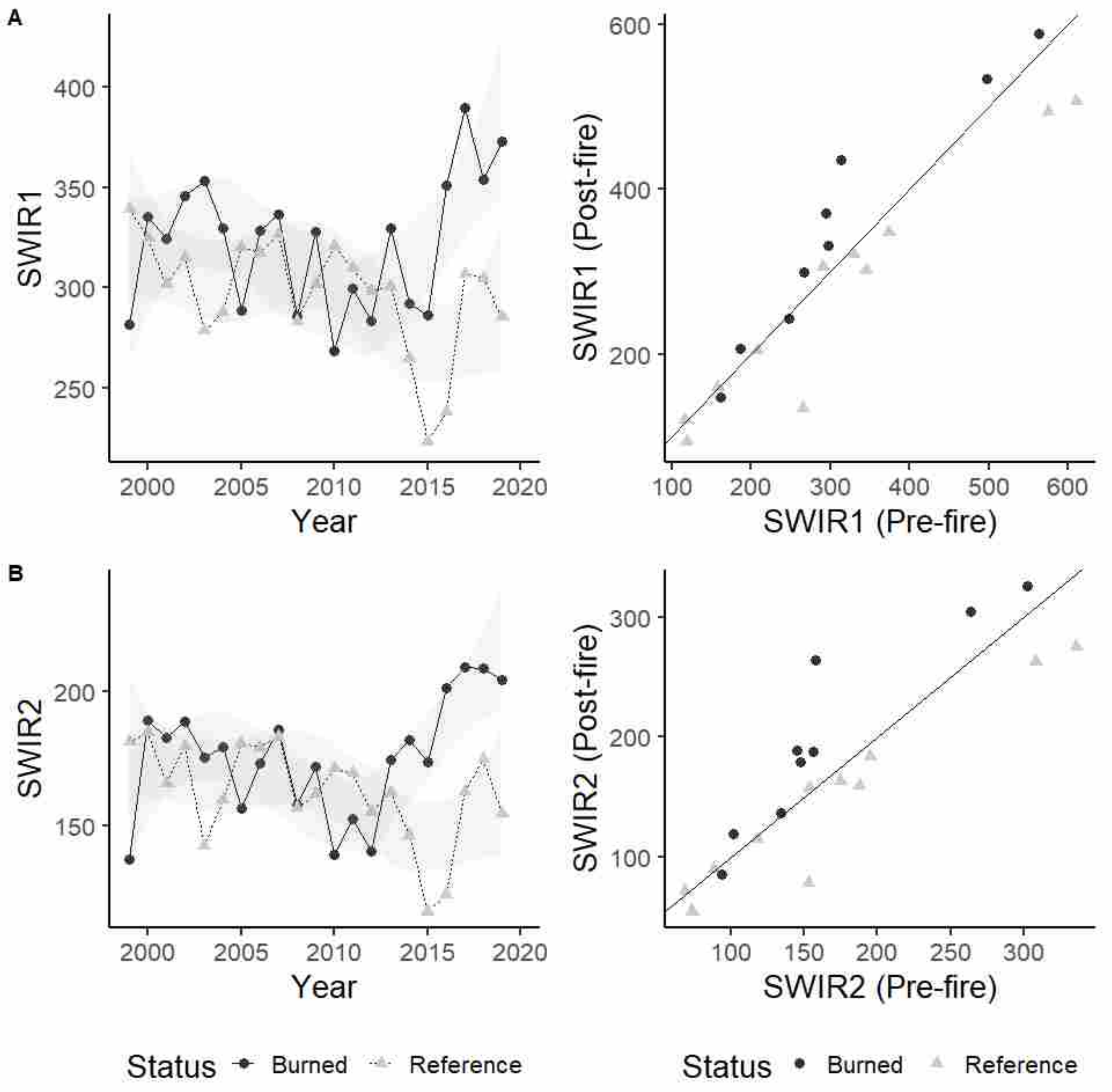


Figure 3: Time-series data of seasonal (June-August) composite medoid atmospheric corrected Landsat 7 and 8 values of mean short-wave infrared imagery SWIR1 and SWIR2 values (left panels). Reference lakes had <16 % of their catchment burned and burned lakes >68 % of their catchment burned. Relationship between mean SWIR values for each lake (9 burned lakes and 11 reference lakes) pre-wildfire (< 2014) and post-wildfire (> 2014) (right panels). The lines in the right panels are 1:1 lines.

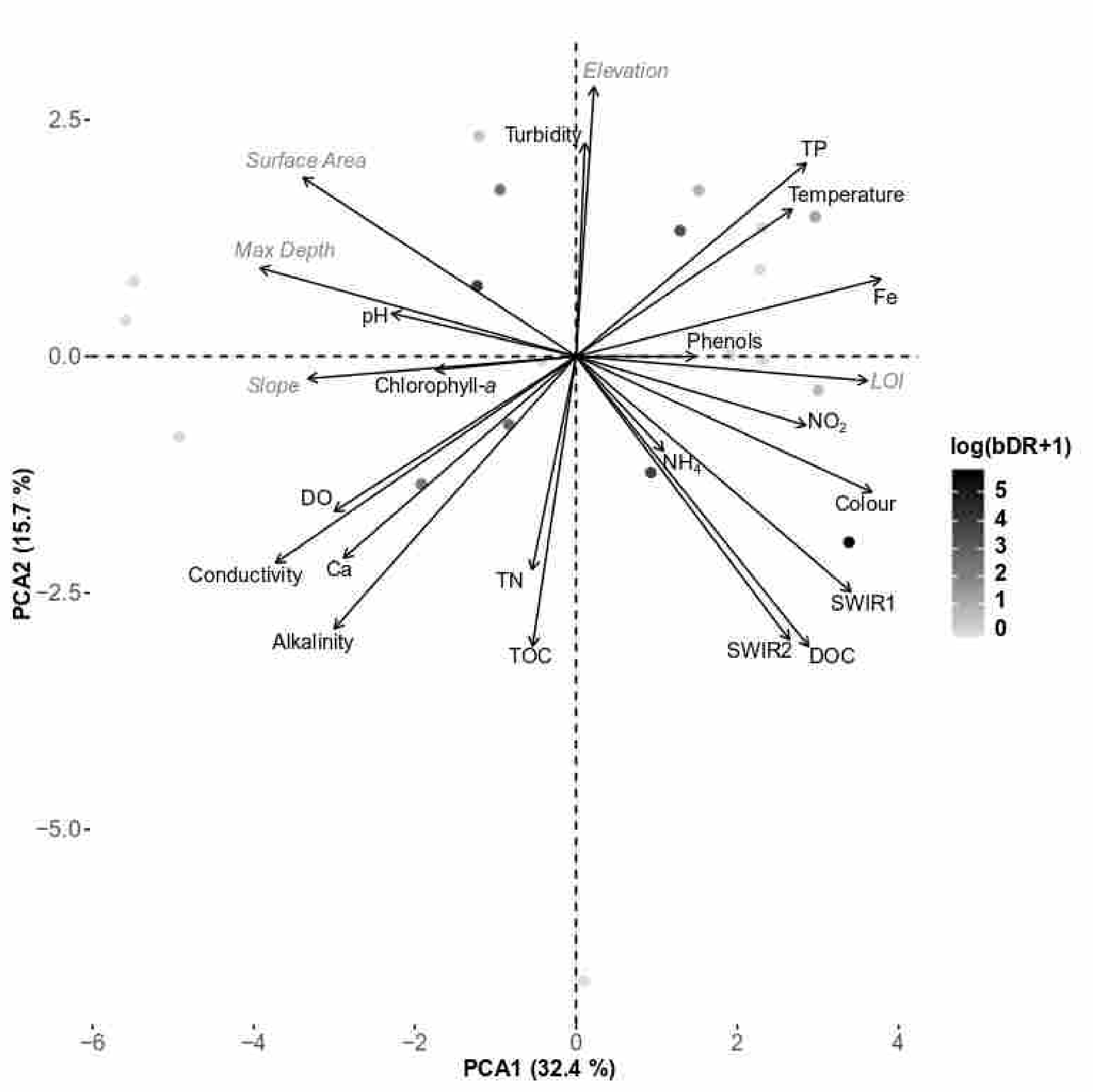
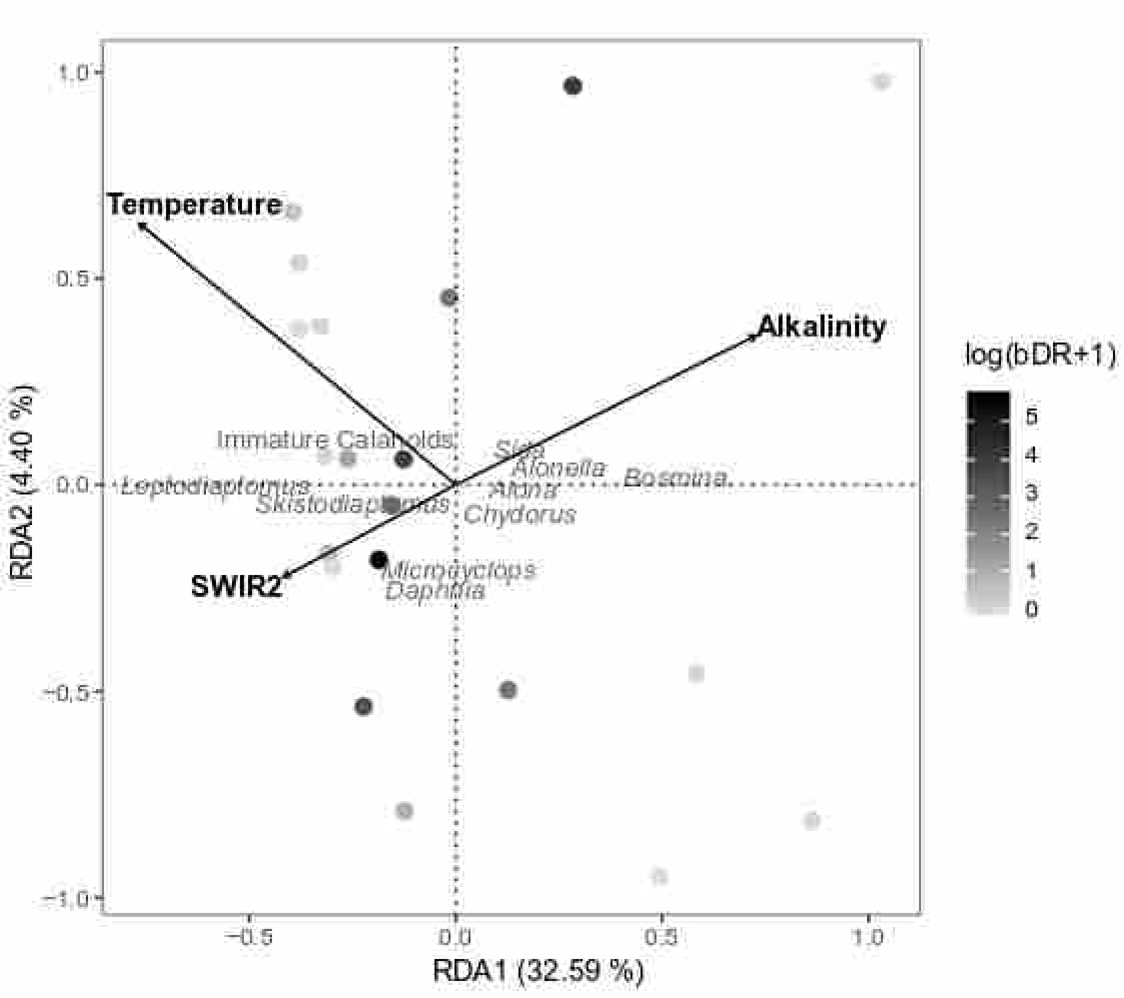


Figure 4: Principal component analysis (PCA) showing water quality variables (black text) and physicochemical variables (gray and italicized text) of lakes sampled in August 2018 and 2019 (points) within the Sahtú Settlement Area (SSA). Darker points represent lakes with a higher log burned drainage ratio +1 (bDR). Calcium = Ca, dissolved oxygen = DO, nitrogen: phosphorous ratio = N/P, total nitrogen = TN, total organic carbon = TOC, macrophyte biomass = SWIR1 and SWIR2, dissolved organic carbon = DOC, ammonium = NH_4, nitrite = NO_2, % organic sediment = LOI, iron = Fe, and total phosphorous = TP.



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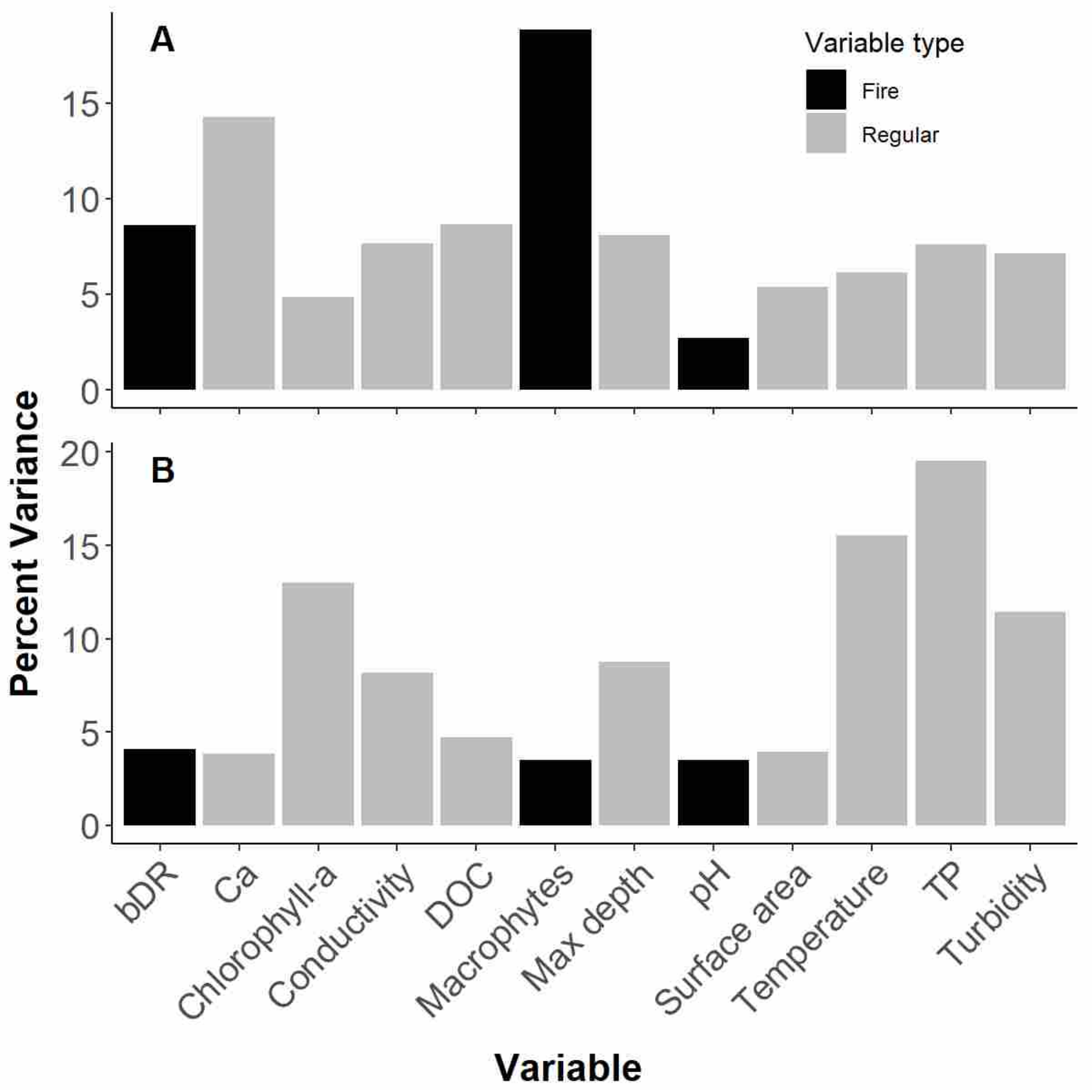

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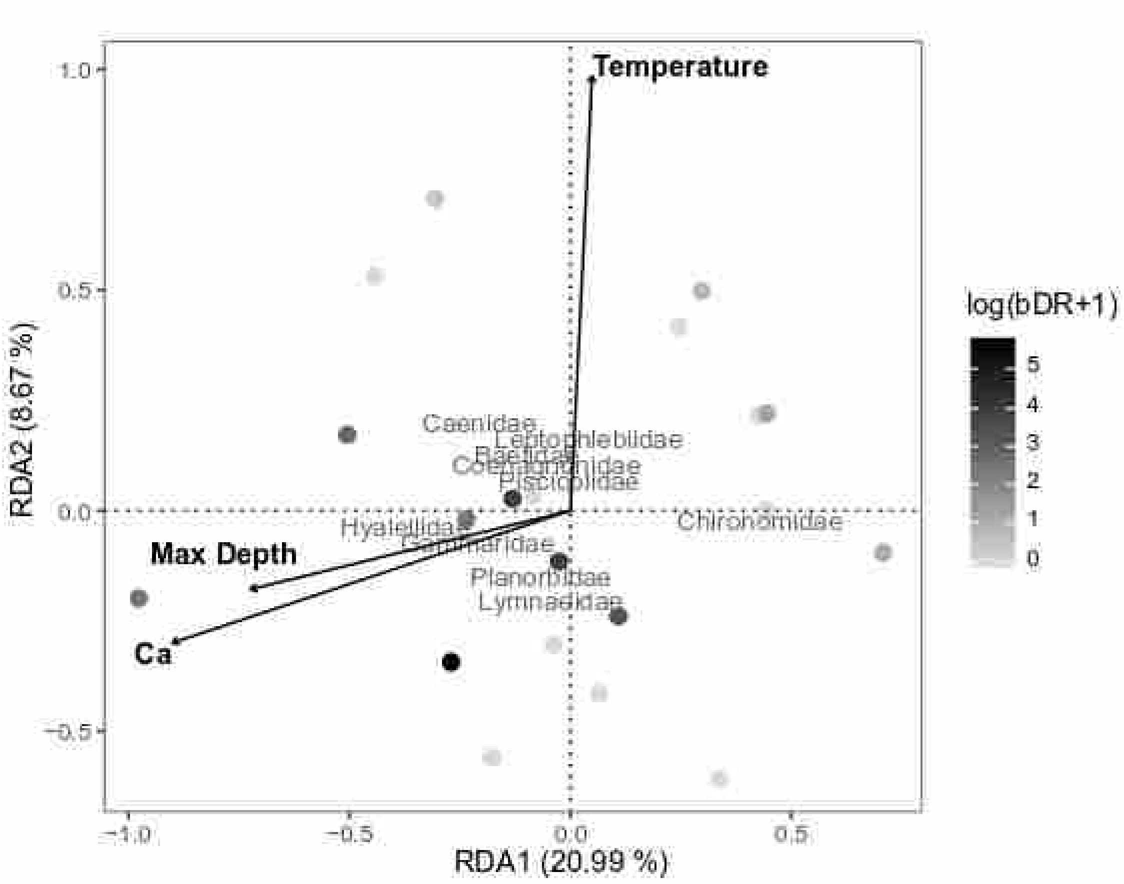

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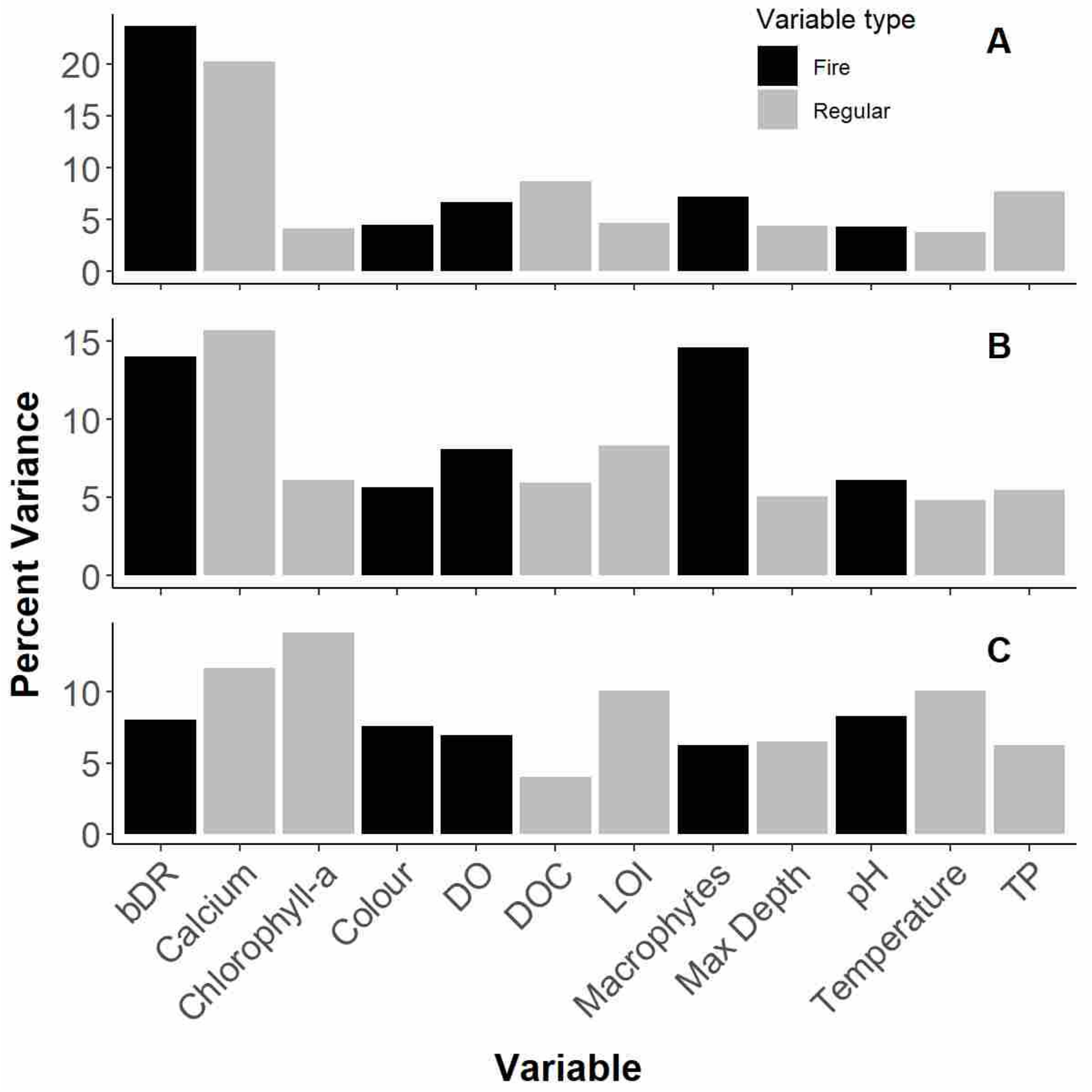
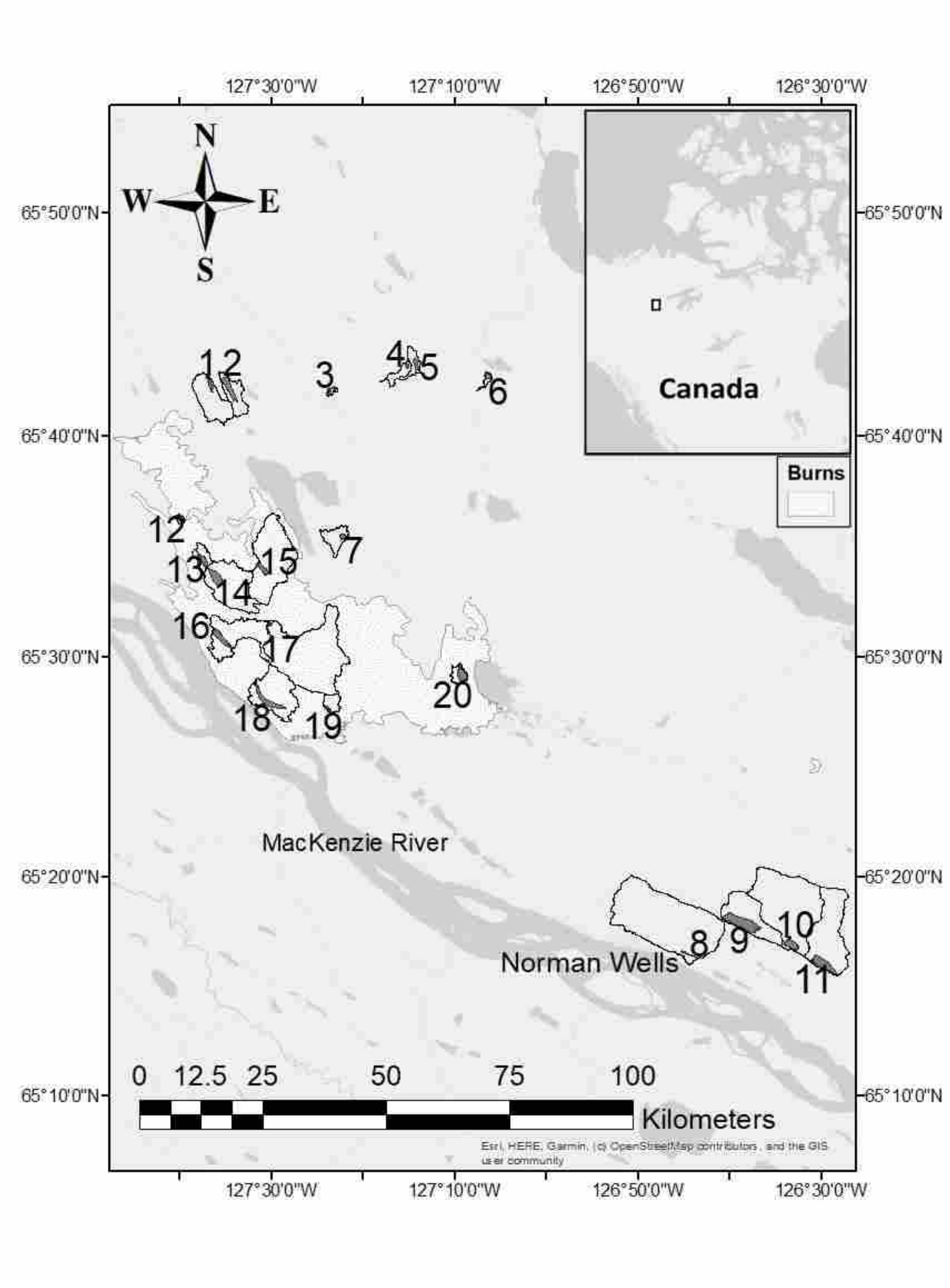


Figure 8*:* Results of hierarchical partitioning to determine variables most strongly associated with differences in benthic macroinvertebrate taxa richness (A), total abundance (B), and EPT abundance (C) among lakes. Variables were identified as "fire" variables if they showed a correlation with bDR (total hectares burned /lake surface area) in Spearman correlations. Dissolved oxygen = DO, dissolved organic carbon = DOC, % organic sediment = LOI, SWIR2 = Macrophytes, and total phosphorous = TP.

B). Finally, water quality variables related to differences in sensitive species (EPT) among lakes included chlorophyll-*a*, calcium, percent organic matter in sediments, and temperature **(**



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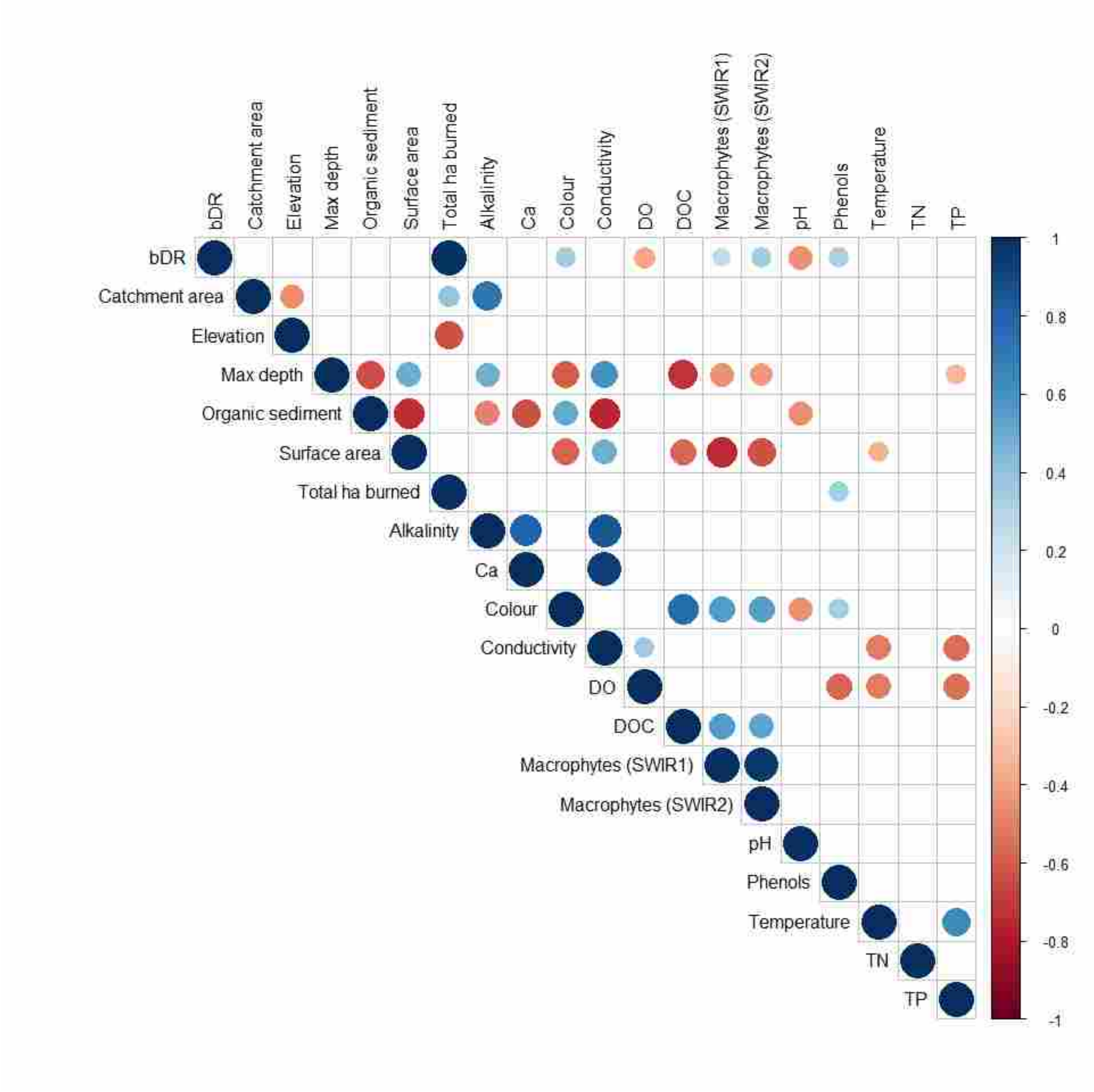

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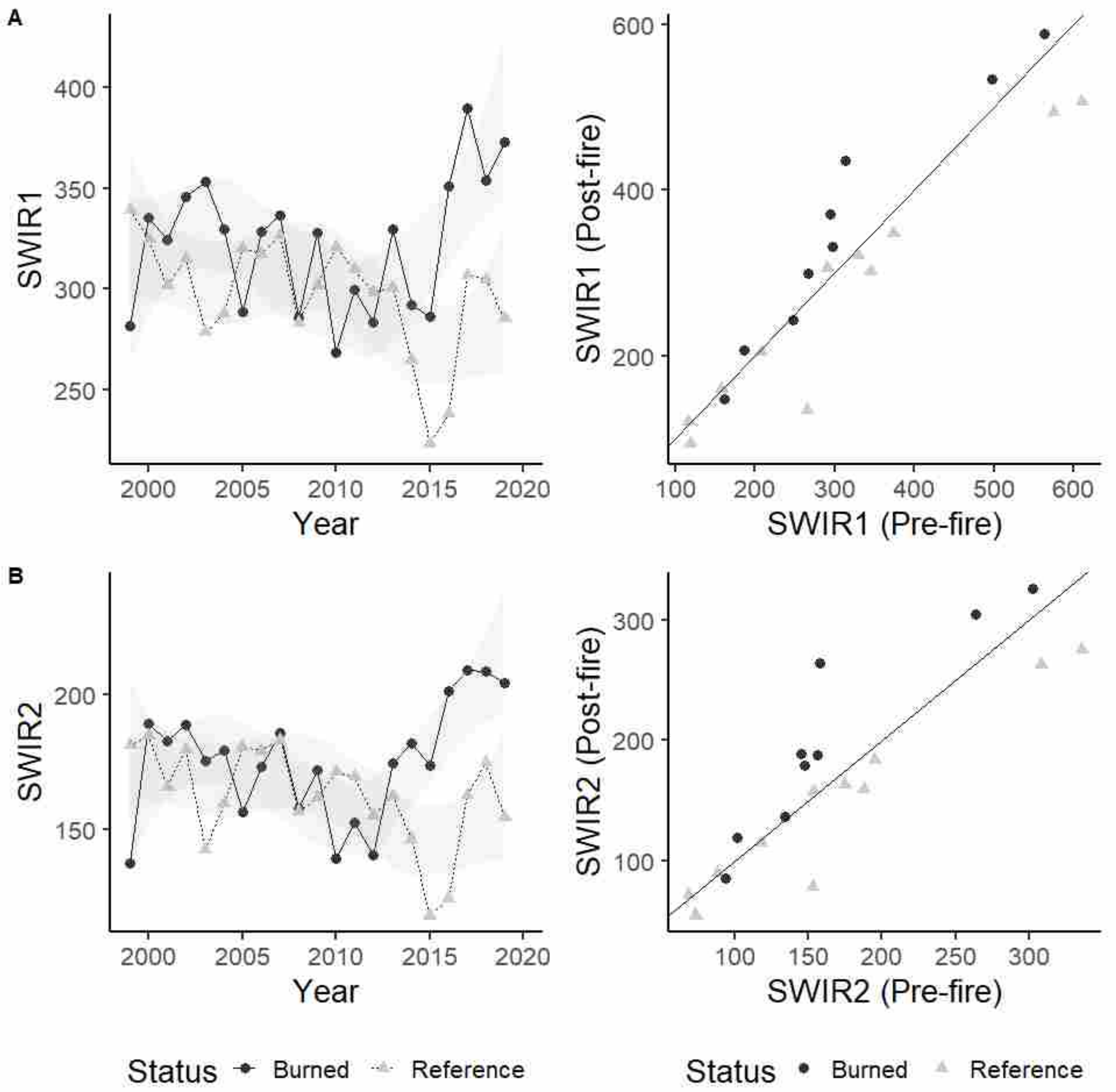

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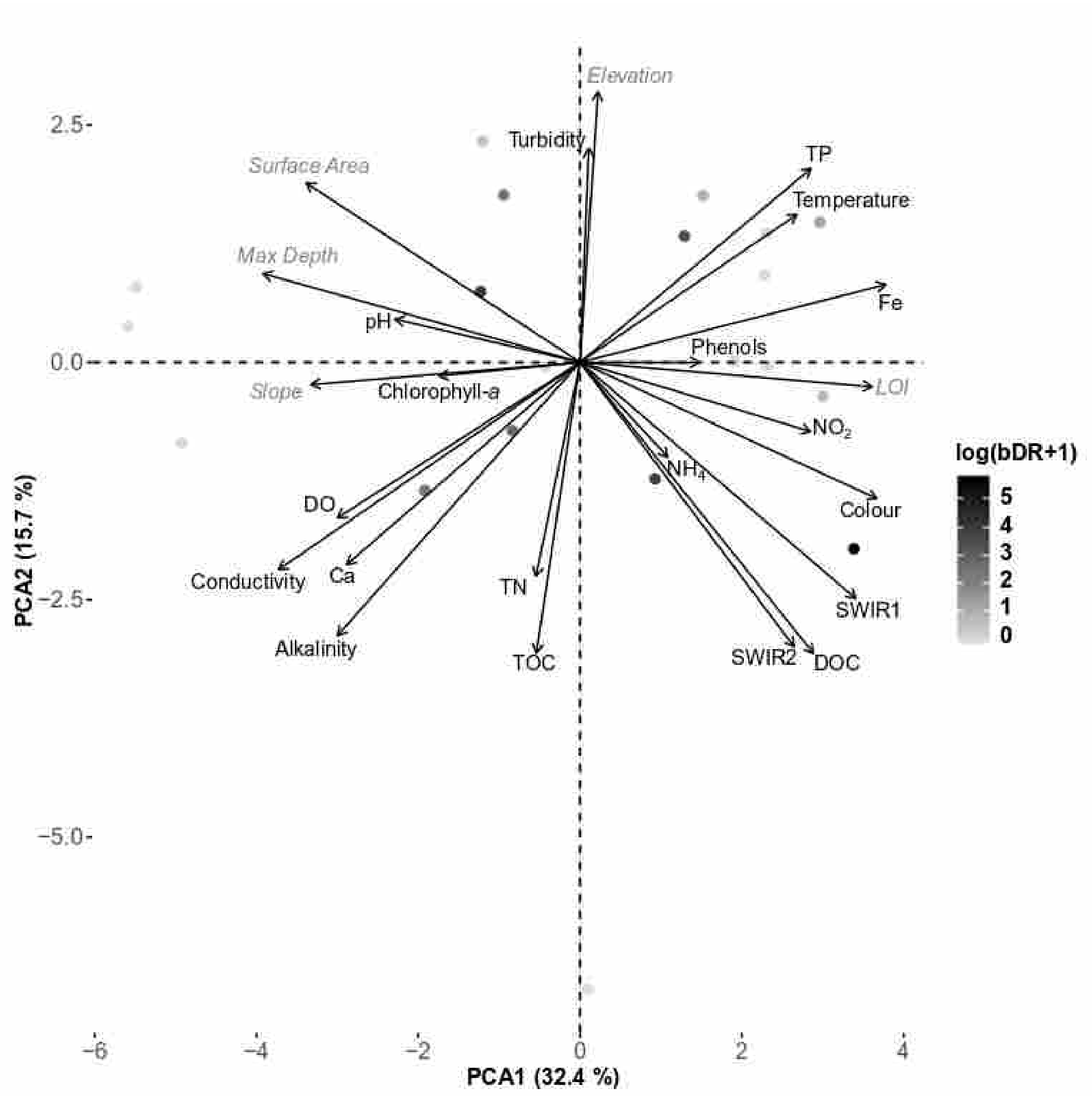

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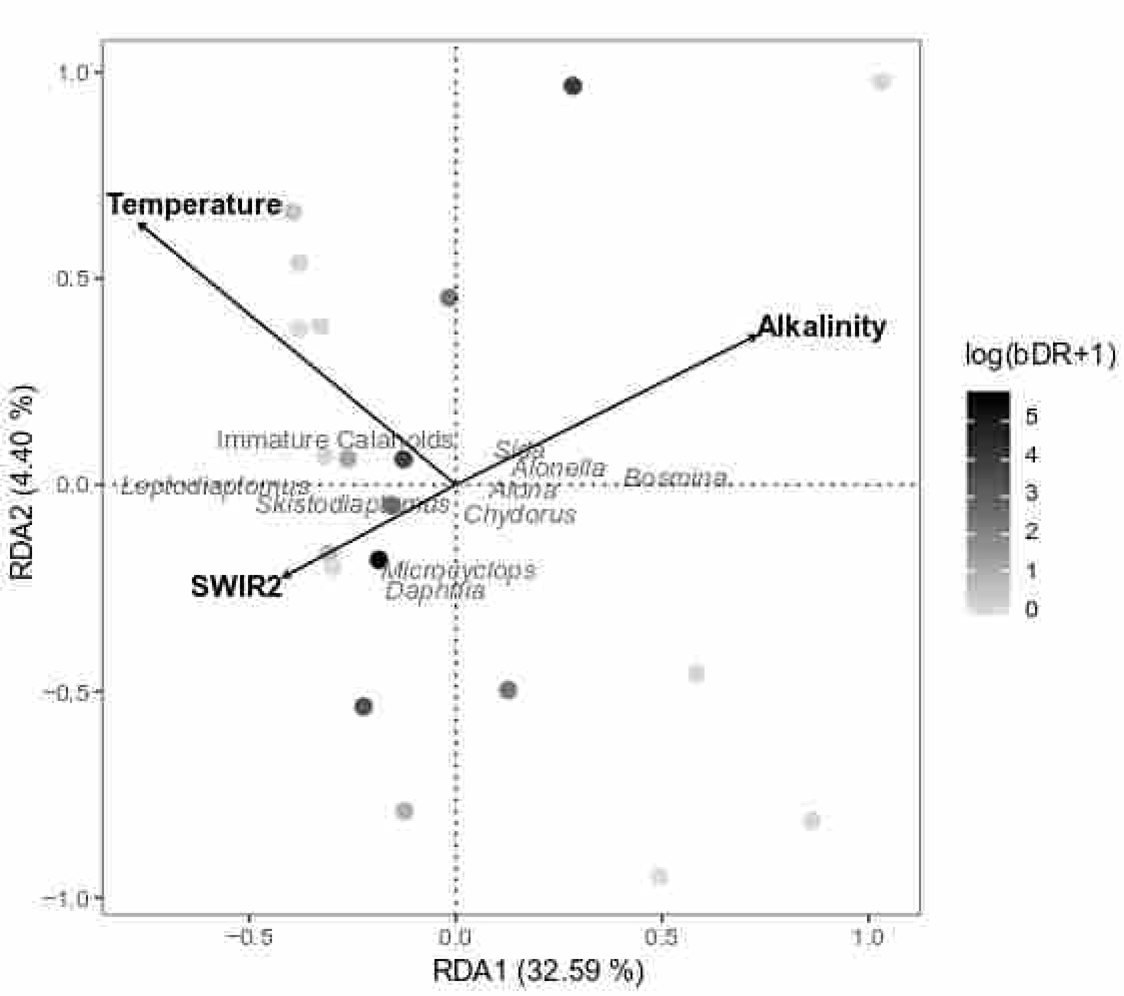

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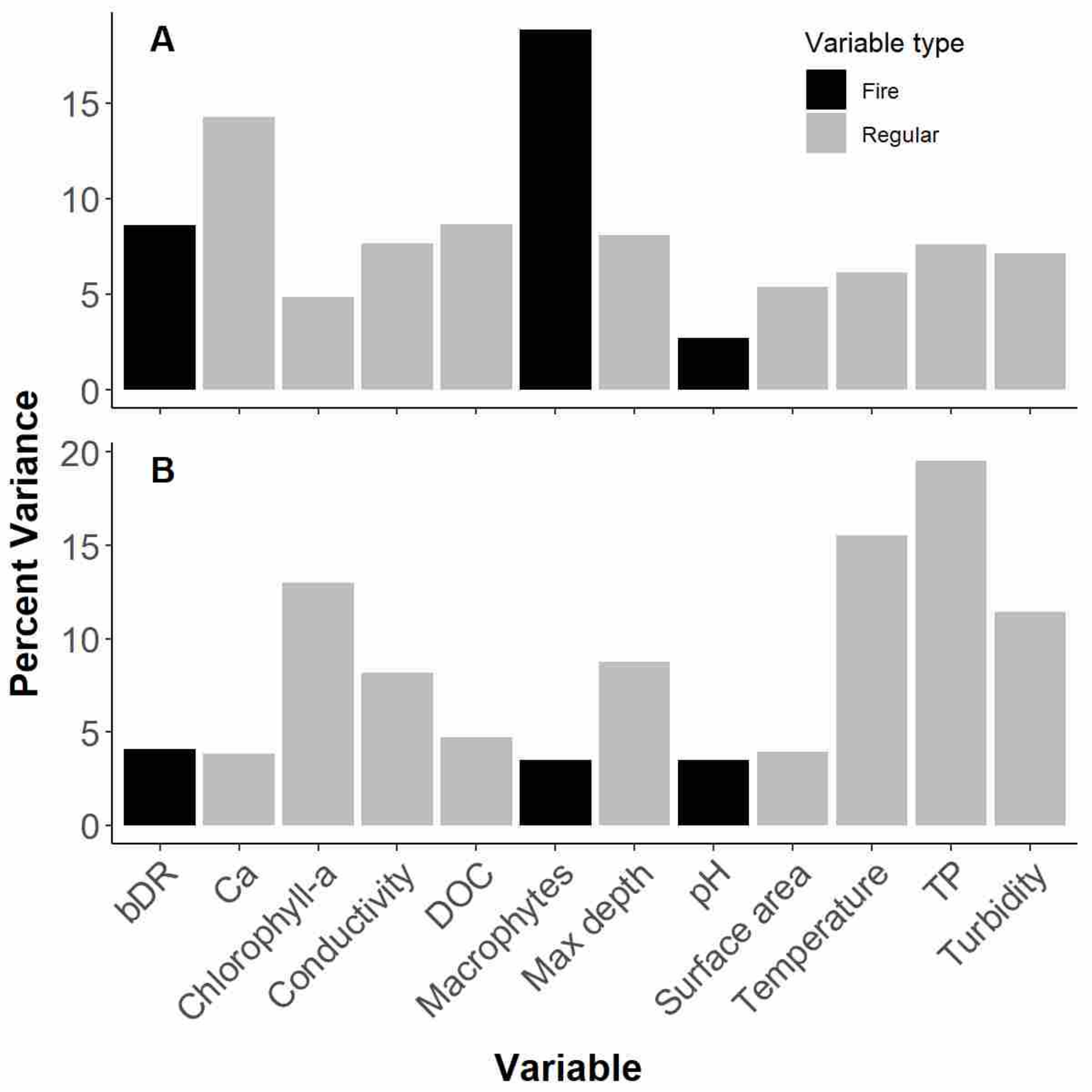

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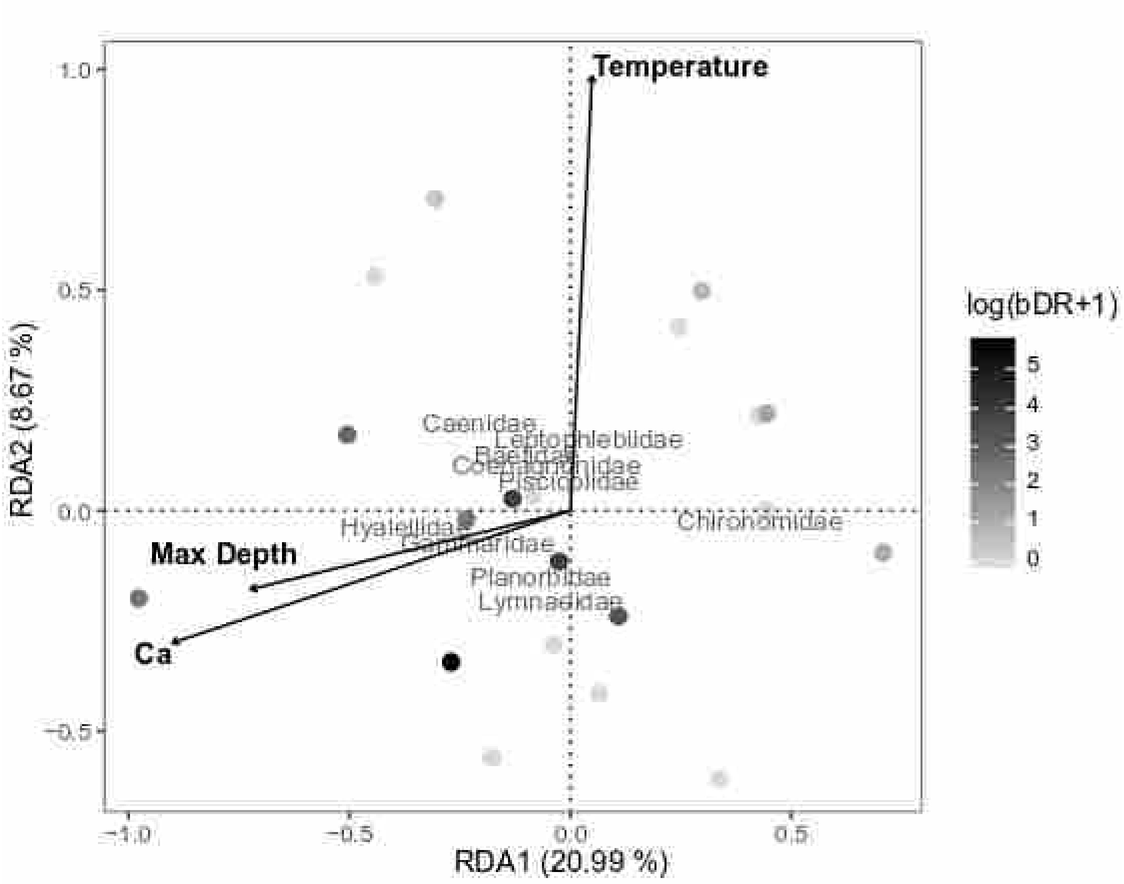

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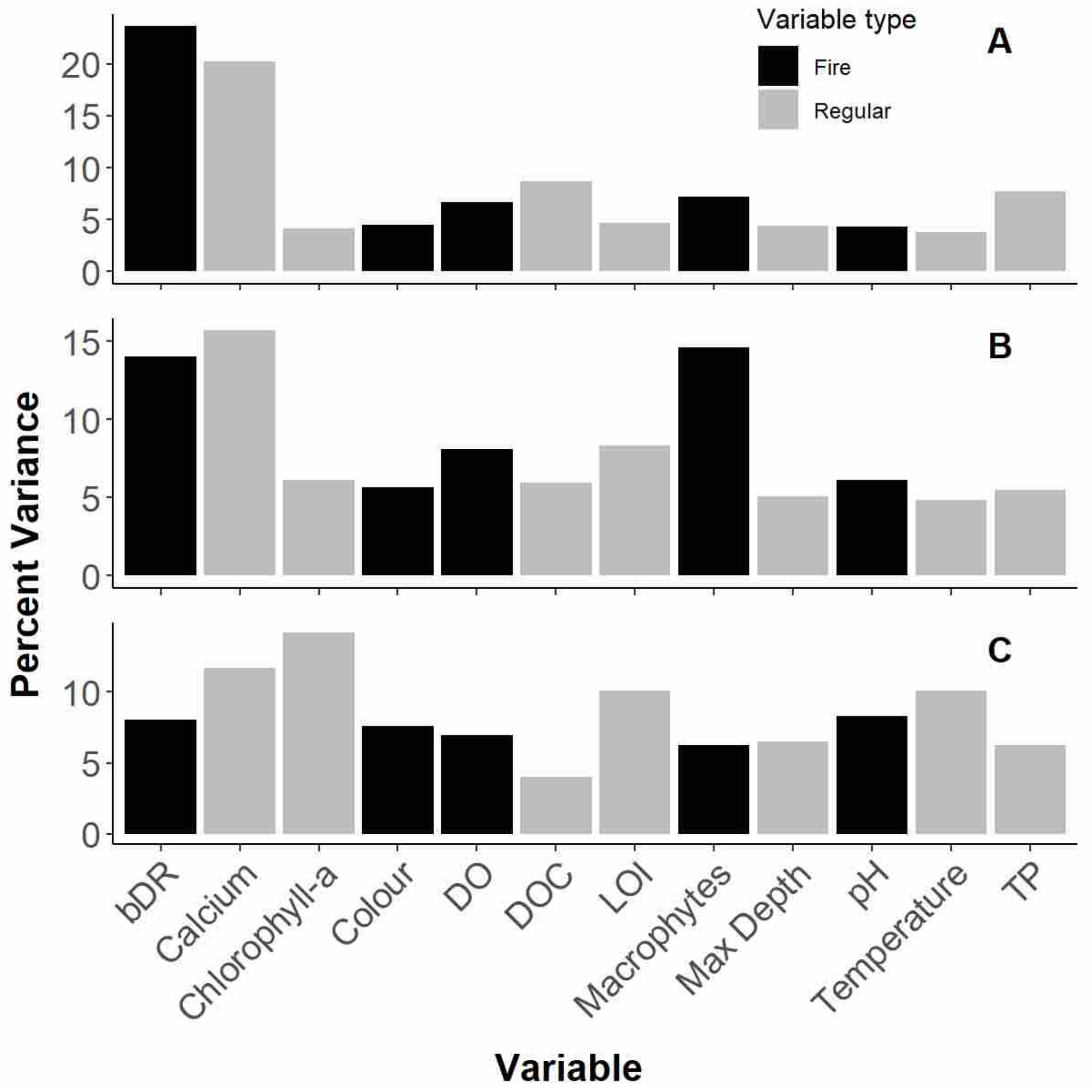

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C).

Discussion

Surprisingly, there were few differences in the water quality of our study lakes based on burn history in their catchments. The only water quality variables correlated with burned drainage ratio (hectares burned/lake surface area) were colour, pH, phenols, and dissolved oxygen. Increased colour and phenols, and decreased pH have been recorded in previous studies following wildfires (Schindler et al. 1996; Garcia and Carignan 1999; Carignan et al. 2000; McEachern et al. 2000; Scrimgeour et al. 2001; Korosi et al. 2015). These changes have been attributed to increases in allochthonous organic matter input from the watershed, which can lead to increased DOC in lakes (composed partly of organic acids; Carignan et al. 2000; McEachern et al. 2000). In addition, McEachern et al. (2000) hypothesized that divalent cations and potassium released following wildfires can undergo cation exchange in peat, releasing protons that can further contribute to acidification.

The reason for the correlation between bDR and dissolved oxygen in our study is less clear, as past studies have not found a significant decrease in DO following wildfires. Since both phenols and colour were positively correlated with bDR, light penetration would likely have been lower in lakes affected by burns, possibly reducing photosynthesis and oxygen production. Alternatively, higher biomass of plants in lakes with higher bDR values could mean that significant shading has occurred, lowering DO production via photosynthesis (Spence and Chrystal 1970; Binzer et al. 2006). This is consistent with the increase in shortwave infrared reflectance observed in the remote sensing data, which is known to covary with aquatic vegetation biomass. It’s also possible that leachates from decaying vegetation (such as phenols) increased the biochemical oxygen demand (Carpenter et al. 1979; Stansbury et al. 2008) in lakes with a high bDR. The reason we did not detect other water quality relationships with burn history (e.g. increased nutrients) could be a result of the timing of our study. While many studies have examined affected lakes 1-2 years after wildfire (McEachern et al. 2000; Patoine et al. 2000; Garcia and Carignan 2005; Allen et al. 2005), our sampling was conducted 4-5 years after the 2014 wildfires in our study region. However, it is worth noting that some studies conducted within 1-2 years after wildfire have also failed to detect increases in nutrients (Lewis et al. 2014).

While few water quality differences were correlated with burn history, we detected changes in one habitat variable important for invertebrates: SWIR median reflectance. With low concentrations and no difference between reference and burned lakes, we do not recognize the increase in SWIR median reflectance in relation to bDR to be resultant of changes in chlorophyll-*a,* cyanobacteria, or lake depths. SWIR reflectance is additionally not affected by increased CDOM and differences in turbidity; SWIR signals are effectively absorbed by surface water with the presence of turbidity (Menken et al. 2006; Wang et al. 2008; Vanhellemont and Ruddick 2015; Tian and Philpot 2015). Based on our observations in the field, and changes observed in the remote sensing data, we speculate that the differences in SWIR reflectance primarily correlate with reflected light from the green leaf surfaces of macrophyte biomass (**see Appendix Figure S1**; Khanna et al. 2011; Oyama et al. 2015; Yadav et al. 2017). Large concentrations of macrophytes were observed in lakes affected by wildfire but were less common at reference lakes (T. Pretty, personal observation; **see Appendix Figure S2**). We could not find any studies that have examined changes in macrophytes following wildfires in lakes, but a recent study on streams showed that deposition of wildfire-related nutrient-rich sediments allowed for a significant increase in macrophytes for three years after the wildfires (Thompson et al. 2019). Unfortunately, we did not measure sediment N and P levels, so we are unable to confirm that the same mechanism is responsible for our study lakes. We should also note that SWIR values were negatively correlated with maximum depth and lake surface area, which were both slightly lower in our study lakes that were exposed to wildfires. It is, therefore, possible that small differences in lake size may have led to unexpectedly large differences in the aquatic plant communities between lakes affected by burns and those that were not (Gasith and Hoyer 1998; Lacoul and Freedman 2006). However, our time-series of SWIR data show divergence after the 2014 wildfires, suggesting that these physical differences in the lakes are not likely to be responsible for the relationship we found between bDR and SWIR reflectance readings. The changes in SWIR would therefore suggest increased presence of macrophytes (Wang et al. 2013) in relation to bDR.

Our redundancy analysis showed that there were no differences in the relative abundance of zooplankton species based on fire history. These results agree with those from Jalal et al. (2005) and Patoine (2002) who both found no differences in zooplankton community composition three years after the wildfire. They also suggest that, contrary to our expectations, the impacts of wildfire do not necessarily extend for longer periods at more northern sites, or that northern sites don’t respond to wildfires. However, as mentioned previously, the biomass of macrophytes was affected by burns following the 2014 wildfires, and macrophyte biomass was an important predictor for the zooplankton community structure.

In addition to macrophyte biomass, our RDAs showed that zooplankton communities were controlled by temperature and alkalinity. Warmer lakes with more macrophytes were dominated by calanoid copepods, while colder lakes with fewer macrophytes and higher alkalinity tended to have more of the cladocerans *Bosmina, Chydorus, Sida,* and *Alonella*. Macrophytes can serve as an important habitat for some zooplankton groups, and as a refuge from fish predation (e.g. Schriver et al. 1995). Unfortunately, we were unable to gather fish data from our study lakes, limiting our ability to interpret the role fish and macrophytes might have played in structuring zooplankton. The importance of alkalinity for zooplankton in our lakes could relate to its effect in buffering pH levels (especially in lakes located within colluvial rubble surficial geology), or its association with calcium, conductivity, and maximum depth in our dataset. Lakes with higher alkalinity, conductivity, and calcium levels tended to be dominated by cladocerans, which are typically more sensitive to changes in pH and levels of calcium (Labaj et al. 2014; Jeziorski et al. 2014), while lakes with lower alkalinity were dominated by copepods, including members of the acid-tolerant genus *Leptodiaptomus* (Gray and Arnott 2012).

Results of our hierarchical partitioning analysis showed that zooplankton richness was most associated with macrophyte biomass and calcium levels. The association of richness with macrophyte biomass is likely related to the role that plants can play in increasing habitat complexity and offering a refuge from fish predators (Lauridsen et al. 1996; Portielje and Van Der Molen 1999).

Calcium is important for supporting daphniids, which require relatively high concentrations to build their carapace (Cairns and Yan 2009). Zooplankton abundance was most strongly associated with temperature, total phosphorus, and chlorophyll-*a*. Temperature affects the metabolic rate of zooplankton, such that warmer temperatures can increase the net reproductive rate, allowing for higher abundances (Heinle 1969; Gillooly 2000). Higher total phosphorus levels typically lead to higher productivity in boreal subarctic lakes, providing more energy to support an abundant zooplankton community (Bergström et al. 2018).

Our RDA showed that benthic macroinvertebrate communities differed among lakes with different catchment burn histories, with calcium and maximum depth playing an important role.

Comparably with daphniids, calcium is an important component to the carapace structure and distribution of amphipods and gastropods (Cairns and Yan 2009; Dalesman and Lukowiak 2010). Alternatively, the increase in amphipods may have something to do with the disturbance of lakes affected by burns. Both Lewis et al. (2014) and Scrimgeour et al. (2001) found more amphipods in lakes affected by wildfires, and they suggested that their success had to do with their ability to opportunistically switch food sources after wildfire, taking advantage of the pulse of allochthonous carbon coming to a lake. Amphipods, such as Gammaridae, also have short generation times (Welton and Clarke 1980), allowing them to increase in abundance faster than other benthic macroinvertebrates such as Ephemeroptera or Odonata, which reproduce at much longer intervals (Lewis et al. 2014). Differences observed in temperature may have derived from sampling over two consecutive years, but warmer temperatures are important for egg development in ephemeropterans (Rotvit and Jacobsen 2014). The differences among benthic macroinvertebrate communities shown in our RDA may have also been related to general differences in small versus large lakes in our dataset. Our PCA showed that larger lakes (surface area and depth) had significantly lower DOC, colour, and macrophyte biomass, and higher calcium/conductivity, while small, shallow lakes tended to have more colour, higher DOC, higher macrophyte biomass, and lower conductivity. Furthermore, past studies have shown that lake size and calcium are important variables structuring benthic macroinvertebrate communities (Mousavi 2002; Namayandeh and Quinlan 2011; White et al. 2011). Therefore, it is difficult to disentangle the suite of characters exhibited by the shallow lakes versus those typical of the deep lakes in our dataset in terms of the contribution of individual variables toward structuring benthic macroinvertebrate communities.

Hierarchical partitioning results for benthic macroinvertebrates showed that burned drainage ratio was the most important predictor for richness and the third most important for abundance. This was an interesting result, as it is unclear which water quality variables affected by bDR could be causing differences in richness or abundance. As mentioned above, bDR was correlated with colour, dissolved oxygen, macrophyte biomass, pH, and phenols in our study. One possibility is that bDR is a good composite variable that expresses differences in these correlated water quality variables among our lakes. Alternatively, bDR may be reflecting some aspect of water or habitat quality related to wildfires that were not quantified in our study. For example, we were unable to measure fish abundance and biomass, or linkages to terrestrial biota which could have been affected by wildfires. Despite the importance of bDR in our analysis, the total variability in the abundance and richness of benthic macroinvertebrates explained by other variables was much greater, suggesting that natural variability among lakes is more important. The abundance of sensitive EPT species in our hierarchical partitioning analysis was also best predicted by natural variation in variables that could not be related to wildfire history in our study, including temperature, chlorophyll-*a*, calcium, and % sediment organic matter. Several past studies show the importance of these variables for determining the abundance of EPT in macroinvertebrate communities (Tabak and Gibbs 1991; Rotvit and Jacobsen 2014; dos Reis Oliveira et al. 2020).

In summary, we found relationships between burn history in lake catchments and some water quality variables, including colour, dissolved oxygen, and pH. A relationship between SWIR reflectance suggestive of macrophyte biomass being affected by burn history was also observed. However, we did not find a relationship between burn history and lake nutrient concentrations, as has been found in past studies. The relative abundance of zooplankton genera in our study lakes did not differ based on wildfire history, but the burned drainage ratio (bDR) was an important predictor of zooplankton genus richness. We detected differences in the relative abundance of benthic macroinvertebrates, with lakes affected by burns having more amphipods. In addition, the burned drainage ratio was an important predictor of richness and abundance for benthic macroinvertebrates. However, the differences in benthic macroinvertebrate communities were also driven by variables that were unrelated to burn history (e.g. lake depth and calcium), and variables unrelated to wildfires together had a much larger effect on richness and abundance than bDR. Taken together, these results suggest that differences in zooplankton, and benthic invertebrate communities among our study lakes can be attributed to a mixture of natural variation in physicochemical variables along with the residual impacts from wildfires (e.g. altered macrophyte biomass). This indicates that effects of wildfire on biological communities in boreal lakes can still be detected 4-5 years after the wildfires, and provides some support for the notion that the effects of wildfires might be longer-lasting in the northern boreal forest due to shorter growing seasons and the involvement of permafrost. Unfortunately, our ability to reach a definitive conclusion on the resilience of these lakes to wildfire is hindered by the correlative nature of our analyses. We hope that future studies will be able to incorporate designs (e.g. BACI, core sampling, or weekly/monthly sampling) that allow for more robust time-series inferences about the response of lakes in the northern boreal to wildfires. Studies on macrophytes in northern regions after fire (e.g. abundance and cover % of macrophytes, or remote sensing over a larger area) would also improve our understanding of the response of lake ecosystems to wildfires.

Table 1: Main physical and water quality parameters observed in lakes sampled in August 2018/2019 in the Sahtú Settlement Area. Significant *P* values between reference lakes and burned lakes are denoted as bold and * when $P < 0.05$, ** when $P < 0.01$, and *** when $P < 0.001$ (Mann-U Test).

	Reference lakes (*N*= 11)			**Burned lakes (*N*= 9)**		
	Percentiles			**Percentiles**		
Variable	**25th**	**Median**	**75th**	**25th**	**Median**	**75th**
burned Drainage Ratio	**0.00**	**0.00*****	**0.00**	**9.78**	**13.52*****	**47.17**
Drainage Ratio (WC/SA)	4.48	14.30	24.84	10.84	15.61	49.59
Elevation (m. a.s.l.)	**245.00**	**260.00****	**301.00**	**99.00**	**213.00****	**229.00**
Maximum Depth (m)	1.18	1.58	6.92	1.86	2.35	3.99
Organic Sed. % (LOI)	35.66	49.33	59.96	32.53	46.76	63.73
Organic Soil % (LOI)	62.06	71.78	91.62	41.59	86.47	94.33
Total ha Burned	**0.00**	**0.00*****	**0.00**	**153.43**	**988.14*****	**1727.69**
Slope (%)	5.70	10.65	14.85	7.10	12.03	12.52
Surface Area (ha)	13.05	25.15	76.03	11.35	40.39	74.36
Water Catchment (ha)	168.52	355.18	1341.16	177.12	1242.07	1816.52
Wetland in WC (ha)	4.69	10.21	28.95	4.31	17.97	26.77
Alkalinity (mg/L $CaCO_3$)	100.00	120.00	155.00	85.00	125.00	150.00
Ca (mg/L)	24.31	45.40	51.73	22.72	38.64	48.00
Chlorophyll-*a* (µg/L)	0.20	0.38	0.56	0.23	0.34	0.47
Colour (mg/L Pt)	7.50	45.00	62.50	30.00	55.00	70.00

Conductivity (µS/cm)	142.21	276.90	502.75	156.65	213.30	248.43
DO (mg/L)	8.62	10.00	11.29	8.78	9.11	10.00
DOC (mg/L)	8.68	11.73	13.91	9.39	10.88	11.30
Fe (mg/L)	0.00	0.01	0.02	0.02	0.02	0.06
pH	**8.21**	**8.36***	**8.39**	**7.82**	**8.14***	**8.26**
Phenols (mg/L C_6H_5OH)	0.10	0.19	0.22	0.00	0.20	0.38
SWIR1	166.04	311.69	359.93	253.62	315.85	365.44
SWIR2	97.90	164.98	205.24	151.44	185.08	211.78
Temperature (℃)	13.17	14.55	15.66	13.95	14.35	14.43
TN (mg/L)	0.34	0.55	0.66	0.35	0.35	0.44
TOC (mg/L)	9.21	12.12	13.52	9.20	10.49	11.01
TP (mg/L)	0.03	0.05	0.07	0.05	0.05	0.08
Turbidity (NTU)	2.00	4.00	8.00	6.00	6.00	245.00

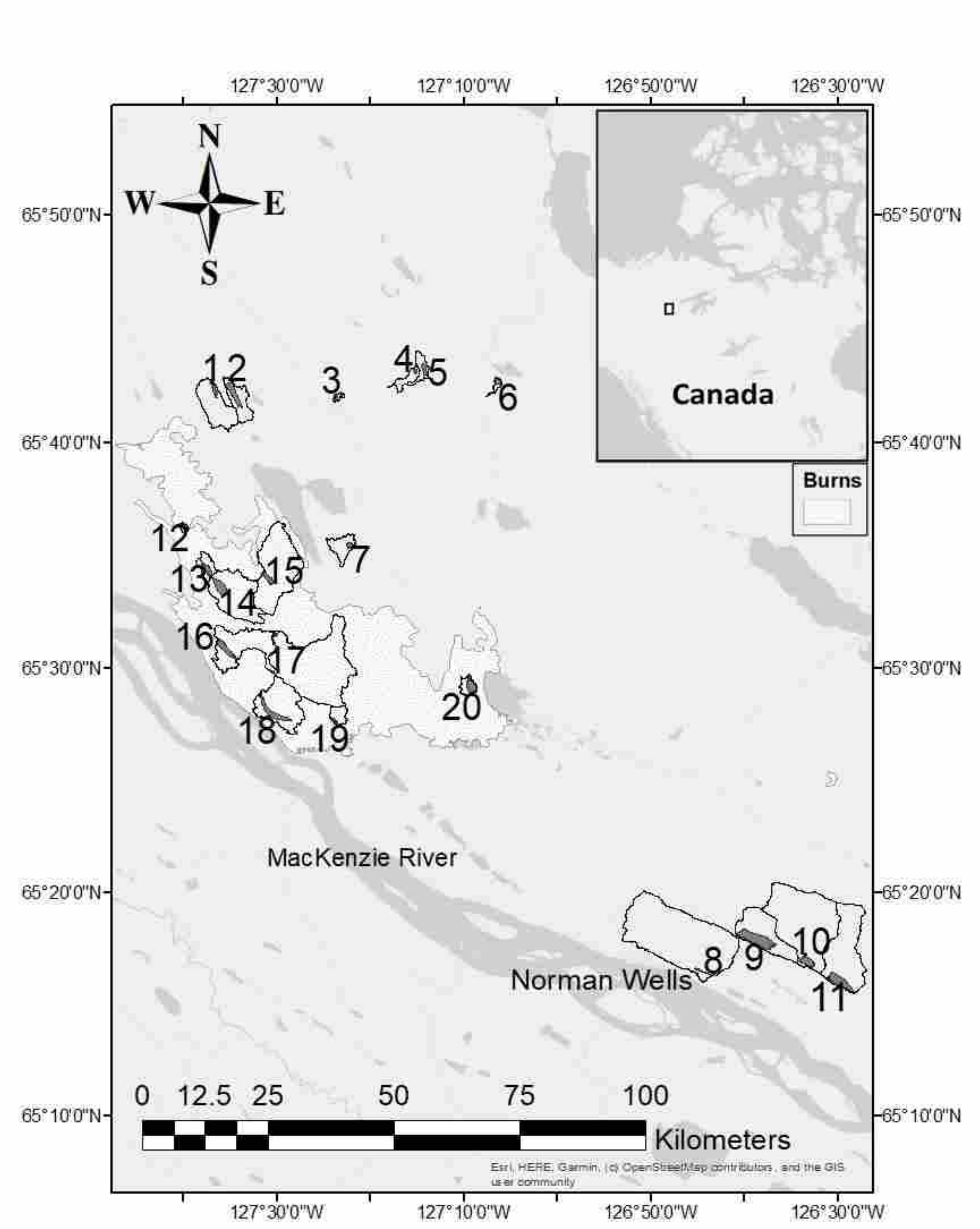

Figure 1: Locations of lakes and respective water catchments (WC) sampled, and the fire-scarred areas from the 2014 and 2015 wildfire years. Numbers represent locations of individual lakes sampled.

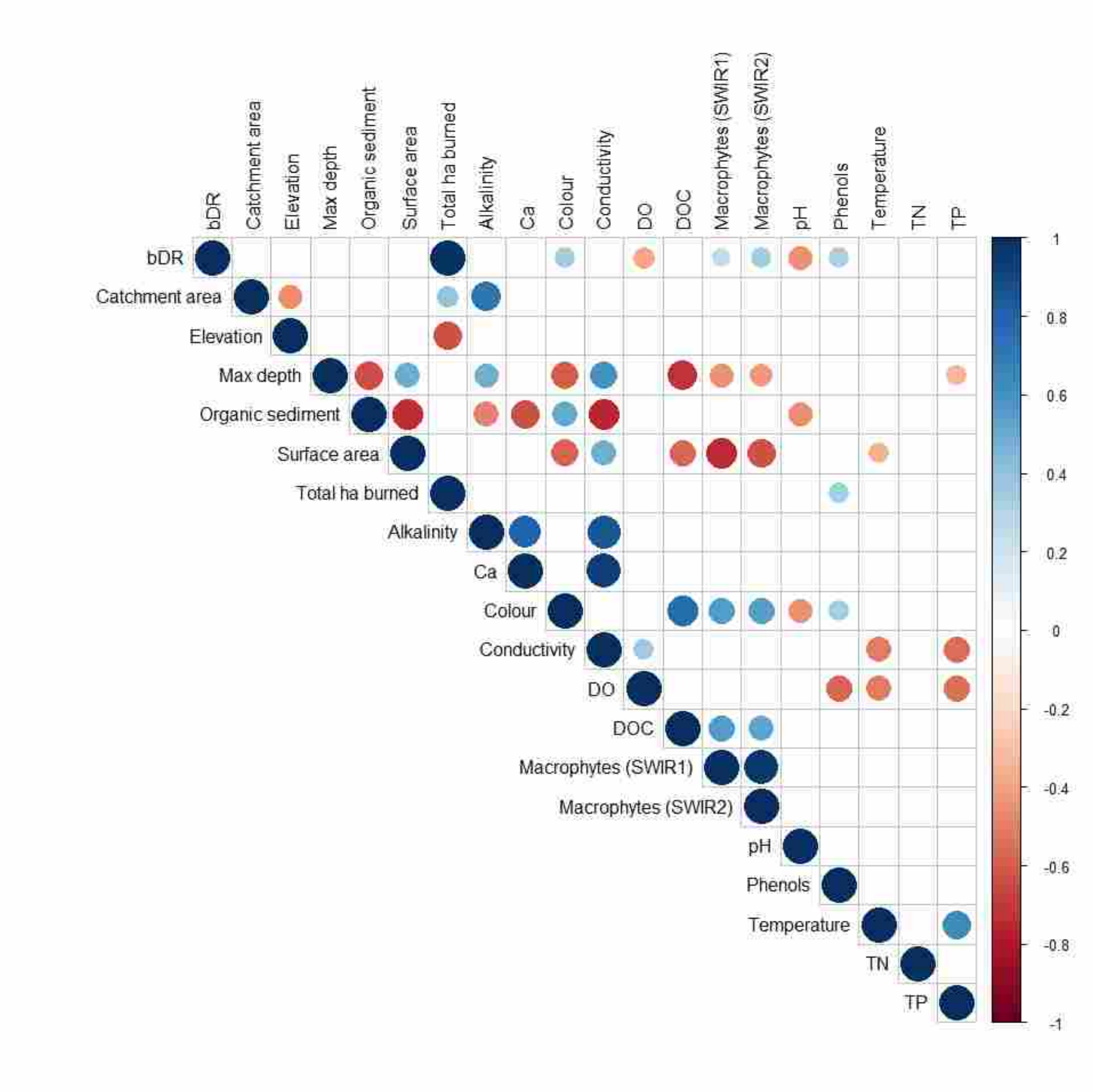

Figure 2: Spearman correlations between the burned drainage ratio (hectares burned/lake surface area), lake physical properties, and water quality variables. Blue is indicative of positive correlations and negative correlations are red. Colour intensity and size of the circle are proportional to the correlation coefficients. Correlations that were not significant (p> 0.05) are indicated by empty cells. Burned drainage ratio = bDR, calcium = Ca, dissolved oxygen = DO, dissolved organic carbon = DOC, total nitrogen = TN, total organic carbon = TOC, and total phosphorous = TP.

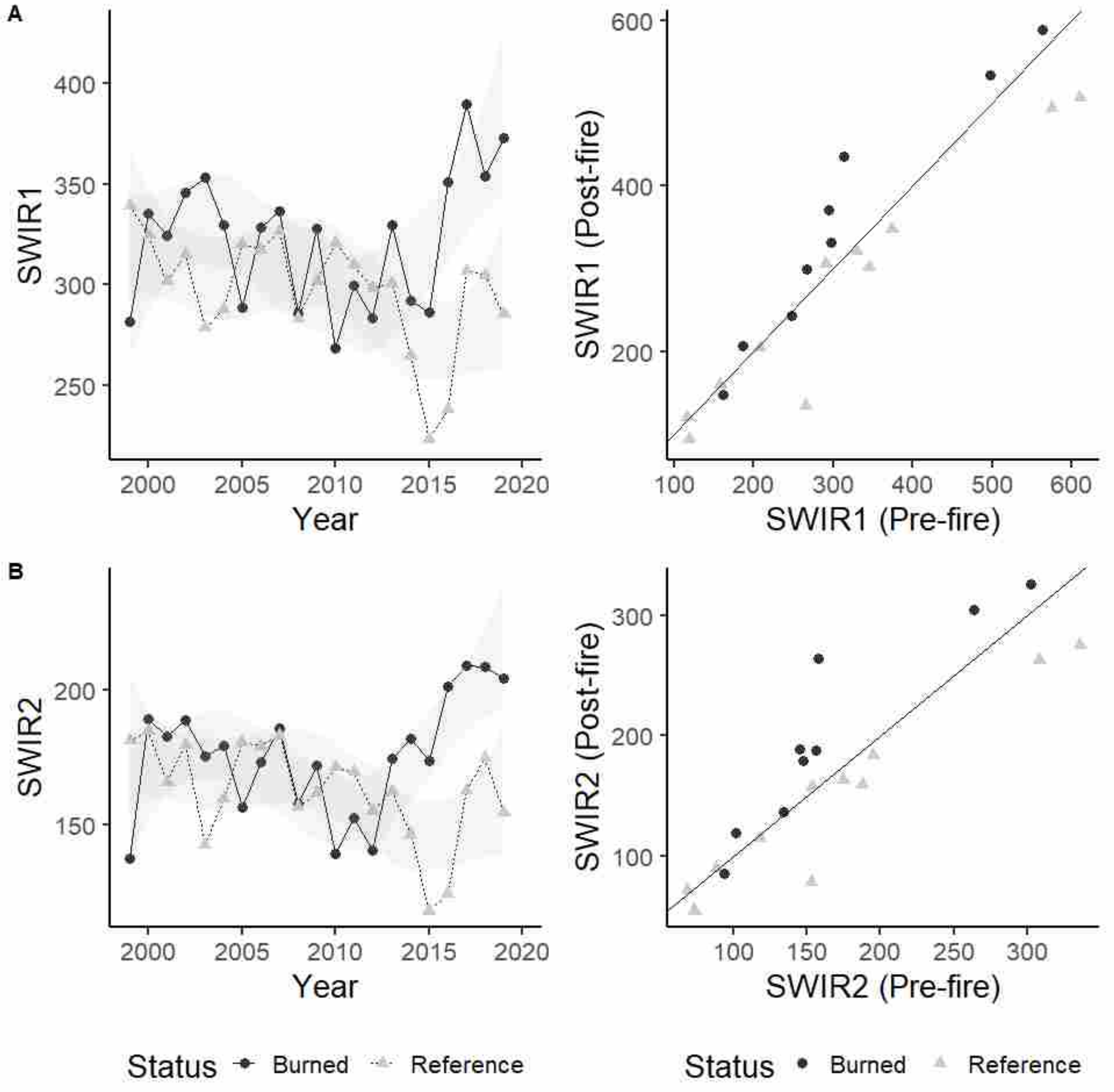

Figure 3: Time-series data of seasonal (June-August) composite medoid atmospheric corrected Landsat 7 and 8 values of mean short-wave infrared imagery SWIR1 and SWIR2 values (left panels). Reference lakes had <16 % of their catchment burned and burned lakes >68 % of their catchment burned. Relationship between mean SWIR values for each lake (9 burned lakes and 11 reference lakes) pre-wildfire (< 2014) and post-wildfire (> 2014) (right panels). The lines in the right panels are 1:1 lines.

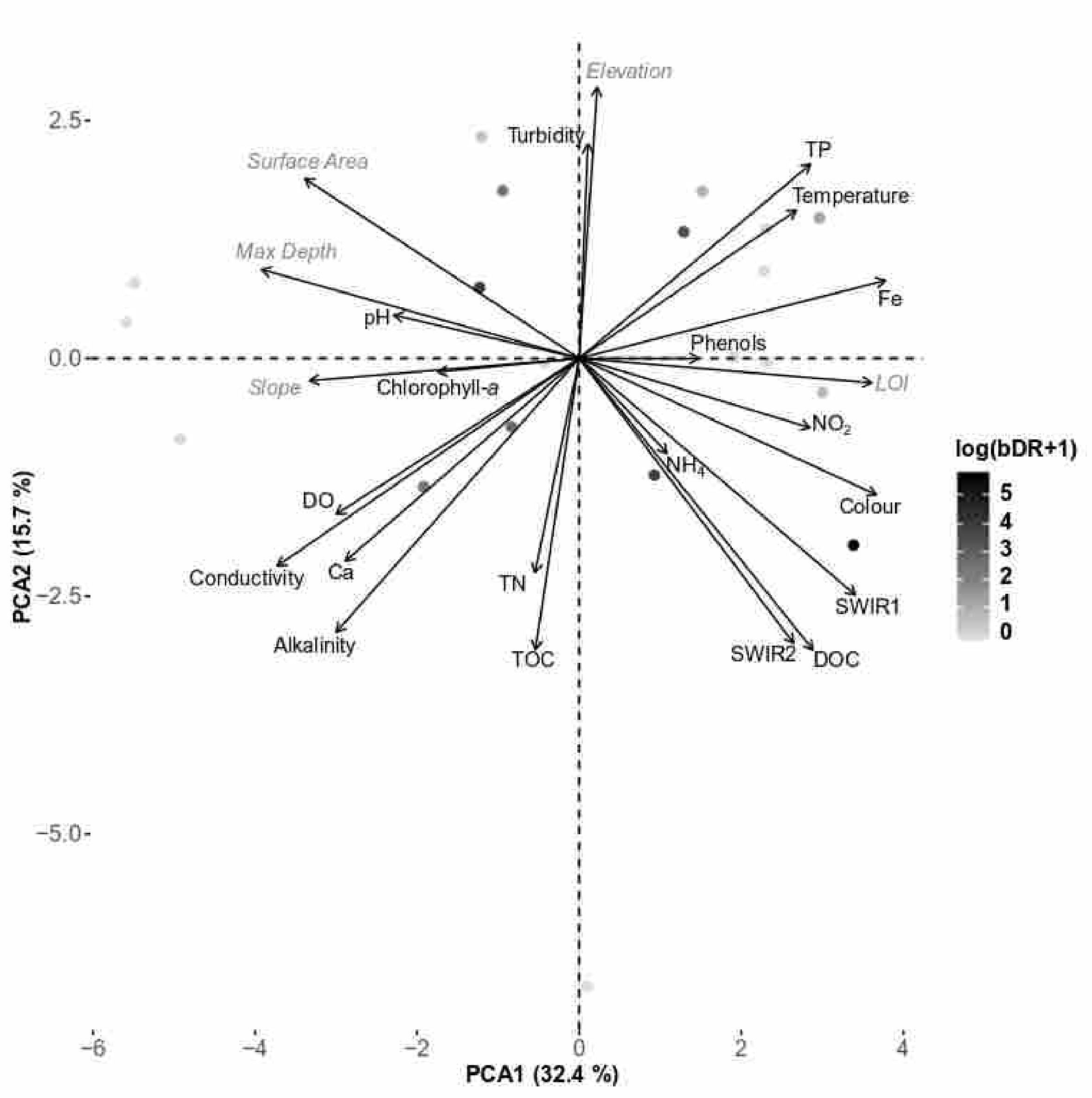

Figure 4: Principal component analysis (PCA) showing water quality variables (black text) and physicochemical variables (gray and italicized text) of lakes sampled in August 2018 and 2019 (points) within the Sahtú Settlement Area (SSA). Darker points represent lakes with a higher log burned drainage ratio +1 (bDR). Calcium = Ca, dissolved oxygen = DO, nitrogen: phosphorous ratio = N/P, total nitrogen = TN, total organic carbon = TOC, macrophyte biomass = SWIR1 and SWIR2, dissolved organic carbon = DOC, ammonium = NH_4, nitrite = NO_2, % organic sediment = LOI, iron = Fe, and total phosphorous = TP.

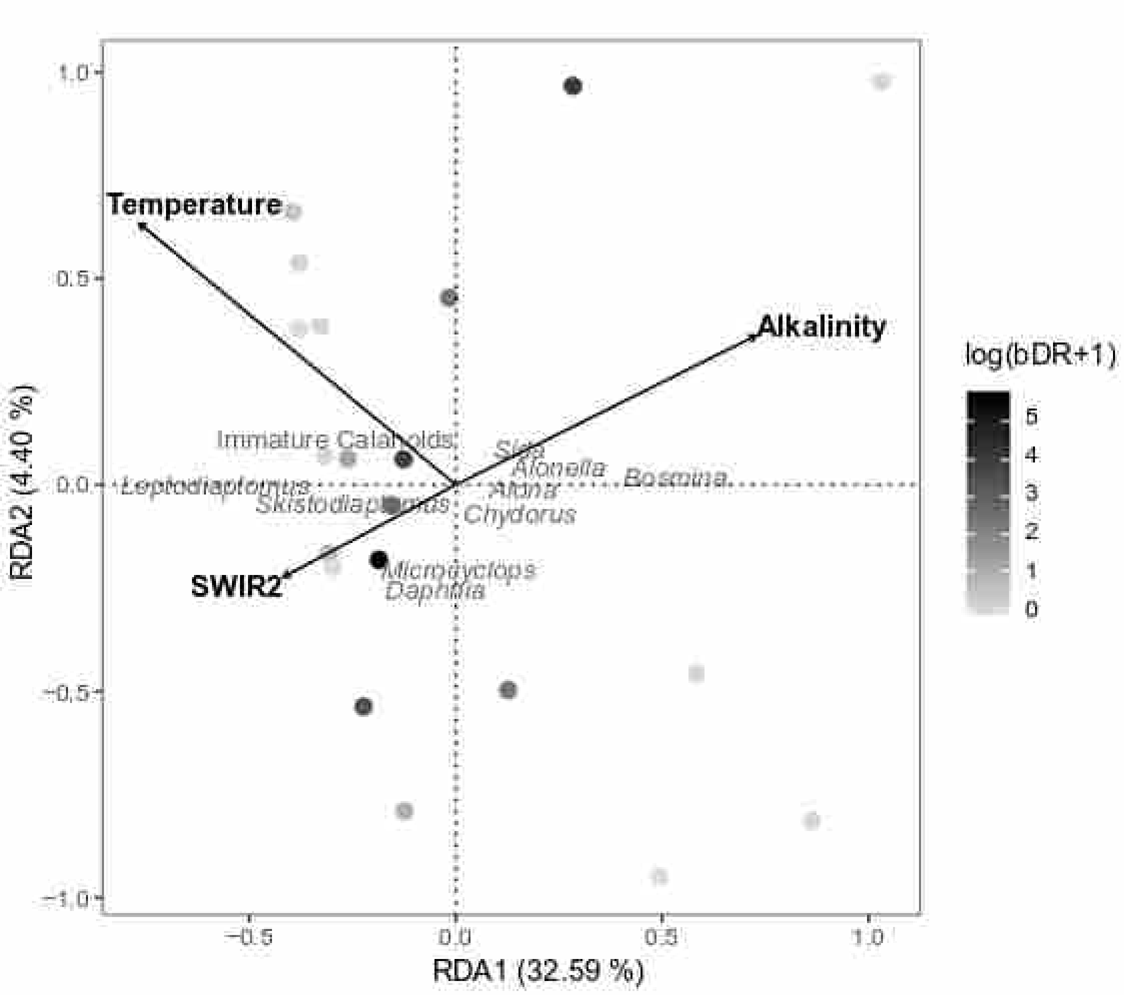

Figure 5: Redundancy analysis (RDA) of zooplankton relative abundances within the Sahtú Settlement Area (SSA). Zooplankton genera labels clustered at the center of the plot were removed for clarity and others were moved slightly to avoid overlap. Darker points represent lakes with a higher log bDR (number of hectares burned / lake surface area). Remotely sensed macrophyte biomass = SWIR2.

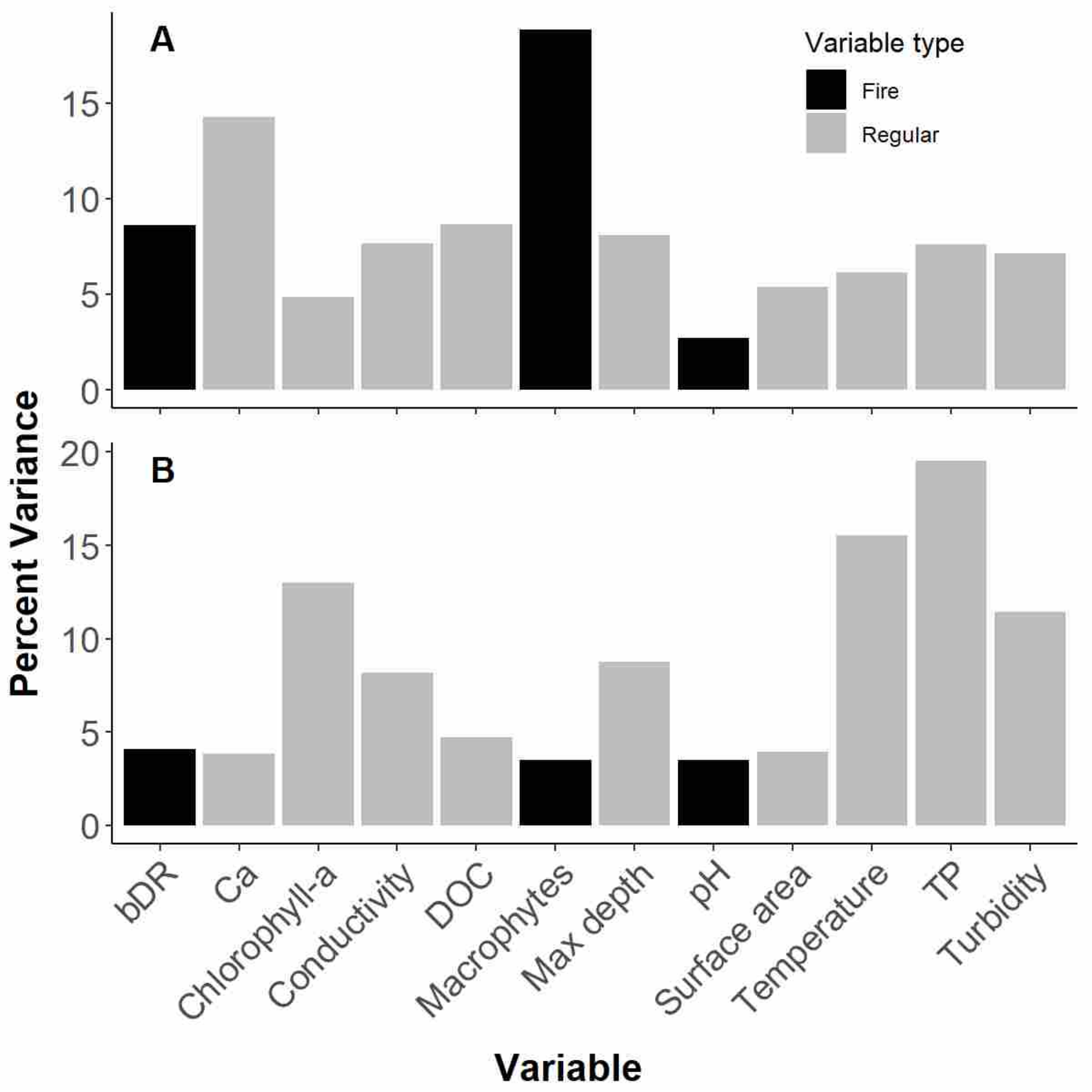

Figure 6: Results of hierarchical partitioning to determine variables most strongly associated with differences in zooplankton richness (A) and abundance (B) among lakes. Variables were identified as “fire” variables if they showed a correlation with bDR (total hectares burned /lake surface area) in Spearman correlations. Calcium = Ca, dissolved organic carbon = DOC, SWIR2 = Macrophytes, and total phosphorous = TP.

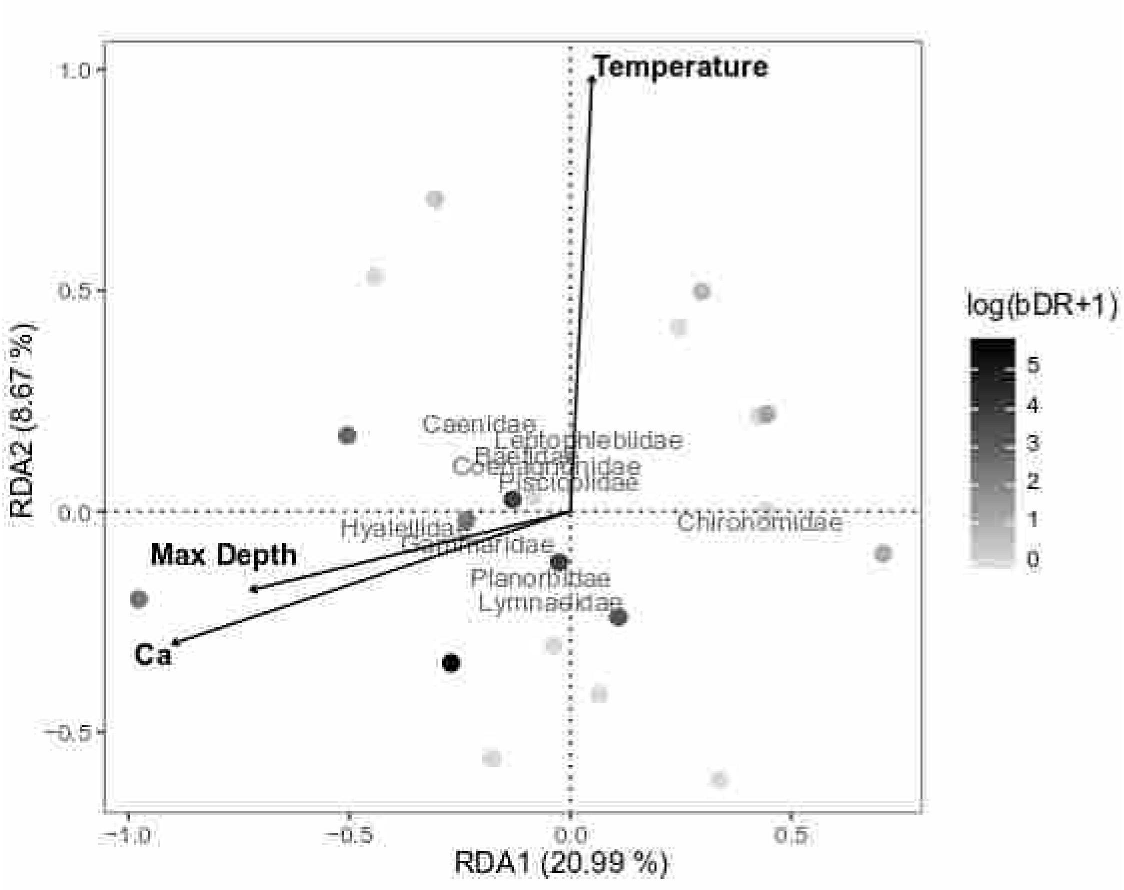

Figure 7: Redundancy analysis (RDA) of benthic invertebrate families found within lakes sampled in the Sahtú Settlement Area (SSA). Benthic macroinvertebrate family labels clustered at the center of the plot were removed for clarity and others were moved slightly to avoid overlap. Darker points represent lakes with a higher log burned drainage ratio +1 (bDR). Calcium = Ca.

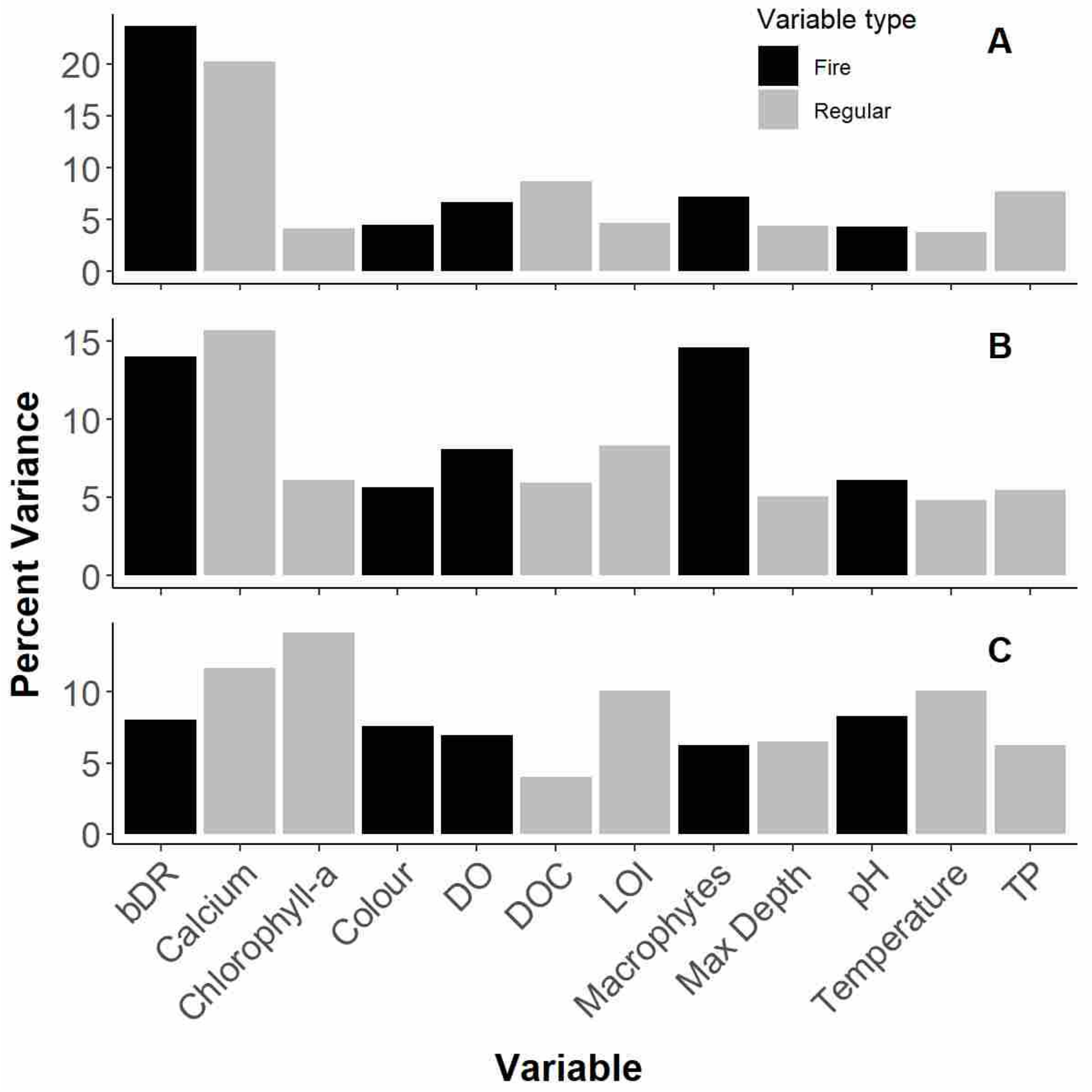

Figure 8*:* Results of hierarchical partitioning to determine variables most strongly associated with differences in benthic macroinvertebrate taxa richness (A), total abundance (B), and EPT abundance (C) among lakes. Variables were identified as “fire” variables if they showed a correlation with bDR (total hectares burned /lake surface area) in Spearman correlations. Dissolved oxygen = DO, dissolved organic carbon = DOC, % organic sediment = LOI, SWIR2 = Macrophytes, and total phosphorous = TP.

References

Allen, E.W., Prepas, E.E., Gabos, S., Strachan, W.M., and Zhang, W. 2005. Methyl mercury concentrations in macroinvertebrates and fish from burned and undisturbed lakes on the Boreal Plain. Can. J. Fish. Aquat. Sci. **62**(9): 1963–1977. doi:10.1139/f05-103.

Angeler, D.G., Allen, C.R., and Johnson, R.K. 2013. Measuring the relative resilience of subarctic lakes to global change: Redundancies of functions within and across temporal scales. J. Appl. Ecol. **50**(3): 572–584. doi:10.1111/1365-2664.12092.

Angeler, D.G., and Goedkoop, W. 2010. Biological responses to liming in boreal lakes: An assessment using plankton, macroinvertebrate and fish communities. J. Appl. Ecol. **47**(2): 478–486. doi:10.1111/j.1365-2664.2010.01794.x.

Bayley, S.E., Schindler, D.W., Beaty, K.G., Parker, B.R., and Stainton, M.P. 1992. Effects of multiple fires on nutrient yields from streams draining boreal forest and fen watersheds: nitrogen and phosphorus. Can. J. Fish. Aquat. Sci. **49**(3): 584–596. doi:10.1139/f92-068.

Bégin, P.N., and Vincent, W.F. 2017. Permafrost thaw lakes and ponds as habitats for abundant rotifer populations. Arct. Sci. **3**(2): 354–377. doi:10.1139/as-2016-0017.

Bergström, A.K., Karlsson, J., Karlsson, D., and Vrede, T. 2018. Contrasting plankton stoichiometry and nutrient regeneration in northern arctic and boreal lakes. Aquat. Sci. **80**(2): 1–14. Springer International Publishing. doi:10.1007/s00027-018-0575-2.

Binzer, T., Sand-Jensen, K., and Middelboe, A.L. 2006. Community photosynthesis of aquatic macrophytes. Limnol. Oceanogr. **51**(6): 2722–2733. doi:10.4319/lo.2006.51.6.2722.

Bond-Lamberty, B., Peckham, S.D., Gower, S.T., and Ewers, B.E. 2009. Effects of fire on regional evapotranspiration in the central Canadian boreal forest. Glob. Chang. Biol. **15**(5): 1242–1254. doi:10.1111/j.1365-2486.2008.01776.x.

Borcard, D., Gilley, F., and Legendre, P. 2011. Numerical ecology with R. Springer, New York, NY. Available from http://public.ebookcentral.proquest.com/choice/publicfullrecord.aspx?p=666583.

Brodersen, K.P., Dall, P.C., and Lindegaard, C. 1998. The fauna in the upper stony littoral of Danish lakes: macroinvertebrates as trophic indicators. Freshw. Biol. **39**(3): 577–592. doi:10.1046/j.1365-2427.1998.00298.x.

Cairns, A., and Yan, N. 2009. A review of the influence of low ambient calcium concentrations on freshwater daphniids, gammarids, and crayfish. Environ. Rev. **17**(NA): 67–79. doi:10.1139/A09-005.

Canadian Forest Service. 2020. Canadian National Fire Database – Agency Fire Data. Available from http://cwfis.cfs.nrcan.gc.ca/en_CA/nfdb.

Carignan, R., D'Arcy, P., and Lamontagne, S. 2000. Comparative impacts of fire and forest harvesting on water quality in Boreal Shield lakes. Can. J. Fish. Aquat. Sci. **57**(SUPPL. 2): 105–117. doi:10.1139/f00-125.

Carpenter, S.R., Gurevitch, A., and Adams, M.S. 1979. Factors causing elevated biological oxygen demand in the littoral zone of lake Wingra, Wisconsin. Hydrobiologia **67**(1): 3–9. doi:10.1007/BF00020871.

CCME. 2011. Protocols Manual for Water Quality Sampling in Canada CCME (2011) 5.1. Can. Counc. Minist. Environ.: 2–3. doi:ISBN 978-1-896997-7-0.

Chevan, A., and Sutherland, M. 1991. Hierarchical partitioning. Am. Stat. **45**(2): 90–96. doi:10.1080/00031305.1991.10475776.

Dalesman, S., and Lukowiak, K. 2010. Effect of acute exposure to low environmental calcium on respiration and locomotion in Lymnaea stagnalis (L.). J. Exp. Biol. **213**(9): 1471–1476. doi:10.1242/jeb.040493.

Dallas, T., and Drake, J.M. 2014. Relative importance of environmental, geographic, and spatial variables on zooplankton metacommunities. Ecosphere **5**(9): 1–13. doi:10.1890/ES14-00071.1.

Flannigan, M., Stocks, B., Turetsky, M., and Wotton, M. 2009. Impacts of climate change on fire activity and fire management in the circumboreal forest. Glob. Chang. Biol. **15**(3): 549–560. doi:10.1111/j.1365-2486.2008.01660.x.

Flood, N. 2013. Seasonal composite landsat TM/ETM+ Images using the medoid (a multi-dimensional median). Remote Sens. **5**(12): 6481–6500. doi:10.3390/rs5126481.

Garcia, E., and Carignan, R. 1999. Impact of wildfire and clear-cutting in the boreal forest on methyl mercury in zooplankton. Can. J. Fish. Aquat. Sci. **56**(2): 339–345. doi:10.1139/f98-164.

Garcia, E., and Carignan, R. 2005. Mercury concentrations in fish from forest harvesting and fire-impacted Canadian Boreal lakes compared using stable isotopes of nitrogen. Environ. Toxicol. Chem. **24**(3): 685–693. doi:10.1897/04-065R.1.

Gasith, A., and Hoyer, M. V. 1998. Structuring role of macrophytes in lakes: changing influence along lake size and depth gradients. *In* Ecological Studies. Springer. pp. 381–392.

doi:10.1007/978-1-4612-0695-8_29.

Gibson, C.M., Chasmer, L.E., Thompson, D.K., Quinton, W.L., Flannigan, M.D., and Olefeldt, D. 2018. Wildfire as a major driver of recent permafrost thaw in boreal peatlands. Nat. Commun. **9**: 1–9. doi:10.1038/s41467-018-05457-1.

Gillooly, J.F. 2000. Effect of body size and temperature on generation time in zooplankton. J. Plankton Res. **22**(2): 241–251. doi:10.1093/plankt/22.2.241.

Golder Associates. 2015. Central Mackenzie surface water and groundwater baseline assessment. Available from https://mvlwb.com/registry/?f[]=region:Sahtu.

Gorelick, N., Hancher, M., Dixon, M., Ilyushchenko, S., Thau, D., and Moore, R. 2017. Google Earth Engine: Planetary-scale geospatial analysis for everyone. Remote Sens. Environ. **202**: 18–27. doi:10.1016/j.rse.2017.06.031.

Gray, D.K., and Arnott, S.E. 2012. The role of dispersal levels, Allee effects and community resistance as zooplankton communities respond to environmental change. J. Appl. Ecol. **49**(6): 1216–1224. doi:10.1111/j.1365-2664.2012.02203.x.

Gray, D.K., Arnott, S.E., Shead, J.A., and Derry, A.M. 2012. The recovery of acid-damaged zooplankton communities in Canadian Lakes: the relative importance of abiotic, biotic and spatial variables. Freshw. Biol. **57**(4): 741–758. doi:10.1111/j.1365-2427.2012.02739.x.

Grosbois, G., del Giorgio, P.A., and Rautio, M. 2017. Zooplankton allochthony is spatially heterogeneous in a boreal lake. Freshw. Biol. **62**(3): 474–490. doi:10.1111/fwb.12879.

Hall, S.R., Pauliukonis, N.K., Mills, E.L., Rudstam, L.G., Schneider, C.P., Lary, S.J., and Arrhenius, F. 2003. A comparison of total phosphorus, chlorophyll a, and zooplankton in embayment, nearshore, and offshore habitats of Lake Ontario. J. Great Lakes Res. **29**(1): 54–69. doi:10.1016/S0380-1330(03)70415-8.

Hanes, C.C., Wang, X., Jain, P., Parisien, M.A., Little, J.M., and Flannigan, M.D. 2019. Fire-regime changes in canada over the last half century. Can. J. For. Res. **49**(3): 256–269. doi:10.1139/cjfr-2018-0293.

Haney, J.F. 2013. An-Image-based Key to the Zooplankton of North America version 5.0. Available from http://cfb.unh.edu/cfbkey/html/.

Heinle, D.R. 1969. Temperature and zooplankton. Chesap. Sci. **10**: 186–209. doi:10.2307/1350456.

Heiri, O., Lotter, A.F., and Lemcke, G. 2001. Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. Entomol. Exp. Appl. **25**: 101–110. doi:10.1023/A.

Hessen, D.O., Faafeng, B.A., Smith, V.H., Bakkestuen, V., and Walseng, B. 2006. Extrinsic and intrinsic controls of zooplankton diversity in lakes. Ecology **87**(2): 433–443. doi:10.1890/05-0352.

Irish, R.R. 2000. Landsat 7 automatic cloud cover assessment. NASA's Goddard Sp. Flight Cent.: 1–8. Available from papers2://publication/uuid/CBAE4EDB-7FBB-483D-89E1-8EF5AB152678.

Jalal, W., Pinel-Alloul, B., and Methot, G. 2005. Mid-term study of the ecological impacts of forest fires and timber harvesting on zooplankton communities in lakes of the boreal ecozone. Rev. Des Sci. L'eau **18**: 221–248.

Jeziorski, A., Paterson, A.M., Watson, I., Cumming, B.F., and Smol, J.P. 2014. The influence of calcium decline and climate change on the cladocerans within low calcium, circumneutral lakes of the Experimental Lakes Area. Hydrobiologia **722**(1): 129–142. doi:10.1007/s10750-013-1691-6.

Jones, F.C., Somers, K.M., Craig, B., and Reynoldson, T.B. 2007. Ontario Benthos Biomonitoring Network: protocol manual.

Jones, S.E., Zwart, J.A., Kelly, P.T., and Solomon, C.T. 2018. Hydrologic setting constrains lake heterotrophy and terrestrial carbon fate. Limnol. Oceanogr. Lett. **3**(3): 256–264. doi:10.1002/lol2.10054.

Karlsson, J., Byström, P., Ask, J., Ask, P., Persson, L., and Jansson, M. 2009. Light limitation of nutrient-poor lake ecosystems. Nature **460**(7254): 506–509. doi:10.1038/nature08179.

Kasischke, E.S., Verbyla, D.L., Rupp, T.S., McGuire, A.D., Murphy, K.A., Jandt, R., Barnes, J.L., Hoy, E.E., Duffy, P.A., Calef, M., and Turetsky, M.R. 2010. Alaska’s changing fire regime — implications for the vulnerability of its boreal forests. Can. J. For. Res. **40**(7): 1313–1324. doi:10.1139/X10-098.

Kassambara, A., and Mundt, F. 2019. factoextra: extract and visualize the results of multivariate data analyses. R package. Available from https://cran.r-project.org/package=factoextra.

Keeley, J.E. 2009. Fire intensity, fire severity and burn severity: A brief review and suggested usage. Int. J. Wildl. Fire **18**(1): 116–126. doi:10.1071/WF07049.

Kelly, E.N., Schindler, D.W., St. Louis, V.L., Donald, D.B., and Vladicka, K.E. 2006. Forest fire increases mercury accumulation by fishes via food web restructuring and increased mercury inputs. Proc. Natl. Acad. Sci. **103**(51): 19380–19385. doi:10.1073/pnas.0609798104.

Khanna, S., Santos, M.J., Ustin, S.L., and Haverkamp, P.J. 2011. An integrated approach to a biophysiologically based classification of floating aquatic macrophytes. Int. J. Remote Sens. **32**(4): 1067–1094. doi:10.1080/01431160903505328.

Korosi, J.B., Mcdonald, J., Coleman, K.A., Palmer, M.J., Smol, J.P., Simpson, M.J., and Blais, J.M. 2015. Long-term changes in organic matter and mercury transport to lakes in the sporadic discontinuous permafrost zone related to peat subsidence. Limnol. Oceanogr. **60**(5): 1550–1561. doi:10.1002/lno.10116.

Labaj, A.L., Jeziorski, A., Kurek, J., and Smol, J.P. 2014. Long-term trends in cladoceran assemblages related to acidification and subsequent liming of Middle Lake (Sudbury, Canada). Water. Air. Soil Pollut. **225**(1868): 1–9. doi:10.1007/s11270-014-1868-2.

Lacoul, P., and Freedman, B. 2006. Environmental influences on aquatic plants in freshwater ecosystems. Environ. Rev. **14**(2): 89–136. doi:10.1139/A06-001.

Lamontagne, S., Carignan, R., D’Arcy, P., Prairie, Y.T., and Pare, D. 2000. Element export in runoff from eastern Canadian Boreal Shield drainage basins following forest harvesting and wildfires. Can. J. Fish. Aquat. Sci. **57**: 118–128. doi:10.1139/f00-108.

Lantz, T.C., and Kokelj, S. V. 2008. Increasing rates of retrogressive thaw slump activity in the Mackenzie Delta region, N.W.T., Canada. Geophys. Res. Lett. **35**(6): 1–5. doi:10.1029/2007GL032433.

Lauridsen, T.L., Pedersen, L.J., Jeppesen, E., and Søndergaard, M. 1996. The importance of macrophyte bed size for cladoceran composition and horizontal migration in a shallow lake.

J. Plankton Res. **18**(12): 2283–2294. doi:10.1093/plankt/18.12.2283.

Legendre, P., and Gallagher, E.D. 2001. Ecologically meaningful transformations for ordination of species data. Oecologia **129**(2): 271–280. doi:10.1007/s004420100716.

Legendre, P., Oksanen, J., and ter Braak, C.J.F. 2011. Testing the significance of canonical axes in redundancy analysis. Methods Ecol. Evol. **2**(3): 269–277. doi:10.1111/j.2041-210X.2010.00078.x.

Lehtovaara, A., Arvola, L., Keskitalo, J., Olin, M., Rask, M., Salonen, K., Sarvala, J., Tulonen, T., and Vuorenmaa, J. 2014. Responses of zooplankton to long-term environmental changes in a small boreal lake. Boreal Environ. Res. **19**: 97–111.

Leppä, M., Hämäläinen, H., and Karjalainen, J. 2003. The response of benthic macroinvertebrates to whole-lake biomanipulation. Hydrobiologia **498**(1): 97–105. doi:10.1023/A:1026224923481.

Lévesque, D., Pinel-Alloul, B., Méthot, G., and Steedman, R. 2017. Effects of climate, limnological features and watershed clearcut logging on long-term variation in zooplankton communities of boreal shield lakes. Water **9**(733): 1–19. doi:10.3390/w9100733.

Lewis, T.L., Lindberg, M.S., Schmutz, J.A., and Bertram, M.R. 2014. Multi-trophic resilience of boreal lake ecosystems to forest fires. **95**(5): 1253–1263.

Mathieu, E. 2018. The effects of fire on snow accumulation , snowmelt and ground thaw on a peat plateau in subarctic Canada.

McDermott, H., Paull, T., and Strachan, S. 2014. CABIN laboratory methods : processing, taxonomy, and quality control of benthic macroinvertebrate samples. Ottawa, ON.

McEachern, P., Prepas, E.E., Gibson, J.J., and Dinsmore, W.P. 2000. Forest fire induced impacts on phosphorus, nitrogen, and chlorophyll a concentrations in boreal subarctic lakes of northern Alberta. Can. J. Fish. Aquat. Sci. **57**(S2): 73–81. doi:10.1139/cjfas-57-S2-73.

McEachern, P., Prepas, E.E., and Planas, D. 2002. Phytoplankton in boreal subarctic lakes following enhanced phosphorus loading from forest fire: Impacts on species richness, nitrogen and light limitation. Lake Reserv. Manag. **18**(2): 138–148. doi:10.1080/07438140209354144.

Menken, K.D., Brezonik, P.L., and Bauer, M.E. 2006. Influence of Chlorophyll and Colored Dissolved Organic Matter (CDOM) on Lake Reflectance Spectra: Implications for Measuring Lake Properties by Remote Sensing. Lake Reserv. Manag. **22**(3): 179–190. doi:10.1080/07438140609353895.

Merritt, R.W., Cummins, K.W., and Berg, M.B. 2008. An introduction to the aquatic insects of North America. *In* 4th edition. Kendall Hunt Pub Co, Dubuque, Iowa.

Mochnacz, N.J., Reist, J.D., and Rt, M. 2007. Biological and habitat data for fish collected during stream surveys in the Sahtu Settlement Region , Northwest Territories , 2006. Can. Data Rep. Fish Aquat. Sci. (1189): 1–40.

Moquin, P.A., Mesquita, P.S., Wrona, F.J., and Prowse, T.D. 2014. Responses of benthic invertebrate communities to shoreline retrogressive thaw slumps in Arctic upland lakes. Freshw. Sci. **33**(4): 1108–1118. doi:10.1086/678700.

Mousavi, S.K. 2002. Boreal chironomid communities and their relations to environmental factors - The impact of lake depth, size and acidity. Boreal Environ. Res. **7**(1): 63–75.

Musetta-Lambert, J., Kreutzweiser, D., and Sibley, P. 2019. Influence of wildfire and harvesting on aquatic and terrestrial invertebrate drift patterns in boreal headwater streams. Hydrobiologia **834**(1): 27–45. Springer International Publishing. doi:10.1007/s10750-019-3907-x.

Mac Nally, R. 2000. Regression and model-building in conservation biology, biogeography and ecology: the distinction between–and reconciliation of–'predictive'and 'explanatory'models. Biodivers. Conserv. **9**(5): 655–671. Springer.

Mac Nally, R. 2002. Multiple regression and inference in ecology and conservation biology: further comments on identifying important predictor variables. Biodivers. Conserv. **11**(8): 1397–1401.

Namayandeh, A., and Quinlan, R. 2011. Benthic macroinvertebrate communities in arctic lakes and ponds of Central Nunavut, Canada. Arctic, Antarct. Alp. Res. **43**(3): 417–428. doi:10.1657/1938-4246-43.3.417.

Norlin, J.I., Bayley, S.E., and Ross, L.C.M. 2006. Zooplankton composition and ecology in western boreal shallow-water wetlands. Hydrobiologia **560**(1): 197–215. doi:10.1007/s10750-005-1185-2.

Oksanen, J. 2017. Vegan: ecological diversity. R Packag. Version 2.4-4 **1**: 11. Available from https://cran.r-project.org/package=vegan.

Oksanen, J., Blanchet, F.G., Kindt, R., Legendre, P., Minchin, P.R., Hara, R.B.O., Simpson, G.L., Solymos, P., Stevens, M.H.H., and Wagner, H. 2015. Package 'vegan.'

Oyama, Y., Matsushita, B., and Fukushima, T. 2015. Distinguishing surface cyanobacterial blooms and aquatic macrophytes using Landsat/TM and ETM+ shortwave infrared bands. Remote Sens. Environ. **157**: 35–47. doi:10.1016/j.rse.2014.04.031.

Parker, D. 2012. Picture guide to the common aquatic “ bugs ” Of Saskatchewan. Available from http://www.aquatax.ca/BugGuide.html.

Patoine, A. 2002. Influence of catchment deforestationby logging and natural forest fires on crustacean community size structure in lakes of the Eastern Boreal Canadian forest. J. Plankton Res. **24**(6): 601–616. doi:10.1093/plankt/24.6.601.

Patoine, A., Pinel-Alloul, B., Prepas, E.E., and Carignan, R. 2000. Do logging and forest fires influence zooplankton biomass in Canadian Boreal Shield Lakes? Can. J. Fish. Aquat. Sci. **57**: 155–164. doi:10.1139/f00-105.

Pomeroy, J.W., Granger, R., Pietroniro, A., Toth, B., and Hedstrom, N. 1999. Classification of the boreal forest for hydrological processes. Ninth Int. Boreal For. Res. Assoc. Conf.: 49–59.

Porter, C., Morin, P., Howat, I., Noh, M.-J., Bates, B., Peterman, K., Keesey, S., Schlenk, M., Gardiner, J., Tomko, K., Willis, M., Kelleher, C., Cloutier, M., Husby, E., Foga, S., Nakamura, H., Platson, M., Wethington Jr., M., Williamson, C., Bauer, G., Enos, J., Arnold, G., Kramer, W., Becker, P., Doshi, A., D’Souza, C., Cummens, P., Laurier, F., Bojesen, M., and Foundation, N.S. 2018. ArcticDEM. Harvard Dataverse. doi:doi/10.7910/DVN/OHHUKH.

Portielje, R., and Van Der Molen, D.T. 1999. Relationships between eutrophication variables: From nutrient loading to transparency. Hydrobiologia **408–409**: 375–387. doi:10.1007/978-

94-017-2986-4_42.

Proctor, H. 2006. Key to aquatic mites known from Alberta. Available from http://www.biology.ualberta.ca/faculty/heather_proctor/uploads/pdfs/AquaticMitesKnown_Alberta Aug_2006.pdf.

Prowse, T.D., Wrona, F.J., Reist, J.D., Gibson, J.J., Hobbie, J.E., Lévesque, L.M.J., and Vincent, W.F. 2006. Climate change effects on hydroecology of arctic freshwater ecosystems. Ambio **35**(7): 347–358. doi:10.1579/0044-7447(2006)35[347:CCEOHO]2.0.CO;2.

R Core Team. 2019. R: A language and environment for statistical computing. Available from http://www.r-project.org/.

Reece, P.F., and Richardson, J.S. 1999. Biomonitoring with the reference condition approach for the detection of aquatic ecosystems at risk. Proc. a Conf. Biol. Manag. species Habitats Risk: 549–552.

dos Reis Oliveira, P.C., Kraak, M.H.S., Pena-Ortiz, M., van der Geest, H.G., and Verdonschot, P.F.M. 2020. Responses of macroinvertebrate communities to land use specific sediment food and habitat characteristics in lowland streams. Sci. Total Environ. **703**: 1–16. The Author(s). doi:10.1016/j.scitotenv.2019.135060.

Reynoldson, T.B., Norris, R.H., Resh, V.H., Day, K.E., and Rosenberg, D.M. 1997. The reference condition: a comparison of multimetric and multivariate approaches to assess water-quality impairment using benthic macroinvertebrates. J. North Am. Benthol. Soc. **16**(4): 833–852. doi:10.2307/1468175.

Reynoldson, T.B., Rosenberg, D.M., and Resh, V.H. 2001. Comparison of models predicting invertebrate assemblages for biomonitoring in the Fraser River catchment, British Columbia. Can. J. Fish. Aquat. Sci. **58**(7): 1395–1410. doi:10.1139/f01-075.

RISC. 2009. The Canadian Aquatic Biomonitoring Network: field manual. Stand. Components Br. Columbia’s Biodivers. (Version 1.0-March 11, 2009): 57. Available from http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.231.5572&rep=rep1&type=pdf.

Robinne, F.-N., Hallema, D.W., Bladon, K.D., and Buttle, J.M. 2020. Wildfire impacts on hydrologic ecosystem services in North American high-latitude forests: A scoping review. J. Hydrol. **581**: 1–33. doi:10.1016/j.jhydrol.2019.124360.

Rotvit, L., and Jacobsen, D. 2014. Egg development of Plecoptera, Ephemeroptera and Odonata along latitudinal gradients. Ecol. Entomol. **39**(2): 177–185. doi:10.1111/een.12085.

Roy, D.P., Wulder, M.A., Loveland, T.R., C.E., W., Allen, R.G., Anderson, M.C., Helder, D., Irons, J.R., Johnson, D.M., Kennedy, R., Scambos, T.A., Schaaf, C.B., Schott, J.R., Sheng, Y., Vermote, E.F., Belward, A.S., Bindschadler, R., Cohen, W.B., Gao, F., Hipple, J.D., Hostert, P., Huntington, J., Justice, C.O., Kilic, A., Kovalskyy, V., Lee, Z.P., Lymburner, L., Masek, J.G., McCorkel, J., Shuai, Y., Trezza, R., Vogelmann, J., Wynne, R.H., and Zhu, Z. 2014. Landsat-8: Science and product vision for terrestrial global change research. Remote Sens. Environ. **145**: 154–172. doi:10.1016/j.rse.2014.02.001.

Schell, V.A., and Kerekes, J.J. 1989. Distribution, abundance and biomass of benthic macroinvertebrates relative to pH and nutrients in eight lakes of Nova Scotia, Canada. Water. Air. Soil Pollut. **46**(1): 359–374. doi:10.1007/BF00192870.

Schindler, D.W., Bayley, S.E., Parker, B.R., Beaty, K.G., Cruikshank, D.R., Fee, E.J., Schindler, E.U., and Stainton, M.P. 1996. The effects of climatic warming on the properties of boreal lakes and streams at the Experimental Lakes Area, northwestern Ontario. Limnol. Oceanogr. **41**(5): 1004–1017. doi:10.4319/lo.1996.41.5.1004.

Schriver, P., Bøgestrand, J., Jeppesen, E., and Søndergaard, M. 1995. Impact of submerged macrophytes on fish-zooplanl phytoplankton interactions: large-scale enclosure experiments in a shallow eutrophic lake. Freshw. Biol. **33**(2): 255–270. doi:10.1111/j.1365-2427.1995.tb01166.x.

Scrimgeour, G.J., Tonn, W.M., Paszkowski, C.A., and Goater, C. 2001. Benthic macroinvertebrate biomass and wildfires: Evidence for enrichment of boreal subarctic lakes. Freshw. Biol. **46**(3): 367–378. doi:10.1046/j.1365-2427.2001.00682.x.

Shuter, B.J., and Ing, K.K. 1997. Factors affecting the production of zooplankton in lakes. Can. J. Fish. Aquat. Sci. **54**(2): 359–377. doi:10.1139/f96-270.

Silins, U., Bladon, K.D., Kelly, E.N., Esch, E., Spence, J.R., Stone, M., Emelko, M.B., Boon, S., Wagner, M.J., Williams, C.H.S., and Tichkowsky, I. 2014. Five-year legacy of wildfire and salvage logging impacts on nutrient runoff and aquatic plant, invertebrate, and fish productivity. Ecohydrology **7**(6): 1508–1523. doi:10.1002/eco.1474.

Spence, D.H.N., and Chrystal, J. 1970. Photosynthesis and zonation of freshwater macrophytes. II. Adaptability of species of deep and shallow water. New Phytol. **69**(1): 217–227. doi:10.1111/j.1469-8137.1970.tb04065.x.

Stansbury, J., Kozimor, L., Admiraal, D., and Dove, E. 2008. Water quality modeling of the effects of macrophytes on dissolved oxygen in a shallow tailwater reservoir. Lake Reserv.

Manag. **24**(4): 339–348. doi:10.1080/07438140809354844.

Stemberger, R.S., Larsen, D.P., and Kincaid, T.M. 2001. Sensitivity of zooplankton for regional lake monitoring. Can. J. Fish. Aquat. Sci. **58**(11): 2222–2232. doi:10.1139/cjfas-58-11-2222.

Stocks, B.J., Mason, J.A., Todd, J.B., Bosch, E.M., Wotton, B.M., Amiro, B.D., Flannigan, M.D., Hirsch, K.G., Logan, K.A., Martell, D.L., and Skinner, W.R. 2003. Large forest fires in Canada, 1959-1997. J. Geophys. Res. D Atmos. **108**(1). doi:10.1029/2001jd000484.

Strom, D., Simpson, S.L., Batley, G.E., and Jolley, D.F. 2011. The influence of sediment particle size and organic carbon on toxicity of copper to benthic invertebrates in oxic/suboxic surface sediments. Environ. Toxicol. Chem. **30**(7): 1599–1610. doi:10.1002/etc.531.

Tabak, L.M., and Gibbs, K.E. 1991. Effects of aluminum, calcium and low pH on egg hatching and nymphal survival of Cloeon triangulifer McDunnough (Ephemeroptera: Baetidae). Hydrobiologia **218**(2): 157–166. doi:10.1007/BF00006788.

Tartarotti, B., Trattner, F., Remias, D., Saul, N., Steinberg, C.E.W., and Sommaruga, R. 2017. Distribution and UV protection strategies of zooplankton in clear and glacier-fed alpine lakes. Sci. Rep. **7**(1): 1–14. doi:10.1038/s41598-017-04836-w.

Thompson, V.F., Marshall, D.L., Reale, J.K., and Dahm, C.N. 2019. The effects of a catastrophic forest fire on the biomass of submerged stream macrophytes. Aquat. Bot. **152**: 36–42. doi:10.1016/j.aquabot.2018.09.001.

Thorp, J.H., and Rogers, C.D. (*Editors*). 2016. Keys to Nearctic Fauna: Thorp and Covich’s Freshwater Invertebrates - Volume II. *In* 4th edition. Academic Press, London. Available from https://www.researchgate.net/profile/Lawrence_Lovell/publication/313429453_Chapter_12_Phylum_Annelida1/links/59c2e86ea6fdcc69b9301287/Chapter-12-Phylum-Annelida1.pdf.

Tian, J., and Philpot, W.D. 2015. Relationship between surface soil water content, evaporation rate, and water absorption band depths in SWIR reflectance spectra. Remote Sens. Environ. **169**: 280–289. doi:10.1016/j.rse.2015.08.007.

U.S. Geological Survey. 2019. Landsat 8 Surface Reflectance Code (LASRC) Poduct Guide. (No. LSDS-1368 Version 2.0). (May): 40. Available from https://www.usgs.gov/media/files/landsat-8-surface-reflectance-code-lasrc-product-guide.

US EPA method 600. 1993. Methods for Determination of Inorganic Substances in Environmental Samples, EPA/600/R-93/100.

Vanhellemont, Q., and Ruddick, K. 2015. Advantages of high quality SWIR bands for ocean colour processing: Examples from Landsat-8. Remote Sens. Environ. **161**: 89–106. Elsevier B.V. doi:10.1016/j.rse.2015.02.007.

Veraverbeke, S., Rogers, B.M., Goulden, M.L., Jandt, R.R., Miller, C.E., Wiggins, E.B., and Randerson, J.T. 2017. Lightning as a major driver of recent large fire years in North American boreal forests. Nat. Clim. Chang. **7**(7): 529–534. doi:10.1038/nclimate3329.

Vermote, E., Justice, C., Claverie, M., and Franch, B. 2016. Preliminary analysis of the performance of the Landsat 8/OLI land surface reflectance product. Remote Sens. Environ. **185**: 46–56. doi:10.1016/j.rse.2016.04.008.

Wagner, M.J., Bladon, K.D., Silins, U., Williams, C.H.S., Martens, A.M., Boon, S., MacDonald, R.J., Stone, M., Emelko, M.B., and Anderson, A. 2014. Catchment-scale stream temperature response to land disturbance by wildfire governed by surface-subsurface energy exchange and atmospheric controls. J. Hydrol. **517**: 328–338. doi:10.1016/j.jhydrol.2014.05.006.

Walsh, C., and Mac Nally, R. 2013. hier.part: hierarchical partitioning.

Wang, L., Qu, J.J., Hao, X., and Zhu, Q. 2008. Sensitivity studies of the moisture effects on MODIS SWIR reflectance and vegetation water indices. Int. J. Remote Sens. **29**(24): 7065–7075. doi:10.1080/01431160802226034.

Wang, M., Son, S., Zhang, Y., and Shi, W. 2013. Remote sensing of water optical property for China's inland lake taihu using the SWIR atmospheric correction with 1640 and 2130 nm bands. IEEE J. Sel. Top. Appl. Earth Obs. Remote Sens. **6**(6): 2505–2516. doi:10.1109/JSTARS.2013.2243820.

Weed, A., Ayres, M., and Hicke, J. 2013. Consequences of climate change for biotic disturbances. Ecol. Monogr. **83**(4): 441–470. doi:10.1890/13-0160.1.

Welton, J., and Clarke, R. 1980. Laboratory studies on the reproduction and growth of the amphipod , Gammarus pulex (L.). J. Anim. Ecol. **49**(2): 581–592.

White, M.S., Xenopoulos, M.A., Metcalfe, R.A., and Somers, K.M. 2011. Water level thresholds of benthic macroinvertebrate richness, structure, and function of boreal lake stony littoral habitats. Can. J. Fish. Aquat. Sci. **68**(10): 1695–1704. doi:10.1139/f2011-094.

Winkler, G., Leclerc, V., Sirois, P., Archambault, P., and Bérubé, P. 2009. Short-term impact of forest harvesting on water quality and zooplankton communities in oligotrophic headwater lakes of the eastern Canadian Boreal shield. Boreal Environ. Res. **14**(2): 323–337.

Wissel, B., and Ramacharan, C.W. 2003. Plasticity of vertical distribution of crustacean zooplankton in lakes with varying levels of water colour. J. Plankton Res. **25**(9): 1047–1057. doi:10.1093/plankt/25.9.1047.

Witty, L. 2004. Practical guide to identifying freshwater crustacean zooplankton. Cooperative Freshwater Ecology Unit, Sudburry, Ontario, Canada.

Yadav, S., Yoneda, M., Tamura, M., Susaki, J., Ishikawa, K., and Yamashiki, Y. 2017. A satellite-based assessment of the distribution and biomass of submerged aquatic vegetation in the optically shallow basin of Lake Biwa. Remote Sens. **9**(966): 1–27. doi:10.3390/rs9090966.

Yue, X., Mickley, L.J., Logan, J.A., Hudman, R.C., Val Martin, M., and Yantosca, R.M. 2015. Impact of 2050 climate change on North American wildfire: consequences for ozone air quality. Atmos. Chem. Phys. Discuss. **15**(9): 13867–13921. doi:10.5194/acpd-15-13867-2015.

Zhang, Q., and Chen, W. 2007. Fire cycle of the Canada’s boreal region and its potential response to global change. J. For. Res. **18**(1): 55–61. doi:10.1007/s11676-007-0010-3.

Chapter 3: Conclusions

In this section, I give a summary of my results, analysis, and interpretation from Chapter 2. I have included an additional analysis that could not be included in my manuscript chapter due to space limitations and/or their speculative nature. I hope that this information will generate ideas for future studies examining the effects of wildfires in boreal regions. I then discuss how these results contribute to the field and how this research is integrative.

Summary

For my thesis, I investigated how wildfires within the water catchments of boreal sub-arctic lakes **(see Appendix Figure S4-S21)** might affect water quality, zooplankton, and benthic macroinvertebrates. My analysis suggested that impacts from the fires can still be detected, even 4-5 years after wildfires. My study also showed that the area burned within a watershed divided by lake surface area (burned drainage ratio or bDR) was a more useful indicator of impact than categorizing lakes into classes of burned versus reference, or simply totalling up hectares burned within a watershed. I was able to detect negative correlations of bDR with pH and dissolved oxygen (DO), and positive correlations of bDR with water colour, phenols, and SWIR reflectance, which is known to correlate with macrophyte biomass. Changes in remotely sensed SWIR reflectance was confirmed using time-series data, which showed that SWIR significantly diverged in burned lakes following the 2014 fires. This divergence in SWIR reflectance likely indicated changes to macrophyte biomass and had a significant effect on both the benthic macroinvertebrates and zooplankton communities. Redundancy analysis showed that SWIR reflectance was an important predictor of differences in macroinvertebrate communities among lakes; lakes with higher SWIR reflectance tended to have more amphipods in the family Gammaridae. Hierarchical partitioning results showed that the SWIR reflectance was an important predictor variable for benthic macroinvertebrate abundance and zooplankton richness. The hierarchical partitioning results also suggested that bDR was an important predictor variable for zooplankton richness, and benthic macroinvertebrate richness and abundance.

While my study did detect effects of wildfire on water quality, zooplankton communities, and benthic communities within boreal subarctic lake ecosystems, the overall picture from my results is one of resilience. Resilience to pulse disturbances in ecological systems is a well-studied phenomenon (Hillebrand and Kunze 2020), however, research is still being conducted to determine what makes an ecosystem stable in the face of perturbations. A major dilemma is that very few studies measure the ecological stability after disturbance through multiple components, with most considering only one facet and the type of disturbance (Donohue et al. 2016; Kéfi et al. 2019). A pulse disturbance, such as wildfire, is expected to have a multifaceted effect on lake ecosystems. A multidimensional quantification for determining the stability of ecosystems is therefore preferential. When few and weak correlations occur between the disturbance and variables impacted, different processes may be responsible for the stabilization of the ecosystem (Donohue et al. 2013). Many strong correlations imply that the stability of the ecosystem is weak (Donohue et al. 2013). When responding to a pulse disturbance, there may be immediate resistance (no change), resilience (approaching reference conditions), recovery (returning to reference conditions after a degree of time) and temporal invariability of the recovery trend (Donohue et al. 2013; Hillebrand and Kunze 2020). In my study, most of the variation in water quality and invertebrate communities seemed to be related to natural variation among lakes, rather than the impacts of the fires. Most of the variation occurred between two categories of lakes; clear, bigger lakes versus smaller, coloured lakes. Correlations with the disturbance were weak, and richness, diversity, and relative abundance of lakes impacted by fires returned to pre-fire conditions or in fact never changed. It is therefore highly plausible that my study lakes had a significant degree of resilience to wildfires.

In addition to observing the effects of wildfire on water quality, zooplankton, and benthic macroinvertebrate communities, I examined the fate of methylmercury in water and invertebrates in my study lakes (MeHg; **see Appendix S2** for Methodology). Very weak correlations existed between bDR and surface water MeHg and sediment MeHg, with no significant results indicating an increase in MeHg in water, sediments, or invertebrates in lakes affected by wildfire (**see Appendix S3** for Results; **see Appendix Figure S3**). These results were contrary to my expectations, as many studies have suggested increased availability of MeHg following fires (Kelly et al. 2006; Witt et al. 2009; Rothenberg et al. 2010). Yet other counterexample studies did not find an increase in MeHg after wildfire (Garcia and Carignan 1999; Allen et al. 2005). This was partially attributed to increased primary production (Garcia and Carignan 1999; Allen et al. 2005). If more MeHg was available in our studied lakes, the question becomes, where did it go (see Appendix S4 for Discussion)? New research has been conducted suggesting that macrophytes can play an important role in mercury methylation at the roots embedded in the sediment in shallow lakes, followed by MeHg uptake (Cosio et al. 2014). These aquatic plants may conversely be an alternative entry pathway for MeHg to entering aquatic food webs (Cosio et al. 2014; Bravo et al. 2014; Beauvais-Flück et al. 2017). The process of phytoremediation of aquatic ecosystems has been widely proposed by using macrophytes as a buffer against MeHg in aquatic ecosystems (Figueira et al. 2012; Chattopadhyay et al. 2012; Beauvais-Flück et al. 2017).

Macrophytes may also be an important actor determining the resilience of boreal subarctic lakes to nutrient increases following wildfires. I saw no differences in nutrients between lakes exposed to burns and those that were not. However, I did see a divergence in

SWIR reflectance between reference and burned lakes associated with vegetation in lakes affected by burns. Aquatic vegetation is well known to remove excess nitrogen and phosphorous and improve water quality (e.g. Lu et al. 2010; Fu and He 2015; Yu et al. 2019). For example, reduced turbidity and phytoplankton in aquatic ecosystems is often associated with increased macrophyte biomass (Scheffer 1999). In our samples, lakes with higher turbidity contained higher macrophytes. Will we see future changes to reduced turbidity as a response to increased macrophyte biomass? Contrary to past studies on boreal wildfires that saw increases in algae production (Garcia and Carignan 1999, 2005; McEachern et al. 2000, 2002; Kelly et al. 2006; Silins et al. 2014), I did not see an increase in chlorophyll-*a*. Instead, might these nutrients have been taken up by macrophytes? Other studies conducted within the boreal have also detected no increase in nitrogen, phosphorous, or chlorophyll-*a* after a wildfire (Lewis et al. 2014), but again, they did not consider macrophytes as a potential sink for those nutrients.

The divergence in macrophytes in lakes affected by burns may relate to the concept of alternative stable states in shallow lakes. For example, shallow lakes are known to shift between a turbid, phytoplankton dominated state and one that has clear water and is dominated by macrophytes (Scheffer 1999; Hilt 2015; Hilt et al. 2018). Is it possible that shallow lakes within the boreal subarctic could undergo regime shifts after nutrient loading from wildfires, resulting in increased macrophytes and clearer water (Scheffer and Van Nes 2007)? Future studies should consider the role that macrophytes might play as lakes respond to the effects of wildfire.

So, what does the increasing fire frequency in the boreal forest mean for lakes? My study suggests that the effects of fires are still detectable after 4-5 years of recovery, and that important components of habitat structure in lakes (i.e. the macrophytes) may change significantly following fires. On average, fires return every 180 years in the Canadian boreal forest, but have a significantly higher return rate of 53 years in the Russian boreal forest as a response to more human ignitions (de Groot et al. 2013). Will the shortening fire cycle time to less than a century in parts of the boreal forest (Zhang and Chen 2007) mean that lakes are never allowed to be within reference conditions before the next disturbance? Unfortunately, I do not know how long the changes I detected will persist, and it is difficult to predict how the fire return interval will change in the future. There are however accounts detecting changes to macroinvertebrate communities persist for 15-20 years after wildfire in the boreal forest (Scrimgeour et al. 2001; Musetta-Lambert et al. 2019). Nevertheless, the combination of long-term impacts of wildfires combined with increasing fire frequency is something that ecologists and resource managers should consider when evaluating the future of lakes in the boreal forest.

Contributions to the Field

Although wildfires are a natural phenomenon in the boreal forest, the frequency of wildfires is increasing in response to climate change. Unfortunately, there is a significant knowledge gap regarding the impacts of wildfires on hydrology, water quality, and biology in subarctic and Arctic regions (Robinne et al. 2020). My study helps to further our knowledge of how water quality and biology of boreal lakes are affected by wildfires. This study is one of few that has been conducted in the northern boreal forest, and it shows that the effects of wildfires are still detectable 4-5 years after the disturbance in my study region. However, my work also shows that much of the variation in invertebrate communities was related to natural environmental variation in physicochemical variables, and therefore supports the conclusions of previous studies that have suggested lakes in the northern boreal forest are resilient to changes caused by wildfires (Jalal et al. 2005; Lewis et al. 2014).

Some of the changes I detected in my study were also attributed to wildfires in past studies. pH was negatively correlated with wildfires (Garcia and Carignan 1999; McEachern et al. 2000), while colour increased, potentially affecting light attenuation with depth (Carignan et al. 2000; McEachern et al. 2000; Scrimgeour et al. 2001). These other studies concluded that much of the response was likely caused by humic input, and my study helps to confirm this assumption by showing that phenolic compounds and colour were higher in lakes affected by wildfires. Finally, incorporating zooplankton into a study on wildfire impacts is rare. I was only able to find two studies conducted in the boreal forest examining the response of zooplankton to wildfire (Patoine et al. 2000; Jalal et al. 2005). Both were conducted on the same location and only looked at the biomass of zooplankton at order taxon or higher. My study shows that the degree of impact by wildfires (bDR) can be an important predictor for zooplankton richness, which has not been shown previously.

How this Research is Integrative

Integrative research requires a study to harness multiple different disciplines into one coherent project. As arguably the most impactful disturbance to the boreal forest, wildfires cause large changes in the terrestrial landscape that affect the hydrology, chemistry, and biology of lake ecosystems. As a result, the very nature of studying aquatic ecology as a response to wildfire needs to be multidimensional and integrate numerous disciplines into the study. In my project, I incorporated data and analyses from hydrology, terrestrial biology, geology, and remote sensing to assess how lakes are affected by wildfire. The very nature of the universe, down to quantum physics, will affect the ecology and has even stemmed a new discipline called quantum ecology on how energy flows from one variable to the next (Orlóci 2015). Finally, communicating all this integrative research requires completely different disciplines in art (production of figures) and writing skills.

References

Ackley, C.J. 2019. Biogeochemical and hydrological impacts of a low-severity wildfire in the wetland-dominated zone of discontinuous permafrost. : 78. Available from https://scholars.wlu.ca/etd/2141/.

Allen, E.W., Prepas, E.E., Gabos, S., Strachan, W.M., and Zhang, W. 2005. Methyl mercury concentrations in macroinvertebrates and fish from burned and undisturbed lakes on the Boreal Plain. Can. J. Fish. Aquat. Sci. **62**(9): 1963–1977. doi:10.1139/f05-103.

Beauvais-Flück, R., Gimbert, F., Méhault, O., and Cosio, C. 2017. Trophic fate of inorganic and methyl-mercury in a macrophyte-chironomid food chain. J. Hazard. Mater. **338**: 140–147. Elsevier B.V. doi:10.1016/j.jhazmat.2017.05.028.

Bigham, G.N., Murray, K.J., Masue-slowey, Y., and Henry, E.A. 2017. Biogeochemical controls on methylmercury in soils and sediments : Implications for site management. **13**(2): 249–263. doi:10.1002/ieam.1822.

Bravo, A.G., Cosio, C., Amouroux, D., Zopfi, J., Chevalley, P.-A., Spangenberg, J.E., Ungureanu, V.-G., and Dominik, J. 2014. Extremely elevated methyl mercury levels in water, sediment and organisms in a Romanian reservoir affected by release of mercury from a chlor-alkali plant. Water Res. **49**: 391–405. doi:10.1016/j.watres.2013.10.024.

Carignan, R., D’Arcy, P., and Lamontagne, S. 2000. Comparative impacts of fire and forest harvesting on water quality in Boreal Shield lakes. Can. J. Fish. Aquat. Sci. **57**(SUPPL. 2):

105–117. doi:10.1139/f00-125.

Chattopadhyay, S., Fimmen, R.L., Yates, B.J., Lal, V., and Randall, P. 2012. Phytoremediation of mercury- and methyl mercury-contaminated sediments by Water Hyacinth (Eichhornia crassipes). Int. J. Phytoremediation **14**(2): 142–161. doi:10.1080/15226514.2010.525557.

Cosio, C., Flück, R., Regier, N., and Slaveykova, V.I. 2014. Effects of macrophytes on the fate of mercury in aquatic systems. Environ. Toxicol. Chem. **33**(6): 1225–1237. doi:10.1002/etc.2499.

Donohue, I., Hillebrand, H., Montoya, J.M., Petchey, O.L., Pimm, S.L., Fowler, M.S., Healy, K., Jackson, A.L., Lurgi, M., McClean, D., O'Connor, N.E., O'Gorman, E.J., and Yang, Q. 2016. Navigating the complexity of ecological stability. Ecol. Lett. **19**(9): 1172–1185. doi:10.1111/ele.12648.

Donohue, I., Petchey, O.L., Montoya, J.M., Jackson, A.L., Mcnally, L., Viana, M., Healy, K., Lurgi, M., O'Connor, N.E., and Emmerson, M.C. 2013. On the dimensionality of ecological stability. Ecol. Lett. **16**(4): 421–429. doi:10.1111/ele.12086.

Figueira, E., Freitas, R., Pereira, E., and Duarte, A. 2012. Mercury uptake and allocation in Juncus maritimus: Implications for phytoremediation and restoration of a mercury contaminated salt marsh. J. Environ. Monit. **14**(8): 2181–2188. doi:10.1039/c2em30076a.

Fu, X., and He, X. 2015. Nitrogen and phosphorus removal from contaminated water by five aquatic plants. *In* Proceedings of the 2015 International Conference on Mechatronics, Electronic, Industrial and Control Engineering. Atlantis Press, Paris, France. pp. 1274–1277. doi:10.2991/meic-15.2015.290.

Garcia, E., and Carignan, R. 1999. Impact of wildfire and clear-cutting in the boreal forest on methyl mercury in zooplankton. Can. J. Fish. Aquat. Sci. **56**(2): 339–345. doi:10.1139/f98-164.

Garcia, E., and Carignan, R. 2005. Mercury concentrations in fish from forest harvesting and fire-impacted Canadian Boreal lakes compared using stable isotopes of nitrogen. Environ. Toxicol. Chem. **24**(3): 685–693. doi:10.1897/04-065R.1.

Giesler, R., Clemmensen, K.E., Wardle, D.A., Klaminder, J., and Bindler, R. 2017. Boreal forests sequester large amounts of mercury over millennial time scales in the absence of wildfire. Environ. Sci. Technol. **51**(5): 2621–2627. doi:10.1021/acs.est.6b06369.

de Groot, W.J., Cantin, A.S., Flannigan, M.D., Soja, A.J., Gowman, L.M., and Newbery, A. 2013. A comparison of Canadian and Russian boreal forest fire regimes. For. Ecol. Manage. **294**: 23–34. doi:10.1016/j.foreco.2012.07.033.

Herrero Ortega, S., Catalán, N., Björn, E., Gröntoft, H., Hilmarsson, T.G., Bertilsson, S., Wu, P., Bishop, K., Levanoni, O., and Bravo, A.G. 2018. High methylmercury formation in ponds fueled by fresh humic and algal derived organic matter. Limnol. Oceanogr. **63**: S44–S53. doi:10.1002/lno.10722.

Hillebrand, H., and Kunze, C. 2020. Meta-analysis on pulse disturbances reveals differences in functional and compositional recovery across ecosystems. Ecol. Lett. **23**(3): 575–585. doi:10.1111/ele.13457.

Hilt, S. 2015. Regime shifts between macrophytes and phytoplankton-concepts beyond shallow lakes, unravelling stabilizing mechanisms and practical consequences. Limnetica **34**(2):

467–480. doi:10.23818/limn.34.35.

Hilt, S., Nuñez, M.M.A., Bakker, E.S., Blindow, I., Davidson, T.A., Gillefalk, M., Hansson, L., Janse, J.H., Janssen, A.B.G., Jeppesen, E., Kabus, T., Kelly, A., Köhler, J., and Lester, R. 2018. Response of submerged macrophyte communities to external and internal restoration measures in north temperate shallow lakes. **9**(February). doi:10.3389/fpls.2018.00194.

Jalal, W., Pinel-Alloul, B., and Methot, G. 2005. Mid-term study of the ecological impacts of forest fires and timber harvesting on zooplankton communities in lakes of the boreal ecozone. Rev. Des Sci. L'eau **18**: 221–248.

Kampalath, R.A., Lin, C.C., and Jay, J.A. 2013. Influences of zero-valent sulfur on mercury methylation in bacterial cocultures. Water. Air. Soil Pollut. **224**(2). doi:10.1007/s11270-012-1399-7.

Kéfi, S., Domínguez-García, V., Donohue, I., Fontaine, C., Thébault, E., and Dakos, V. 2019. Advancing our understanding of ecological stability. Ecol. Lett. **22**(9): 1349–1356. doi:10.1111/ele.13340.

Kelly, E.N., Schindler, D.W., St. Louis, V.L., Donald, D.B., and Vladicka, K.E. 2006. Forest fire increases mercury accumulation by fishes via food web restructuring and increased mercury inputs. Proc. Natl. Acad. Sci. **103**(51): 19380–19385. doi:10.1073/pnas.0609798104.

Korosi, J.B., Mcdonald, J., Coleman, K.A., Palmer, M.J., Smol, J.P., Simpson, M.J., and Blais, J.M. 2015. Long-term changes in organic matter and mercury transport to lakes in the sporadic discontinuous permafrost zone related to peat subsidence. Limnol. Oceanogr. **60**(5): 1550–1561. doi:10.1002/lno.10116.

Lewis, T.L., Lindberg, M.S., Schmutz, J.A., and Bertram, M.R. 2014. Multi-trophic resilience of boreal lake ecosystems to forest fires. **95**(5): 1253–1263.

Lu, Q., He, Z.L., Graetz, D.A., Stoffella, P.J., and Yang, X. 2010. Phytoremediation to remove nutrients and improve eutrophic stormwaters using water lettuce (Pistia stratiotes L.). Environ. Sci. Pollut. Res. **17**(1): 84–96. doi:10.1007/s11356-008-0094-0.

McEachern, P., Prepas, E.E., Gibson, J.J., and Dinsmore, W.P. 2000. Forest fire induced impacts on phosphorus, nitrogen, and chlorophyll a concentrations in boreal subarctic lakes of northern Alberta. Can. J. Fish. Aquat. Sci. **57**(S2): 73–81. doi:10.1139/cjfas-57-S2-73.

McEachern, P., Prepas, E.E., and Planas, D. 2002. Phytoplankton in boreal subarctic lakes following enhanced phosphorus loading from forest fire: Impacts on species richness, nitrogen and light limitation. Lake Reserv. Manag. **18**(2): 138–148. doi:10.1080/07438140209354144.

Mierle, G., and Ingram, R. 1991. The role of humic substances in the mobilization of mercury from watersheds. Water, Air, Soil Pollut. **56**(1): 349–357. doi:10.1007/BF00342282.

Musetta-Lambert, J., Kreutzweiser, D., and Sibley, P. 2019. Influence of wildfire and harvesting on aquatic and terrestrial invertebrate drift patterns in boreal headwater streams. Hydrobiologia **834**(1): 27–45. Springer International Publishing. doi:10.1007/s10750-019-3907-x.

Orlóci, L. 2015. Quantum Ecology. Energy structure and its analysis (2nd enlarged edition). doi:10.13140/2.1.1572.5124.

Paranjape, A.R., and Hall, B.D. 2017. Recent advances in the study of mercury methylation in aquatic systems. Facets **2**(1): 85–119. doi:10.1139/facets-2016-0027.

Patoine, A., Pinel-Alloul, B., Prepas, E.E., and Carignan, R. 2000. Do logging and forest fires influence zooplankton biomass in Canadian Boreal Shield Lakes? Can. J. Fish. Aquat. Sci. **57**: 155–164. doi:10.1139/f00-105.

Pickhardt, P.C., Folt, C.L., Chen, C.Y., Klaue, B., and Blum, J.D. 2002. Algal blooms reduce the uptake of toxic methylmercury in freshwater food webs. Proc. Natl. Acad. Sci. **99**(7): 4419–4423. doi:10.1073/pnas.072531099.

Robinne, F.-N., Hallema, D.W., Bladon, K.D., and Buttle, J.M. 2020. Wildfire impacts on hydrologic ecosystem services in North American high-latitude forests: A scoping review. J. Hydrol. **581**: 1–33. doi:10.1016/j.jhydrol.2019.124360.

Rothenberg, S.E., Kirby, M.E., Bird, B.W., DeRose, M.B., Lin, C.C., Feng, X., Ambrose, R.F., and Jay, J.A. 2010. The impact of over 100 years of wildfires on mercury levels and accumulation rates in two lakes in southern California, USA. Environ. Earth Sci. **60**(5): 993–1005. doi:10.1007/s12665-009-0238-7.

Scheffer, M. 1999. The effect of aquatic vegetation on turbidity; how important are the filter feeders? Hydrobiologia **408**–**409**: 307–316. doi:10.1007/978-94-017-2986-4_34.

Scheffer, M., and Van Nes, E.H. 2007. Shallow lakes theory revisited: Various alternative regimes driven by climate, nutrients, depth and lake size. Hydrobiologia **584**(1): 455–466. doi:10.1007/s10750-007-0616-7.

Scrimgeour, G.J., Tonn, W.M., Paszkowski, C.A., and Goater, C. 2001. Benthic macroinvertebrate biomass and wildfires: Evidence for enrichment of boreal subarctic lakes.

Freshw. Biol. **46**(3): 367–378. doi:10.1046/j.1365-2427.2001.00682.x.

Silins, U., Bladon, K.D., Kelly, E.N., Esch, E., Spence, J.R., Stone, M., Emelko, M.B., Boon, S., Wagner, M.J., Williams, C.H.S., and Tichkowsky, I. 2014. Five-year legacy of wildfire and salvage logging impacts on nutrient runoff and aquatic plant, invertebrate, and fish productivity. Ecohydrology **7**(6): 1508–1523. doi:10.1002/eco.1474.

U.S. Environmental Protection Agency. 1998. Method 1630, methyl mercury in water by distillation, aqueous ethylation, purge and trap, and cold-vapor atomic fluorescence spectrometry; EPA-821-R-01-020. (4303): 1–55.

Witt, E.L., Kolka, R.K., Nater, E.A., and Wickman, T.R. 2009. Forest fire effects on mercury deposition in the boreal forest. Environ. Sci. Technol. **43**(6): 1776–1782. doi:10.1021/es802634y.

Yu, S., Miao, C., Song, H., Huang, Y., Chen, W., and He, X. 2019. Efficiency of nitrogen and phosphorus removal by six macrophytes from eutrophic water. Int. J. Phytoremediation **21**(7): 643–651. doi:10.1080/15226514.2018.1556582.

Zhang, Q., and Chen, W. 2007. Fire cycle of the Canada’s boreal region and its potential response to global change. J. For. Res. **18**(1): 55–61. doi:10.1007/s11676-007-0010-3.

Appendix

Appendix S1. Remote-sensing for macrophytes and estimating burn area

To estimate aquatic macrophyte cover for our study lakes we used remote-sensing data. Landsat 8's Operational Land Imager (OLI) sensor is capable of imaging the earth at 30 m spatial resolution in the visible and near-infrared bands with a return period of 16 days (Roy et al. 2014). The Landsat 8 (L8) archive was screened to identify Collection 1 Surface Reflectance scenes (Vermote et al. 2016) collected over the study area during the growing season, which was defined as June, July, and August for each year of L8's operation. Growing season images were identified and manually observed for each year of L8 operation (2013 - 2019). Images were quality-controlled for clouds, cloud shadows, snow and ice using the pixel Quality Assessment (QA) band (U.S. Geological Survey 2019). We calculated the median band values for each lake during the growing season to generate a seasonal composite medoid time-series across the years of operation of Landsat 8 (Flood 2013).

Using higher reflectance values of short-wave infrared (SWIR) reflectance imaging is highly effective at detecting macrophytes (Khanna et al. 2011; Oyama et al. 2015; Yadav et al. 2017). For our analysis, we used a combination to detect macrophytes of the Landsat 8 band 6 and band 7 with SWIR readings at a wavelength of 1.57-1.65 μm (SWIR1), and 2.11-2.29 μm (SWIR2) at 30 m resolution (Roy et al. 2014). All analysis was conducted using Google Earth Engine, a cloud-based geospatial analysis platform (Gorelick et al. 2017), and results were exported in .csv format for statistical analysis in R (R Core Team 2019).

A long term time-series of macrophytes was combined with the addition of Landsat 7, which was launched in 1999 at a similar 30 m resolution, using band 5 as the SWIR1 value the wavelength of 1.55-1.75 μm, and band 7 as the SWIR2 value at the wavelength of 2.09-2.35 μm (Irish 2000). Mean values of lakes with > 68 % of the WC burned were compared with reference lakes with < 16 % of the WC burned. The mean values pre-fire (< 2014) were then regressed against the mean values post-fire (> 2014) to detect differences between pre and post-fire.

To estimate the burned area in the catchment of each lake, we used Normalized Burn Ratios, which is an index calculated through remote sensing that indicates burned area and severity (Keeley 2009). Normalized burn ratios (NBR) within the WC were calculated through Google Earth Engine from bands generated from the Landsat 8 satellite imagery with the equation:

$$NBR = \frac{(NIR - SWIR)}{(NIR + SWIR)}$$

Differences in NBR (dNBR) were then calculated by subtracting the pre-fire NBR (compiled between 10/07/2012 to 10/09/2013) from post-fire NBR (compiled between 15/08/2014 to 10/09/2014). dNBR ranges were then exported to .csv files and analysed in R (R Core Team 2019).

Appendix S2. Methodology of methylmercury, carbon 13 ($\delta^{13}C$) analysis

Predator odonates were separated from other benthic macroinvertebrates and washed according to EPA Protocol 1630 (U.S. Environmental Protection Agency 1998) with Milli-Q. Surface sediment samples were collected at each of the three stations per lake and kept frozen (-80°C) for < 1 year. I also collected a sample of bulk seston and littoral zooplankton at standard depth (~1 m; ~5 m oblique tow; 64 µm mesh net) at each of three randomly selected locations on each lake. The bulk seston zooplankton sample was preserved frozen (-80 °C) in a Nalgene bottle (125 mL; 24hr HCl acid-washed) for isotopic and MeHg analysis. I then dried these samples using a freeze drier (-60°C) for > 3 days.

To measure MeHg content of the lake water, I collected an integrated sample of water (250 mL; filtered with a 0.45µm Whatman 934-AH glass microfiber filter) with a polyethylene tube sampler (~2 m depth) at the center of the lake then poured it into a borosilicate glass container (500 mL; 24hr HCl acid-washed). I preserved the sample with trace metal grade HCl (1.5 mL) and refrigerated (4°C). I then sent the water samples, freeze-dried sediment (0.5g per station), predator odonates, and bulk seston of zooplankton to the Biotron analytical lab at Western University for measurements of MeHg. At the Biotron analytical lab, MeHg was measured using a Cold Vapour Atomic Fluorescence-Digestion, with a modified protocol from EPA 1630 (Lab method ID -TM.0812; U.S. Environmental Protection Agency 1998). All equipment was bathed in 10% HCl acid (24 hr) between subsamples and washed with Milli-q water. I only sent samples collected in 2018 for MeHg analysis due to cost issues (N = 16 lakes).

To explore whether allochthonous input of carbon and nitrogen affected biota, samples of the predator odonates, bulk seston with zooplankton, and sediment were quantified for $\delta^{13}C$ isotopes at the University of Waterloo, Canada (4010 Elemental Analyzer coupled to a Delta plus XL continuous flow isotope ratio mass spectrometer; CFIRMS). Due to cost, I only sent samples collected in 2018 for analysis of $\delta^{13}C$ (n=16 lakes). Before isotope analysis, I removed inorganic carbonates from sediment samples using a 10% HCl solution. For analysis, I calculated the mean sediment $\delta^{13}C$ concentrations over all three sampling stations on each lake.

Appendix S3. Results of methylmercury, carbon 13 ($\delta^{13}C$) analysis

My results showed increased allochthonous carbon use in the food web in relation to burn impacts: Hectares burned/lake surface area (bDR) was negatively correlated with $\delta^{13}C$ in sediment (r_s = -0.34) and predator odonates (r_s = -0.55; **see Appendix Figure S3**). Surface water

MeHg concentrations were very weakly correlated with bDR (r_s = -0.04) but showed a stronger relationship with variables affected by fire, including colour (r_s = 0.82), pH (r_s = -0.44), and DO (r_s = -0.15). Correlations were also observed with DOC and surface water MeHg (r_s = 0.73), and the bulk seston of zooplankton (r_s = 0.34). The sediment MeHg concentrations were also correlated with bDR (r_s = 0.10) and pH (r_s = -0.36; **see Appendix Figure S3**). Correlations existed between TOC and the surface water MeHg (r_s = 0.78) and sediment MeHg concentrations (r_s = 0.22). Although $\delta^{13}C$ in predator odanates were not observed to be correlated with bDR, the total hectares burned in the water catchment showed positive correlations (r_s = 0.34). Positive correlations between macrophyte readings of SWIR1 and sediment MeHg concentrations (r_s = 0.31) and surface water MeHg concentrations (r_s = 0.41; **see Appendix Figure S3**) were also present. SWIR2 correlations with sediment MeHg concentrations (r_s = 0.37) and surface water MeHg concentrations (r_s = 0.44) indicate positive relationships.

Appendix S4. Discussion of methylmercury, carbon 13 ($\delta^{13}C$) analysis

The volatility of lignin-derived phenolic terrestrial organic carbon after wildfire into lakes combined with lower pH, DO and light may have lead to increases in MeHg (Korosi et al. 2015; Bigham et al. 2017; Giesler et al. 2017; Paranjape and Hall 2017). Increases in MeHg after wildfire in the boreal forest have been directly correlated with DOC, pH, DO, and lake water colour (Garcia and Carignan 1999, 2005). Extensive research has correlated lower DO, pH, and light attenuation with increase methylation rates of anaerobic bacteria in aquatic systems, particularly with high concentrations of organic carbon (Kampalath et al. 2013; Bigham et al. 2017; Paranjape and Hall 2017). The intricate complexities around methylation of mercury are largely unknown, but there is evidence that humic substance is an excellent source of Hg into water (Mierle and Ingram 1991; Herrero Ortega et al. 2018). The decomposing humic peat surrounding the lakes can be high in amino acids and lignin-rich phenols which transport Hg (Korosi et al. 2015). Very strong correlations with colour and surface water MeHg would support this conclusion (**see Appendix S3** for Results; **see Appendix Figure S3**).

By all accounts, there should be an increase in MeHg, however, there are very weak correlations that exists **(see Appendix S3** for Results; **see Appendix Figure S3)**. Porewater from peat soils in the subarctic, when exposed to wildfire, may have higher concentrations of MeHg (Ackley 2019). However, one study reported no change in MeHg concentrations in benthic macroinvertebrates after a wildfire, indicating that the porosity of the soil may have helped reduce the runoff of mercury into the surrounding water bodies (Allen et al. 2005). As such, most potential mercury runoff might never have been able to enter the ecosystem to begin with (Allen et al. 2005). This explanation was primarily speculation on the part of Allen et al. (2005), so a degree of skepticism should be considered. Another possibility is that biodilution spread the MeHg across more algae due to increased primary production, reducing concentrations in individual algal cells, and thereby reducing bioaccumulation in biota at higher trophic levels (Pickhardt et al. 2002). After the wildfire, Allen et al. (2005), concluded that most of the MeHg was accounted for from biodilution effects in conjunction with negative correlations with chlorophyll-*a*. However, no correlations existed between chlorophyll-*a* and MeHg concentrations in my data set **(see Appendix** Error! Reference source not found.**3)**. With positive correlations between macrophyte readings of SWIR1 and SWIR 2 and sediment MeHg concentrations and surface water MeHg concentrations **(see Appendix Figure S3)**, there is a possibility that the macrophytes diluted the MeHg concentrations. Macrophytes are known to uptake MeHg from their roots (Cosio et al. 2014) and reduce MeHg toxicants in aquatic ecosystems through phytoremediation (Figueira et al. 2012; Chattopadhyay et al. 2012; Beauvais-Flück et al. 2017).

Table S1: Transformations applied to variables before running the Principal Component Analysis (PCA) and Redundancy Analysis (RDA).

Parameter	**Transformation**
calcium, chlorophyll-a, conductivity, dissolved organic carbon, SWIR1, maximum depth, pH, surface area, total organic carbon, and water catchment	log
burned drainage ratio, and nitrogen: phosphorous ratio	$log(x + 1)$
Ammonium, iron, nitrite, and turbidity	$log(x + 0.01)$
Alkalinity, total nitrogen, colour, % organic sediment, phenols, temperature, total phosphorous, elevation, SWIR2	*No transformation applied*

Table S2: Variables included for possible selection in RDA models for benthic macroinvertebrates and zooplankton

Group	**Variable**	**Supporting references**
Benthic Macroinvertebrates	Alkalinity	(Jones et al. 2007; RISC 2009)
	Calcium (Ca)	(Schell and Kerekes 1989; RISC 2009)
	Chlorophyll-*a*	(Leppä et al. 2003; CCME 2011)
	Colour	(Karlsson et al. 2009)
	Conductivity	(Jones et al. 2007; RISC 2009)
	Dissolved Oxygen (DO)	(Jones et al. 2007; RISC 2009)
	Dissolved Organic Carbon (DOC)	(Strom et al. 2011; Moquin et al. 2014)
	Drainage Ratio (DR)	(Garcia and Carignan 1999; Lamontagne et al. 2000; McEachern et al. 2000)
	Macrophytes-SWIR1	(Jones et al. 2007; RISC 2009; Khanna et al. 2011; Oyama et al. 2015)

	Macrophytes-SWIR2	(Jones et al. 2007; RISC 2009; Khanna et al. 2011; Oyama et al. 2015)
	Maximum Depth (Z_{max})	(Jones et al. 2007; RISC 2009)
	pH	(Jones et al. 2007; RISC 2009)
	Temperature	(Jones et al. 2007; RISC 2009)
	Total Nitrogen (TN)	(Brodersen et al. 1998; Namayandeh and Quinlan 2011)
	Total Phosphorous (TP)	(Leppä et al. 2003; Namayandeh and Quinlan 2011)
	Turbidity	(RISC 2009)
	Surface area (SA)	(Namayandeh and Quinlan 2011)
	Water Catchment area (WC)	(Namayandeh and Quinlan 2011)
	% Organic Sediment (LOI)	(Jones et al. 2007; RISC 2009)
Zooplankton	Alkalinity	(Patoine 2002; Angeler and Goedkoop 2010; Lehtovaara et al. 2014)
	Calcium (Ca)	(Cairns and Yan 2009; Angeler and Goedkoop 2010; Gray et al. 2012)
	Chlorophyll-*a*	(Hall et al. 2003; Hessen et al. 2006; Gray et al. 2012; Dallas and Drake 2014)
	Colour	(Wissel and Ramacharan 2003; Angeler et al. 2013; Lehtovaara et al. 2014)
	Conductivity	(Angeler et al. 2013; Bégin and Vincent 2017)
	Dissolved Oxygen (DO)	(Wissel and Ramacharan 2003; Lehtovaara et al. 2014)
	Dissolved Organic Carbon (DOC)	(Stemberger et al. 2001; Patoine 2002)
	Drainage Ratio (DR)	(Patoine 2002; Jalal et al. 2005)
	Macrophytes-SWIR1	(Norlin et al. 2006; Khanna et al. 2011; Oyama et al. 2015; Grosbois et al. 2017)
	Macrophytes-SWIR2	(Norlin et al. 2006; Khanna et al. 2011; Oyama et al. 2015; Grosbois et al. 2017)
	Maximum Depth (Z_{max})	(Wissel and Ramacharan 2003; Lévesque et al. 2017)
	pH	(Angeler and Goedkoop 2010; Angeler et al. 2013; Lehtovaara et al. 2014)
	Temperature	(Shuter and Ing 1997; Lehtovaara et al. 2014)
	Total Nitrogen (TN)	(Stemberger et al. 2001; Angeler et al. 2013)
	Total Phosphorous (TP)	(Patoine 2002; Lehtovaara et al. 2014)
	Turbidity	(Angeler et al. 2013; Tartarotti et al. 2017)

	Surface area (SA)	(Stemberger et al. 2001; Lévesque et al. 2017)
	Water Catchment area (WC)	(Patoine 2002; Lévesque et al. 2017)

Table S3. Raw data of burned area

Lake	Total Burned (%)	Unburned (%)	Low Severity (%)	Moderate-low Severity (%)	Moderate-high Severity (%)	High Severity (%)	Total Burned (Ha)	bDR Total (Ha/SA)	bDR Low Severity (Ha/SA)	bDR Moderate-low Severity (Ha/SA)	bDR Moderate-high Severity (Ha/SA)	bDR High Severity (Ha/SA)
1	3.53	96.45	2.20	0.73	0.33	0.27	27.77	1.10	0.69	0.23	0.10	0.08
2	15.94	83.74	5.53	2.22	2.06	6.13	56.62	0.63	0.22	0.09	0.082	0.24
3	0.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
4	0.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
5	0.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
6	0.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
7	0.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
8	0.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
9	0.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
10	0.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
11	0.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
12	89.91	10.10	11.07	13.00	32.23	33.61	25.12	2.41	0.30	0.35	0.86	0.90
13	70.86	29.00	8.14	11.27	17.36	34.09	1074.24	26.59	3.06	4.23	6.52	12.79
14	68.60	31.24	8.27	10.83	16.35	33.15	852.06	10.31	1.24	1.63	2.46	4.98
15	95.11	4.86	5.79	12.18	30.54	46.60	1727.70	47.17	2.87	6.04	15.15	23.11
16	98.01	1.98	4.37	8.23	23.77	61.64	4202.71	56.52	2.52	4.75	13.71	35.55
17	98.17	1.82	4.12	7.35	24.13	62.57	3030.57	292.11	12.26	21.87	71.80	186.18
18	90.20	9.78	6.60	9.63	23.35	50.62	988.14	9.78	0.72	1.04	2.53	5.49
19	86.62	13.39	9.56	7.58	15.86	53.62	153.43	13.52	1.49	1.18	2.48	8.37
20	95.38	4.62	17.93	28.24	38.9	10.31	99.00	1.65	0.31	0.49	0.67	0.18

Table S4. Raw data of physical characteristics

Lake	Water Catchment Area (Ha)	Drainage Ratio (WC/SA)	Elevation (m. a.s.l.)	LOI Organic Sed. (%)	LOI Organic Soil (%)	Maximum Depth (m)	Perimeter (km)	Slope (%)	Surface Area (Ha)	Wetland in Catchment (Ha)
1	786.56	31.27	230.00	59.85	22.39	1.10	4.42	10.93	25.15	4.95
2	355.18	3.96	292.00	36.97	45.07	3.17	5.43	14.67	89.70	8.15
3	32.52	4.81	274.00	60.07	85.36	1.58	1.19	8.83	6.76	1.72
4	171.62	17.94	320.00	47.08	62.56	7.13	1.43	10.65	9.57	4.35
5	165.41	5.59	310.00	49.33	69.96	1.19	2.59	4.03	29.61	10.21
6	42.70	2.50	244.00	86.59	95.00	1.22	2.37	2.88	17.05	4.43
7	304.16	18.40	332.00	97.68	93.79	1.16	2.08	4.26	16.53	30.03
8	4036.52	460.91	55.00	51.48	61.56	0.85	3.78	7.13	8.76	95.73
9	951.02	4.15	246.00	6.38	71.78	9.78	9.81	15.04	228.97	61.69
10	2822.46	45.26	259.00	34.36	89.44	6.71	3.93	22.75	62.36	27.87
11	1731.29	14.30	260.00	5.17	96.60	9.45	5.40	19.63	121.05	17.60
12	27.94	2.68	238.00	84.41	96.42	1.83	1.76	5.66	10.42	1.27
13	1516.01	37.53	229.00	46.76	86.47	1.92	3.97	12.17	40.39	20.56
14	1242.07	15.03	256.00	7.14	90.77	3.99	4.45	12.52	82.62	17.97
15	1816.52	49.59	213.00	18.13	41.59	1.86	3.48	7.10	36.63	7.36
16	4288.04	57.67	76.00	38.95	37.98	6.77	6.05	12.03	74.36	29.53
17	3087.07	297.55	135.00	71.98	73.49	2.35	4.03	12.76	10.37	40.58
18	1095.50	10.84	77.00	32.53	38.03	2.87	8.53	11.99	101.03	26.77
19	177.12	15.61	99.00	62.07	94.33	5.06	1.62	24.68	11.35	2.63
20	103.79	1.73	213.00	63.73	98.20	0.91	3.86	3.10	60.16	4.31

Table S5. Raw data of water quality characteristics (Bellow Level of Quantification = BLQ) (1)

Lake	Alkalinity (mg/L CaCO3)	Ammonium (mg/L NH_4)	Chlorophyll-*a* (µg/L)	Colour (mg/L Pt)	Conductivity (µS/cm)	DO (mg/L)	DOC (mg/L)	Nitrite (mg/L NO_2)	pH
1	100.00	0.04	0.18	55.00	166.23	7.50	13.79	0.07	7.17
2	120.00	0.03	0.20	25.00	294.65	8.66	8.86	0.00	8.47
3	90.00	0.15	0.13	95.00	107.10	10.00	14.03	0.02	8.75
4	155.00	0.09	0.21	5.00	276.90	8.49	8.49	0.01	8.36
5	100.00	0.15	0.77	65.00	150.13	8.58	16.60	0.03	8.38
6	65.00	0.07	0.36	45.00	121.68	9.92	11.73	0.01	8.15
7	120.00	BLQ	0.62	85.00	134.30	10.18	13.76	0.02	6.46
8	185.00	0.50	0.40	60.00	468.25	12.03	28.17	0.02	8.27
9	150.00	0.00	0.38	0.00	550.75	11.47	5.08	0.01	8.33
10	205.00	BLQ	1.13	10.00	537.25	11.43	10.29	0.00	8.40
11	155.00	0.11	0.50	0.00	663.25	11.15	6.56	0.00	8.38
12	40.00	0.07	0.23	70.00	66.70	10.00	10.54	0.01	7.82
13	85.00	0.36	0.21	70.00	156.65	9.11	9.39	0.03	7.41
14	95.00	0.01	0.33	30.00	178.50	9.06	10.97	0.02	8.03
15	150.00	0.03	0.83	95.00	445.45	8.78	11.30	0.04	8.28
16	165.00	0.02	0.47	25.00	227.68	9.55	8.64	0.02	8.32
17	150.00	BLQ	0.15	120.00	248.43	7.14	15.04	0.01	6.51
18	155.00	0.06	0.44	55.00	566.75	10.10	11.88	0.01	8.14
19	125.00	0.01	0.34	50.00	213.30	10.54	10.88	0.01	8.26
20	60.00	0.01	0.87	25.00	91.83	8.66	8.97	0.01	8.17

Table S6. Raw data of water quality characteristics (Bellow Level of Quantification = BLQ) (2)

Lake	Phenols (mg/L C_6H_5OH)	Phosphate (mg/L PO_4)	Sulfate (mg/L SO_4)	SWIR1	SWIR2	Temperature (°C)	TN (mg/L)	TOC (mg/L)	TP (mg/L)	Turbidity (NTU)
1	0.22	0.23	1.00	400.16	236.34	16.23	0.64	12.73	0.06	2.00
2	0.19	0.09	61.00	199.96	127.11	16.98	0.43	9.19	0.09	240.00
3	0.37	0.07	2.00	317.75	165.78	15.10	0.63	11.62	0.05	8.00
4	0.19	0.02	1.00	493.13	280.63	15.05	0.32	8.61	0.04	2.00
5	0.29	0.25	3.00	311.69	174.13	16.38	0.02	9.22	0.08	8.00
6	0.01	0.04	4.00	319.70	164.98	14.55	0.55	12.12	0.08	8.00
7	0.17	0.05	2.00	198.17	119.50	14.55	0.68	14.30	0.06	4.00
8	0.10	0.19	1.00	632.47	344.39	13.23	0.78	34.11	0.02	0.00
9	0.00	0.32	148.00	95.50	51.87	11.88	0.27	35.63	0.03	2.00
10	0.09	0.06	160.00	133.91	76.31	13.10	1.11	12.52	0.04	4.00
11	0.22	0.30	205.00	110.64	60.91	12.22	0.35	7.69	0.02	4.00
12	BLQ	0.08	2.00	319.24	185.08	15.33	0.35	11.97	0.06	6.00
13	0.20	0.62	63.00	253.62	151.44	13.23	0.37	9.68	0.08	245.00
14	0.38	0.08	2.00	163.89	101.77	14.35	0.30	8.60	0.05	4.00
15	0.46	0.20	108.00	620.12	373.56	12.88	0.28	10.67	0.05	660.00
16	0.37	0.09	2.00	206.95	128.59	15.40	0.35	9.11	0.08	6.00
17	0.57	0.07	2.00	634.69	363.74	14.43	0.44	15.61	0.08	6.00
18	BLQ	0.01	205.00	315.85	190.86	14.13	0.60	11.01	0.04	620.00
19	BLQ	BLQ	6.00	365.44	211.78	14.43	0.35	10.49	0.04	18.00
20	0.13	0.08	0.00	305.39	170.52	13.95	0.46	9.20	0.05	4.00

Table S7. Raw data of water chemistry (Bellow Level of Detection = BLD, Bellow Level of Quantification = BLQ)

Lake	Ag (mg/L)	Ba (mg/L)	Ca (mg/L)	Cu (mg/L)	Fe (mg/L)	K (mg/L)	Li (mg/L)	Mg (mg/L)	Mn (mg/L)	Na (mg/L)	S (mg/L)	Sr (mg/L)	Zn (mg/L)
1	0.15	0.05	30.07	BLD	0.07	0.21	BLD	6.05	0.03	0.56	4.87	0.11	0.02
2	0.04	0.08	53.72	BLD	0.00	0.67	0.01	9.05	0.03	1.39	22.57	0.46	0.00
3	0.02	0.03	19.03	BLQ	0.02	0.14	BLD	5.58	0.03	0.40	2.52	0.05	0.00
4	0.01	0.07	50.22	BLQ	0.01	0.40	0.01	13.73	0.03	1.05	8.39	0.08	BLD
5	0.00	0.05	24.36	6.65	0.02	0.38	BLD	7.52	0.03	0.44	2.37	0.05	0.00
6	0.00	0.05	20.52	3.15	0.07	0.20	BLD	6.77	0.03	0.92	0.80	0.04	0.00
7	0.01	0.03	24.25	2.05	0.02	0.21	BLD	6.50	0.03	0.27	4.14	0.10	0.03
8	0.01	0.02	53.24	0.01	BLQ	0.76	BLD	17.90	0.02	8.71	22.47	0.15	BLQ
9	0.01	0.03	47.33	0.01	BLQ	0.93	BLD	31.39	0.01	2.37	40.27	1.86	BLQ
10	0.01	0.06	45.40	0.01	BLQ	1.19	0.00	32.78	0.01	2.72	44.83	1.43	BLQ
11	0.01	0.02	64.33	0.01	BLQ	1.39	0.00	37.72	0.01	3.40	64.75	0.79	BLQ
12	BLQ	0.02	9.51	BLQ	0.05	0.31	BLD	3.72	0.03	0.64	2.92	0.04	0.00
13	BLQ	0.06	22.72	0.19	0.02	0.45	0.01	5.21	0.03	4.95	4.13	0.14	BLQ
14	BLD	0.07	28.10	1.51	0.02	0.47	BLD	6.44	0.03	3.58	3.65	0.12	0.00
15	BLD	0.08	53.28	4.08	0.06	0.49	0.01	18.83	0.03	3.80	16.72	0.14	0.01
16	0.00	0.11	41.20	BLQ	0.01	0.68	BLD	10.67	0.03	1.85	4.15	0.11	0.00
17	0.00	0.07	48.00	1.17	0.10	0.53	BLD	14.54	0.03	2.16	5.88	0.10	0.01
18	BLD	0.10	109.59	0.83	0.02	0.74	0.01	20.00	0.03	8.40	34.65	1.08	0.00
19	BLD	0.07	38.64	1.21	0.02	0.53	BLD	12.77	0.03	0.67	0.98	0.11	BLD
20	BLD	0.05	15.02	39.85	0.12	0.15	BLD	5.82	0.03	0.74	0.92	0.05	0.01

Table S8. Raw data of methylmercury (Mehg), carbon 13 (δ^{13}C), and nitrogen 15 (δ^{15}N) (Bellow Level of Quantification = BLQ)

Lake	Predator MeHg	Sediment MeHg	Water MeHg	Zooplankton MeHg	Predator δ^{13}C	Predator δ^{15}N	Sediment δ^{13}C	Sediment δ^{15}N	Zooplankton δ^{13}C	Zooplankton δ^{15}N
1	20.34	2.46	0.15	9.96	-29.71	2.36	-28.53	-1.27	-28.50	2.83
2	111.89	1.37	0.03	3.28	-29.29	3.43	-25.01	-0.46	-30.44	9.05
3	60.73	0.00	0.05	8.74	-30.20	3.49	-27.35	-0.96	-27.68	1.43
4	56.88	3.84	0.05	24.00	-30.98	3.38	-29.13	-0.41	-37.38	7.45
5	86.70	0.92	0.13	52.27	-25.02	3.98	-26.53	-0.34	-28.95	3.67
6	21.04	0.53	0.03	3.99	-28.07	4.06	-28.91	-0.37	-29.98	4.25
7	43.70	3.59	0.15	58.38	-27.40	4.75	-27.00	-0.69	-25.10	3.62
12	8.96	3.64	0.05	14.93	-30.53	6.38	-28.84	1.42	-30.05	4.11
13	23.54	0.97	0.06	17.90	-31.46	2.02	-27.95	0.56	-33.22	4.97
14	58.44	0.00	0.02	12.66	-30.25	5.46	-28.35	-0.64	-33.81	7.07
15	65.10	0.81	0.08	35.24	-29.80	3.74	-27.26	0.59	-32.64	6.25
16	121.41	1.31	BLQ	4.49	-29.67	4.85	-28.38	-0.26	-30.42	6.13
17	80.96	8.75	0.26	36.00	-35.55	2.25	-31.50	-0.70	-35.45	3.36
18	75.77	1.16	0.05	10.62	-33.07	3.97	-30.06	-0.71	-33.65	4.04
19	40.57	3.23	0.05	43.79	-33.32	4.91	-30.13	0.08	-34.78	6.84
20	36.79	2.92	0.02	9.46	-31.66	4.62	-28.26	1.32	-30.30	3.74

Table S9. Raw data of zooplankton Genus estimated relative abundance (1)

Lake	*Skistodiaptomus*	*Leptodiaptomus*	*Heterocope*	*Diacyclops*	*Epischura*	*Tropocyclops*	*Microcyclops*	*Ergasilus*	*Acanthocyclops*
1	21	201	22	3	0	0	17	0	0
2	31	476	8	0	0	0	13	0	0
3	74	137	12	0	0	0	10	0	0
4	21	330	8	0	0	2	8	0	0
5	0	350	0	0	0	0	62	0	0
6	96	186	16	0	0	0	9	0	1
7	60	195	23	0	0	0	8	0	0
8	0	12	0	0	0	0	160	11	0
9	0	15	0	0	0	0	25	0	0
10	0	0	0	0	0	0	0	0	0
11	0	0	0	0	0	0	9	0	0
12	3	163	0	0	1	0	66	0	0
13	30	363	7	0	0	0	308	1	0
14	7	107	1	0	0	0	269	3	0
15	9	76	28	0	0	0	23	0	0
16	0	1	14	0	65	1	20	0	0
17	2	195	3	0	0	0	33	1	1
18	5	34	25	0	124	10	19	1	0
19	34	43	12	0	0	0	15	0	0
20	4	84	26	0	0	0	14	2	11

Table S10. Raw data of zooplankton Genus estimated relative abundance (2)

Lake	*Daphnia*	*Alonella*	*Alona*	*Chydorus*	*Bosmina*	*Sida*	*Ceriodaphnia*	*Diaphanosoma*	Immature Calanoids	*Polyphemus*	*Harpacticoida*
1	12	0	0	16	0	0	0	0	35	0	0
2	0	0	0	0	0	0	0	0	179	0	0
3	5	2	0	0	0	0	0	0	109	0	0
4	4	1	0	0	0	0	7	0	51	0	0
5	20	0	0	0	0	0	18	1	36	0	0
6	14	3	0	1	0	0	0	0	54	0	0
7	0	0	0	0	0	0	0	0	79	0	0
8	7	12	29	41	18	0	23	0	5	37	25
9	0	6	7	19	19	1	0	3	0	1	0
10	0	20	5	0	21	2	0	0	0	0	0
11	60	0	0	13	108	8	0	0	0	0	0
12	0	0	1	0	0	0	15	16	54	0	0
13	3	0	0	0	0	0	0	0	48	0	0
14	1	0	0	0	44	10	0	2	50	0	0
15	41	0	0	0	5	14	0	0	142	0	0
16	0	0	0	0	11	71	1	0	116	0	0
17	7	0	7	15	1	2	9	3	29	0	0
18	0	2	0	2	6	2	22	6	52	0	0
19	93	0	0	0	0	5	0	0	230	0	0
20	123	0	0	0	0	12	0	0	25	0	0

Table S11. Raw data of shredder benthic macroinvertebrate estimated family abundance

Lake	Calamoceratidae	Capniidae	Chrysomelidae	Ephydridae	Gammaridae	Haliplidae	Hyalellidae	Leptoceridae	Molannidae	Phryganeidae
1	0.00	0.00	0.00	0.00	0.00	0.00	127.97	0.00	0.00	9.52
2	0.00	0.00	3.03	0.00	29.70	13.33	480.61	6.67	0.00	13.33
3	0.00	0.00	0.00	0.00	0.00	2.22	35.56	0.00	0.00	2.22
4	0.00	0.00	0.00	0.00	5.56	0.00	993.33	0.00	0.00	12.22
5	0.00	0.00	0.00	0.00	0.00	0.00	57.16	5.24	0.00	1.19
6	0.00	0.00	0.00	0.00	152.86	0.00	384.29	12.86	0.00	35.71
7	0.00	0.00	0.00	1.08	0.00	0.00	25.98	1.76	0.00	0.00
8	0.00	0.00	0.00	0.00	0.00	38.89	43.06	0.00	0.00	37.50
9	0.00	0.00	0.00	0.00	64.35	0.00	191.90	22.69	0.00	5.79
10	0.00	0.00	0.00	0.00	61.46	1.52	167.02	2.22	0.00	12.88
11	0.00	0.00	0.00	0.00	0.00	1.67	275.61	37.88	0.00	0.00
12	0.00	0.00	0.00	0.00	6.43	0.00	56.81	0.00	4.76	0.33
13	0.00	0.00	0.00	0.00	1.90	0.00	410.48	0.00	0.00	20.00
14	0.00	0.00	0.00	0.00	43.75	2.08	430.42	0.00	0.00	10.42
15	0.00	0.00	6.67	6.67	46.67	6.67	1400.00	0.00	0.00	26.67
16	5.56	0.00	0.00	0.00	213.42	20.00	587.52	0.00	0.00	10.68
17	0.00	5.56	0.00	0.00	94.44	9.26	1140.74	0.00	0.00	16.67
18	0.00	0.00	0.00	0.00	619.36	14.39	857.23	0.00	0.00	2.08
19	0.00	0.00	2.08	0.00	224.43	4.17	228.60	4.17	1.52	0.00
20	0.00	0.00	0.00	0.00	12.50	8.33	24.40	4.17	0.00	0.00

Table S12. Raw data of scraper benthic macroinvertebrate estimated family abundance

Lake	Corixidae	Elmidae	Heptageniidae	Hydrobiidae	Limnephildae	Lymnaeidae	Physidae	Planorbidae	Scirtidae	Valvatidae
1	0.88	0.00	0.00	0.00	0.00	0.00	0.00	22.86	0.00	2.63
2	6.67	0.00	0.00	0.00	13.33	15.15	0.00	126.67	0.00	13.33
3	0.00	0.00	0.00	0.00	0.00	36.67	0.00	17.22	0.00	9.44
4	6.67	0.00	0.00	6.67	0.00	0.00	0.00	33.33	0.00	0.00
5	1.39	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
6	4.76	0.00	0.00	0.00	0.00	0.00	0.00	19.52	0.00	0.00
7	0.00	0.00	0.00	0.00	0.00	1.08	0.00	2.22	4.30	8.24
8	88.89	0.00	0.00	0.00	0.00	163.89	0.00	281.94	5.56	5.56
9	13.66	0.00	35.42	0.00	0.00	102.31	0.00	61.11	0.00	3.70
10	3.74	0.00	0.00	0.00	0.00	23.06	0.00	12.07	0.00	0.00
11	1.52	0.00	8.18	0.00	0.00	33.18	0.00	69.55	0.00	6.67
12	36.67	0.00	0.00	0.00	1.67	0.00	0.00	2.33	0.00	0.00
13	0.00	0.00	0.00	0.95	0.00	32.38	6.67	105.71	0.00	6.67
14	5.56	0.00	0.00	5.56	0.00	6.94	20.83	45.97	0.00	0.00
15	0.00	0.00	0.00	6.67	0.00	13.33	13.33	106.67	0.00	0.00
16	0.00	0.00	0.00	0.00	0.00	0.00	0.00	48.12	0.00	0.00
17	0.00	0.00	0.00	0.00	0.00	57.41	5.56	33.33	0.00	0.00
18	135.34	0.00	0.00	20.00	0.00	8.75	28.75	64.17	0.00	0.00
19	8.71	0.00	0.00	0.00	0.00	0.00	0.00	4.17	0.00	0.00
20	5.36	4.17	0.00	0.00	0.00	0.00	0.00	15.71	0.00	0.00

Table S13. Raw data of gathering collector benthic macroinvertebrate estimated family abundance

Lake	Baetidae	Caenidae	Chironomidae	Dixidae	Ephemeridae	Leptophlebiidae	Psychodidae
1	49.52	114.06	830.33	0.00	0.00	14.29	0.00
2	209.70	1036.97	754.55	0.00	306.67	162.42	0.00
3	85.00	10.00	516.67	0.00	0.00	0.00	0.00
4	392.22	86.67	404.44	0.00	0.00	40.00	0.00
5	20.73	26.66	253.97	0.00	0.00	0.00	0.00
6	21.43	70.95	795.24	0.00	0.00	0.00	0.00
7	0.00	46.47	253.48	0.00	0.00	0.00	0.00
8	5.56	94.44	884.72	0.00	0.00	0.00	0.00
9	63.89	22.69	397.92	0.00	0.00	0.00	0.00
10	1.52	16.94	410.98	0.00	0.00	0.00	0.00
11	38.33	14.55	545.76	0.00	0.00	0.00	0.00
12	22.52	0.33	479.71	0.00	0.00	39.24	0.00
13	44.76	9.52	832.38	1.90	0.00	0.00	0.00
14	17.78	80.97	469.86	0.00	6.67	6.67	0.00
15	40.00	73.33	2040.00	46.67	0.00	0.00	0.00
16	8.12	22.22	713.50	0.00	0.00	2.56	0.00
17	5.56	0.00	600.00	0.00	0.00	0.00	0.00
18	17.50	105.83	59.51	6.67	0.00	0.00	0.00
19	8.33	37.50	181.25	0.00	0.00	0.00	0.00
20	0.00	0.00	1035.83	0.00	0.00	13.33	12.50

Table S14. Raw data of filtering collector benthic macroinvertebrate estimated family abundance

Lake	Brachycentridae	Hydroptilidae	Polycentropoidae	Simuliidae	Sphaeriidae
1	0.00	0.00	4.76	0.00	11.93
2	0.00	0.00	20.00	0.00	106.06
3	0.00	0.00	2.22	0.00	8.33
4	0.00	0.00	18.89	0.00	0.00
5	0.00	0.00	2.56	0.00	2.38
6	0.00	0.00	8.10	0.00	43.33
7	0.00	0.00	0.00	0.00	7.09
8	0.00	0.00	0.00	0.00	0.00
9	3.70	0.00	0.00	0.00	35.19
10	1.52	0.00	1.52	0.00	85.56
11	0.00	0.00	0.00	0.00	34.70
12	0.00	1.00	0.33	0.00	5.10
13	0.00	11.43	14.29	0.00	0.00
14	0.00	0.00	2.08	2.78	17.08
15	0.00	6.67	6.67	0.00	20.00
16	6.67	0.00	0.00	0.00	9.23
17	5.56	0.00	0.00	0.00	14.81
18	0.00	0.00	0.00	3.03	8.75
19	0.00	0.00	4.17	0.00	4.17
20	0.00	0.00	0.00	0.00	0.00

Table S15. Raw data of predator benthic macroinvertebrate estimated family abundance (1)

Lake	Aeshnidae	Athericidae	Ceratopogonidae	Chaoboridae	Coenagrionidae	Corduliidae	Dolichipodidae	Dytiscidae	Hebridae
1	6.67	0.00	30.18	0.00	24.76	0.00	0.00	0.00	0.00
2	0.00	0.00	12.73	0.00	73.33	13.33	0.00	6.67	0.00
3	4.44	0.00	71.67	2.22	10.00	10.00	0.00	10.56	0.00
4	5.56	0.00	12.22	0.00	62.22	6.67	0.00	0.00	0.00
5	1.28	0.00	1.28	0.00	12.48	0.00	0.00	0.00	0.00
6	0.00	0.00	18.10	0.00	4.76	4.76	0.00	0.00	0.00
7	0.00	0.00	42.43	0.00	0.68	0.00	2.22	0.00	0.00
8	0.00	11.11	41.67	0.00	0.00	0.00	0.00	5.56	0.00
9	0.00	3.70	0.00	0.00	2.08	0.00	0.00	0.00	0.00
10	1.52	0.00	9.85	0.00	1.52	0.00	0.00	0.00	0.00
11	0.00	1.67	28.18	0.00	0.00	0.00	0.00	0.00	0.00
12	0.95	0.00	14.62	68.71	0.67	0.00	0.00	0.00	0.00
13	2.78	0.00	45.71	0.00	51.43	0.00	0.00	0.00	0.00
14	0.00	0.00	7.64	0.00	12.92	0.00	0.00	8.33	0.00
15	13.33	0.00	33.33	0.00	60.00	40.00	0.00	20.00	0.00
16	5.56	0.00	306.67	0.00	52.22	0.00	0.00	20.00	0.00
17	0.00	14.81	300.00	0.00	61.11	0.00	0.00	3.70	5.56
18	8.75	0.00	0.00	0.00	19.58	0.00	0.00	0.00	0.00
19	0.00	0.00	1.52	4.17	78.60	0.00	0.00	0.00	0.00
20	2.38	4.17	60.00	0.00	14.88	2.38	4.17	4.17	12.50

Table S16. Raw data of predator benthic macroinvertebrate estimated family abundance (2)

Lake	Glossiphoniidae	Libellulidae	Macromiidae	Mesoveliidae	Muscidae	Notonectidae	Tabanidae	Tipulidae	Veliidae
1	0.00	0.00	0.00	0.00	0.00	4.76	0.00	0.00	0.00
2	0.00	0.00	6.67	0.00	0.00	0.00	0.00	0.00	0.00
3	0.00	0.00	0.00	0.00	2.78	0.00	0.00	2.22	0.00
4	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	5.56
5	0.00	0.00	0.00	0.00	0.00	1.19	0.00	0.00	0.00
6	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
7	0.00	2.22	0.00	0.00	0.00	0.00	0.00	0.00	0.00
8	0.00	0.00	0.00	0.00	0.00	0.00	0.00	16.67	0.00
9	0.00	0.00	0.00	0.00	0.00	0.00	9.49	0.00	0.00
10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
11	0.00	0.00	0.00	0.00	0.00	0.00	3.18	0.00	0.00
12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
13	0.00	6.67	0.00	0.95	0.00	0.00	4.76	0.95	0.00
14	0.00	0.00	0.00	0.00	0.00	0.00	13.33	2.78	0.00
15	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
16	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
17	0.00	0.00	0.00	0.00	5.56	0.00	0.00	3.70	0.00
18	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
19	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
20	4.17	0.00	0.00	0.00	0.00	0.00	4.17	8.33	0.00

Table S17. Raw data of parasitic benthic macroinvertebrate estimated family abundance (1)

Lake	Arrenuridae	Erpobdellidae	Feltriidae	Hirudinidae	Hydrodromidae	Hygrobatidae
1	0.00	0.00	0.00	0.00	0.00	0.00
2	0.00	0.00	0.00	0.00	0.00	0.00
3	0.00	0.00	0.00	2.78	0.00	2.78
4	0.00	0.00	0.00	0.00	0.00	0.00
5	1.19	0.00	0.00	0.00	0.00	0.00
6	0.00	3.33	0.00	0.00	0.00	0.00
7	0.00	0.00	0.00	0.00	0.00	0.00
8	4.17	0.00	0.00	0.00	0.00	11.11
9	0.00	0.00	0.00	0.00	0.00	0.00
10	1.52	0.00	0.00	0.00	0.00	0.00
11	0.00	0.00	0.00	0.00	0.00	0.00
12	0.33	0.00	0.00	1.67	0.00	0.00
13	7.62	0.00	0.00	0.00	0.00	1.27
14	0.00	0.00	0.00	0.00	0.00	0.00
15	6.67	0.00	0.00	0.00	0.00	6.67
16	12.22	0.00	0.00	5.56	0.00	0.00
17	5.56	5.56	5.56	0.00	25.93	11.11
18	0.00	0.00	0.00	0.00	0.00	0.00
19	0.00	0.00	0.00	0.00	0.00	0.00
20	25.95	4.17	0.00	0.00	0.00	0.00

Table S18. Raw data of parasitic benthic macroinvertebrate estimated family abundance (2)

Lake	Lebertiidae	Limnesiidae	Mideopsidae	Pionidae	Piscicolidae	Unionicolidae
1	4.76	0.00	0.00	0.00	34.21	0.00
2	0.00	0.00	6.67	0.00	6.67	0.00
3	0.00	2.78	0.00	0.00	0.00	0.00
4	0.00	0.00	0.00	0.00	22.22	0.00
5	0.00	0.00	0.00	0.00	1.19	0.00
6	8.10	0.00	9.52	0.00	16.67	0.00
7	0.00	0.68	2.22	0.00	0.68	0.00
8	0.00	0.00	0.00	0.00	0.00	0.00
9	0.00	0.00	3.70	0.00	2.08	0.00
10	0.00	0.00	0.00	0.00	9.85	0.00
11	1.67	0.00	0.00	0.00	0.00	0.00
12	0.00	4.76	4.76	1.67	11.76	4.76
13	0.00	0.00	6.67	0.00	0.00	0.00
14	0.00	0.00	20.00	0.00	6.67	0.00
15	0.00	6.67	0.00	0.00	0.00	0.00
16	9.23	2.56	0.00	2.56	11.79	10.68
17	0.00	0.00	0.00	0.00	5.56	0.00
18	0.00	0.00	3.03	0.00	6.67	0.00
19	0.00	7.20	0.00	3.03	4.17	0.00
20	0.00	6.67	0.00	0.00	0.00	0.00

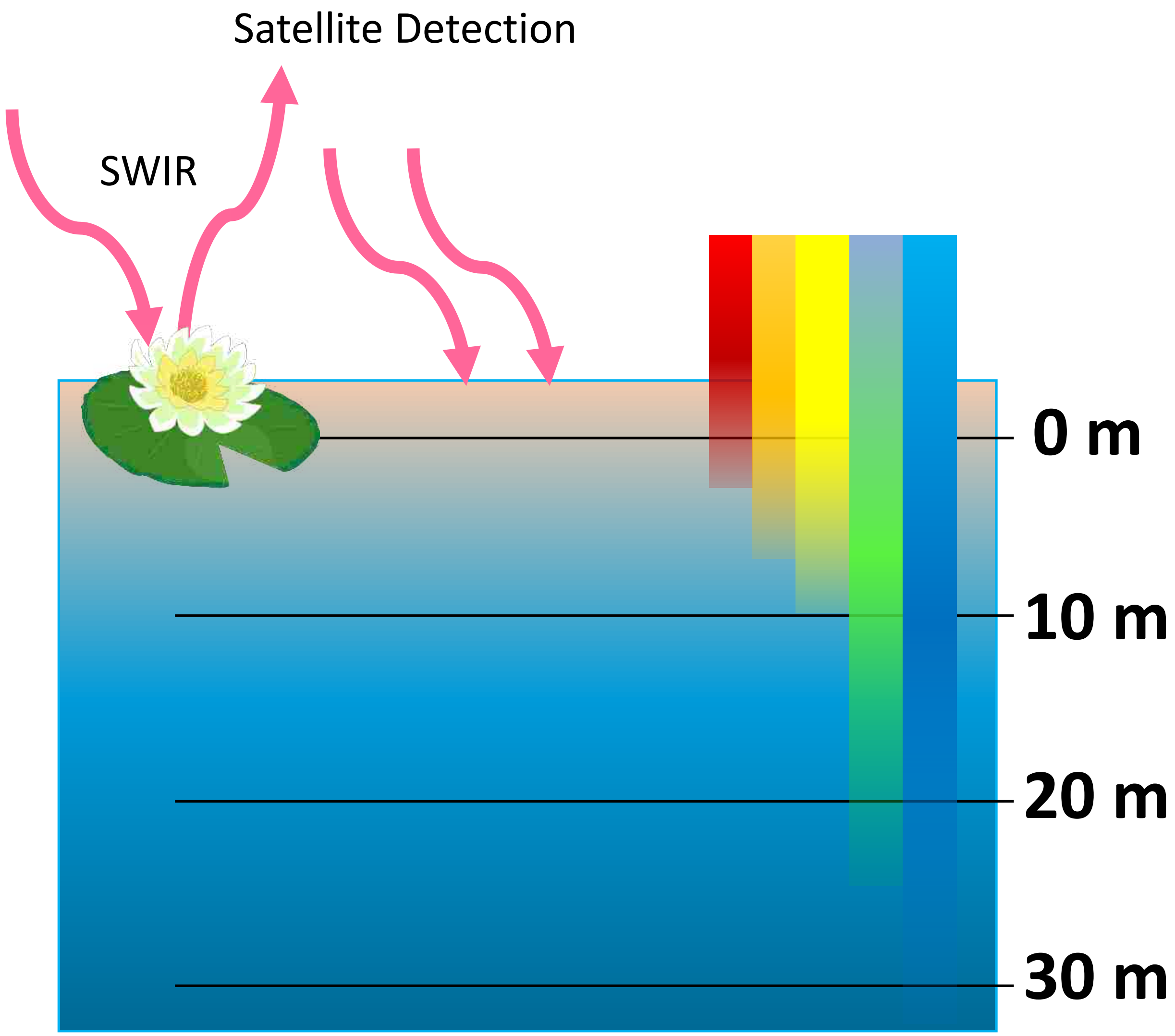

Figure S1: Depiction of light absorbance in water, where short-wave infrared (SWIR) wavelengths are instantaneously absorbed and need to reflect off the surface of vegetation.

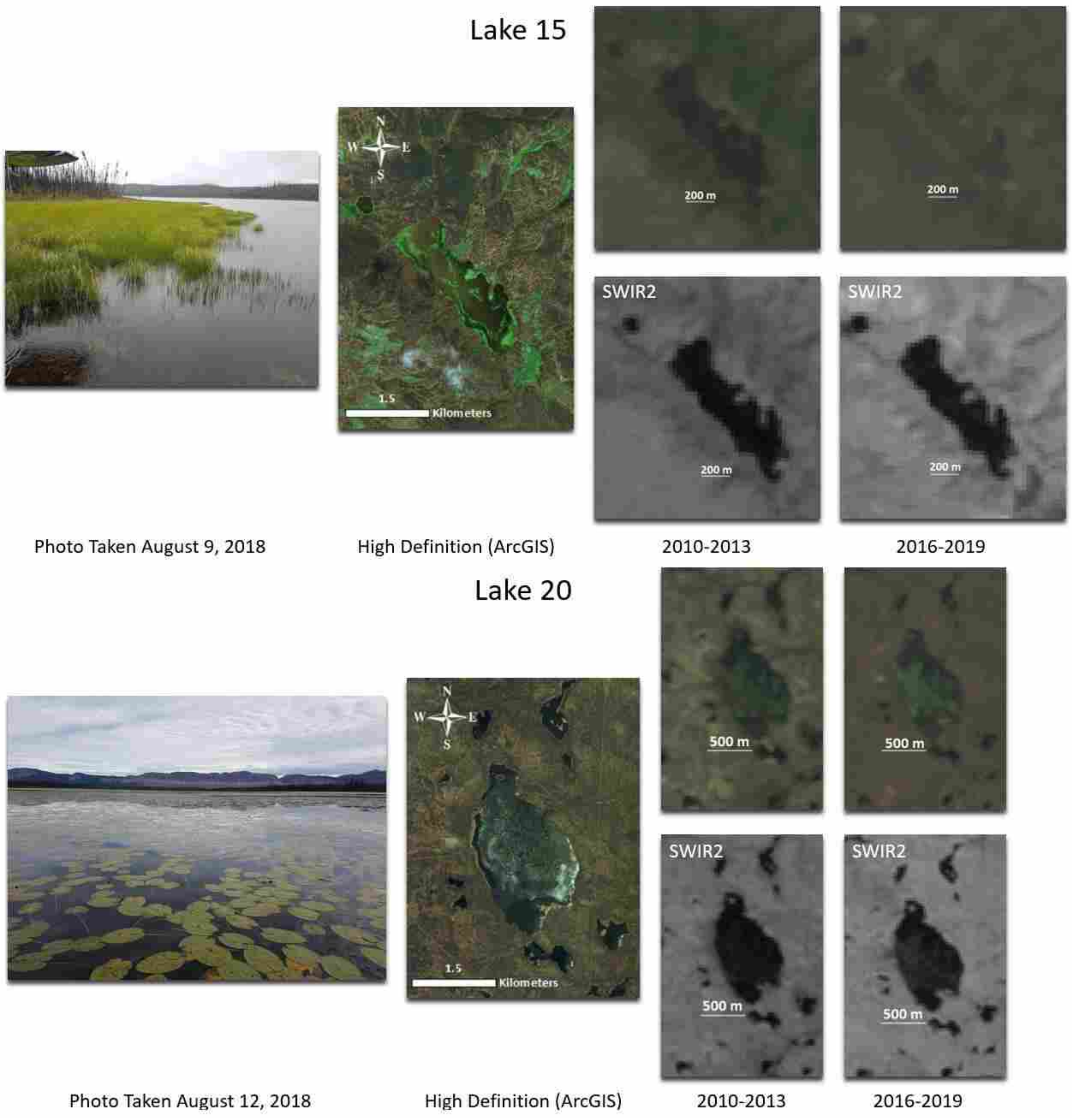

Figure S2: Photographs of Lake 15 and 20 at the time of sampling, high definition image from the basemap of GIS software (ArcGIS) compiled after the 2014 wildfire, compiled natural colour images and SWIR2 reflectance images from Landsat 7 from June-August 2010-2013 (pre-fire) and June-August 2016-2019 (post-fire).

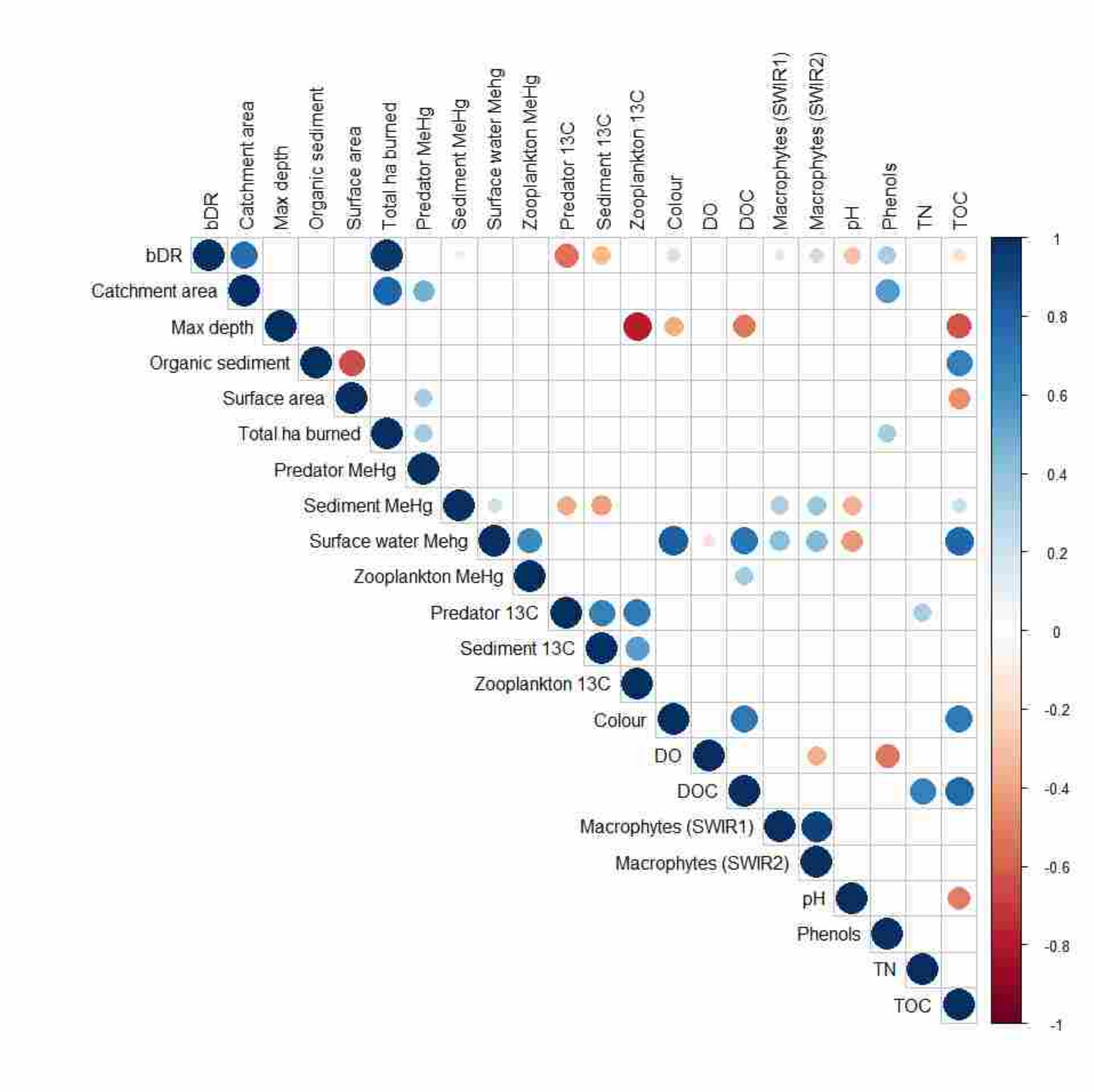

Figure S3: Spearman correlations between the burn drainage ratio (hectares burned/lake surface area), lake physical properties, methylmercury (MeHg), carbon ($\delta^{13}C$) isotopes, and water quality variables. Blue is indicative of positive correlations and negative correlations are red. Colour intensity and size of the circle are proportional to the correlation coefficients. Correlations that were not significant ($p > 0.05$) are indicated by empty cells. Burned drainage ratio = bDR, dissolved oxygen = DO, dissolved organic carbon = DOC, total nitrogen = TN, and total organic carbon = TOC.

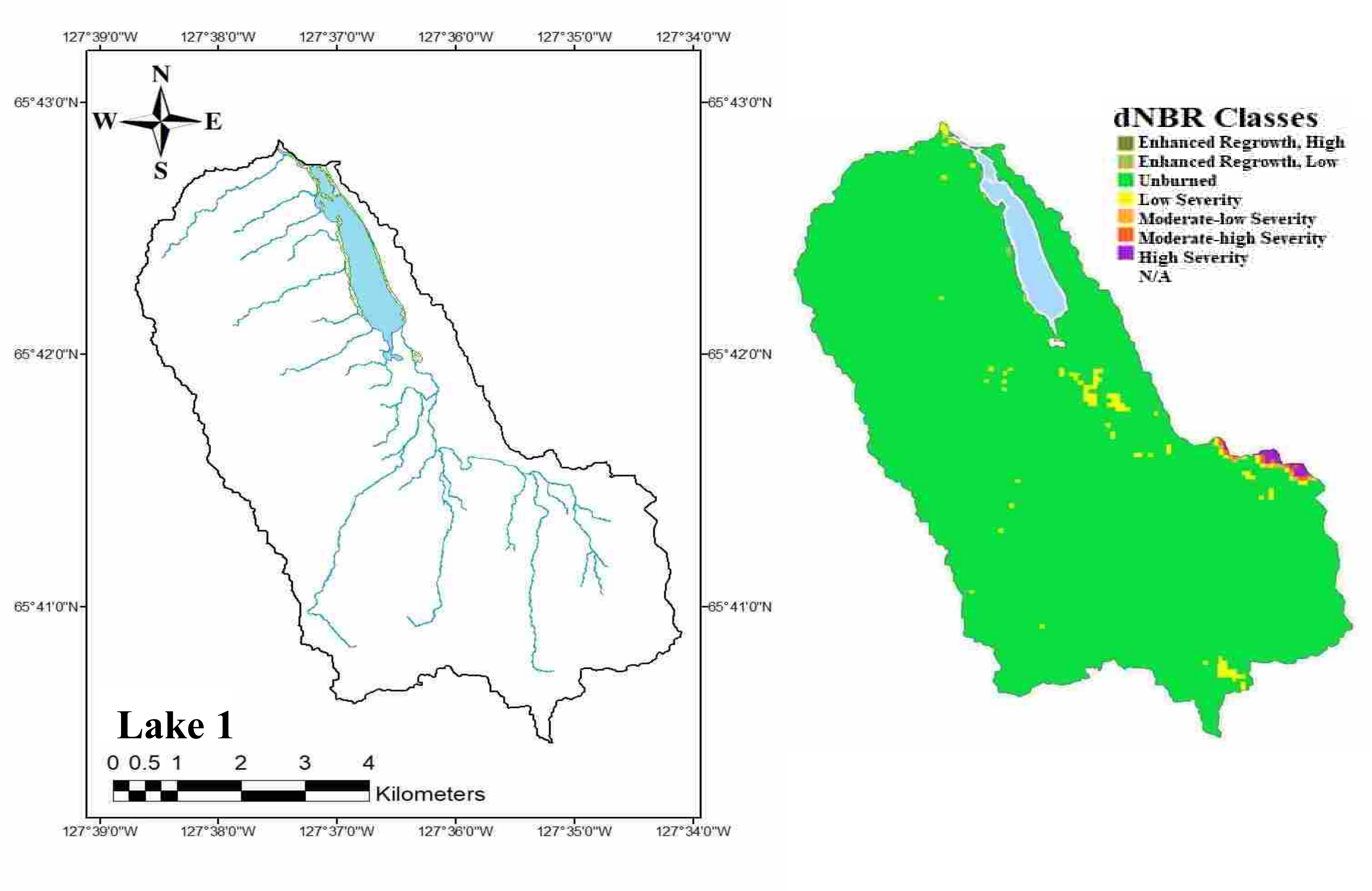

Figure S4: Lake 1 water catchment, hydrological inflow (blue outline), wetland area (green outline), and areas affected by fire scaring.

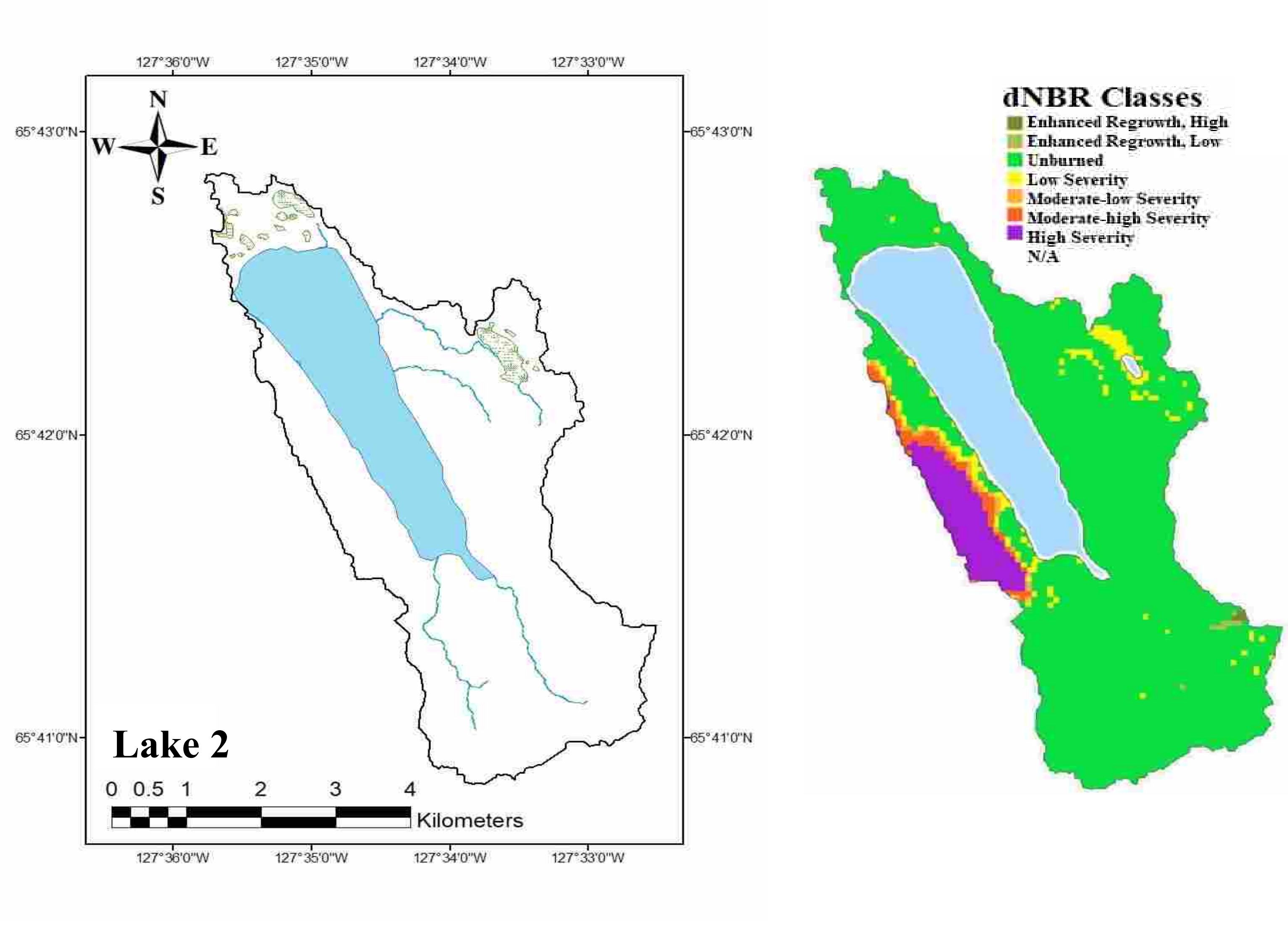

Figure S5: Lake 2 water catchment, hydrological inflow (blue outline), wetland area (green outline), and areas affected by fire scaring.

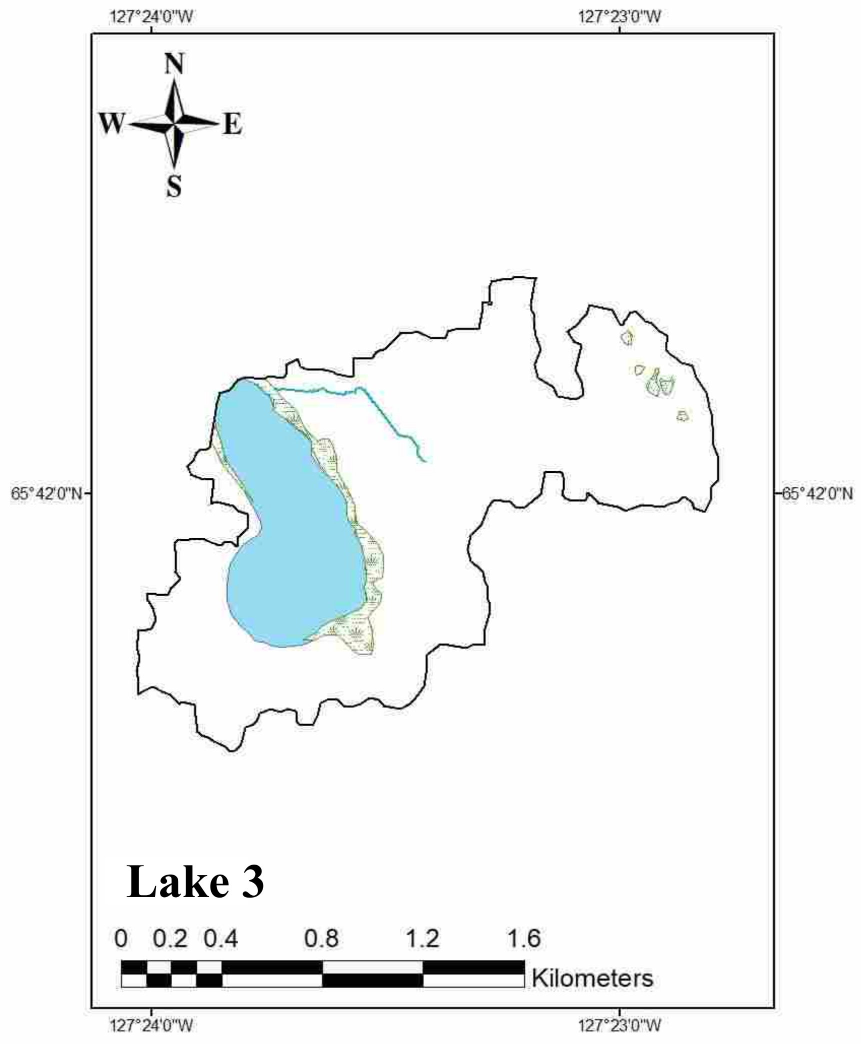

Figure S6: Lake 3 water catchment, hydrological inflow (blue outline), and wetland area (green outline).

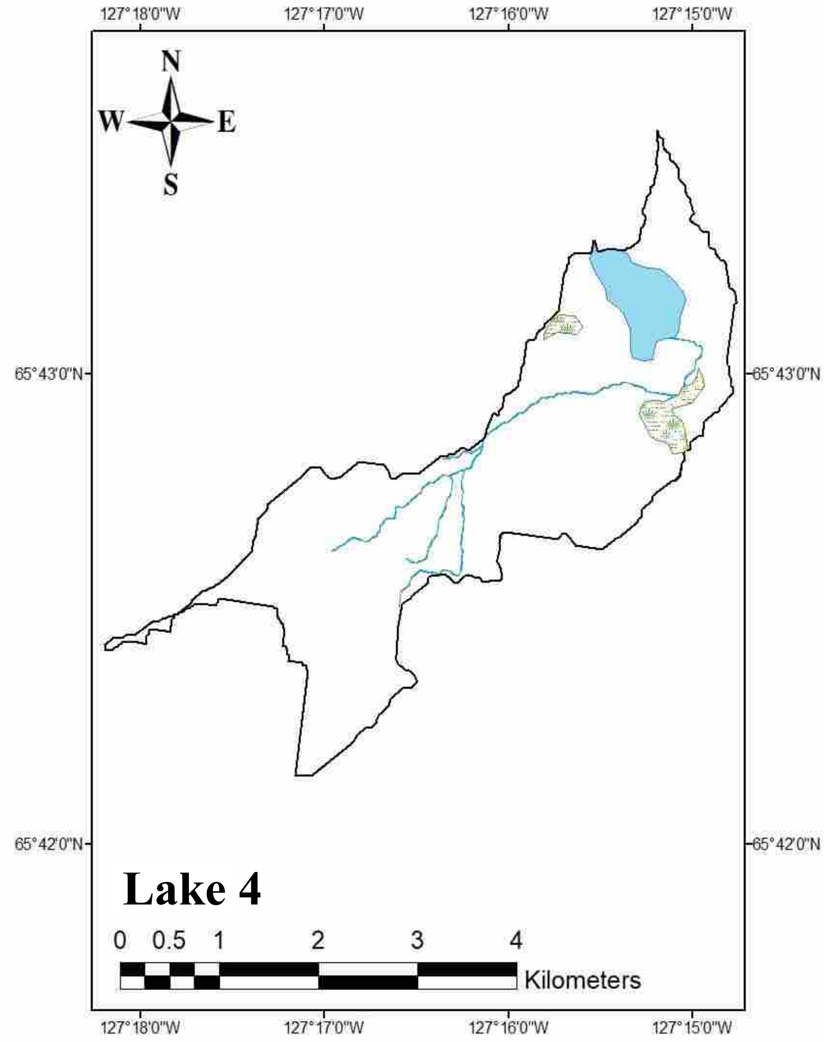

Figure S7: Lake 4 water catchment, hydrological inflow (blue outline), and wetland area (green outline).

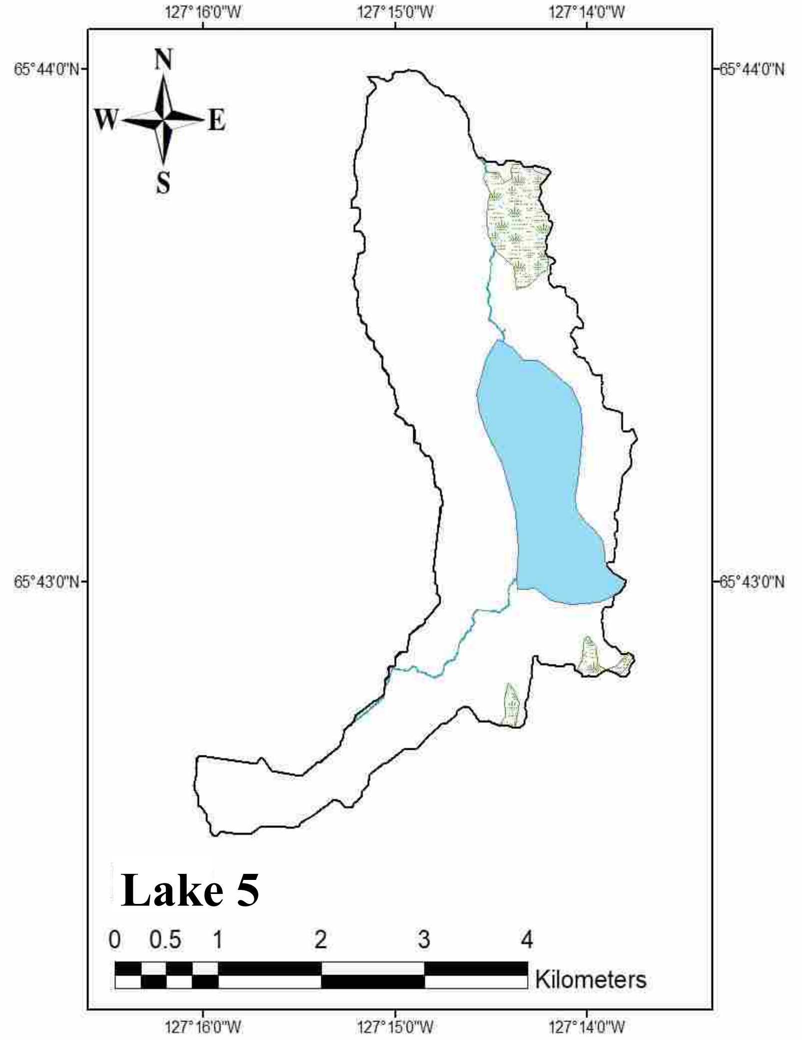

Figure S8: Lake 5 water catchment, hydrological inflow (blue outline), and wetland area (green outline).

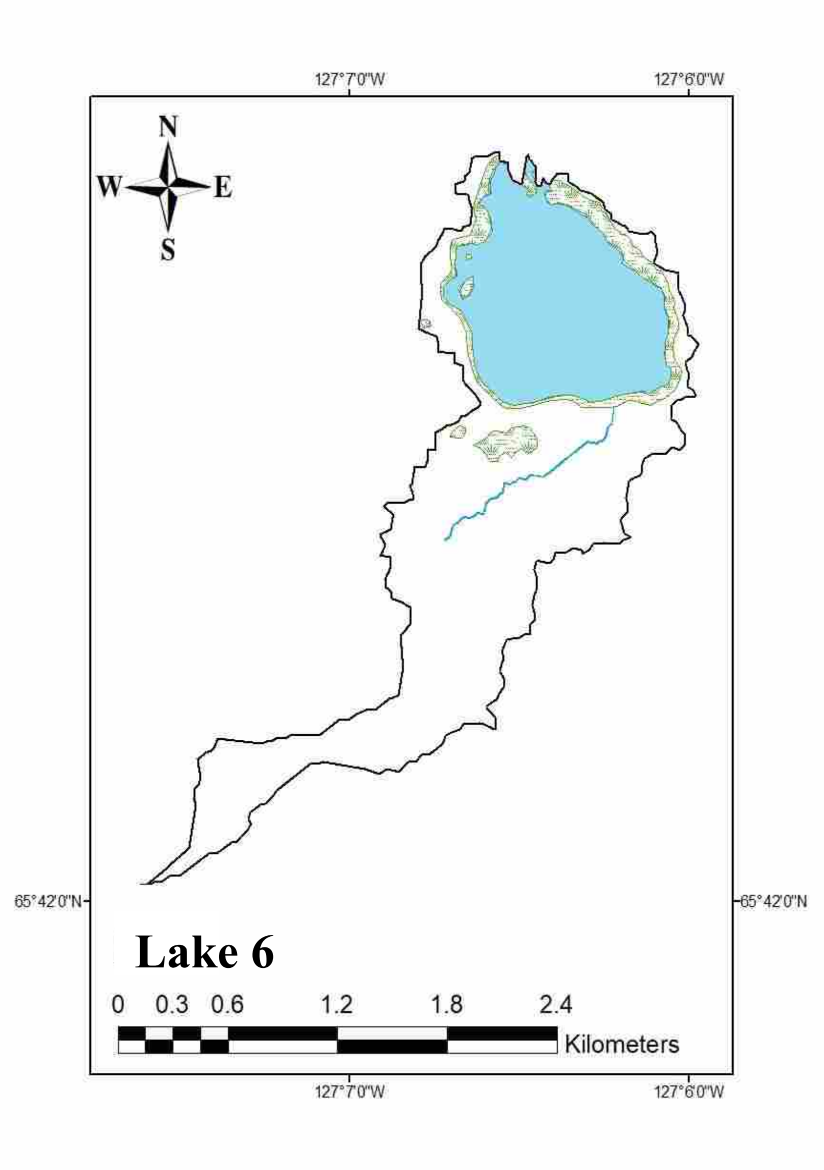

Figure S9: Lake 6 water catchment, hydrological inflow (blue outline), and wetland area (green outline).

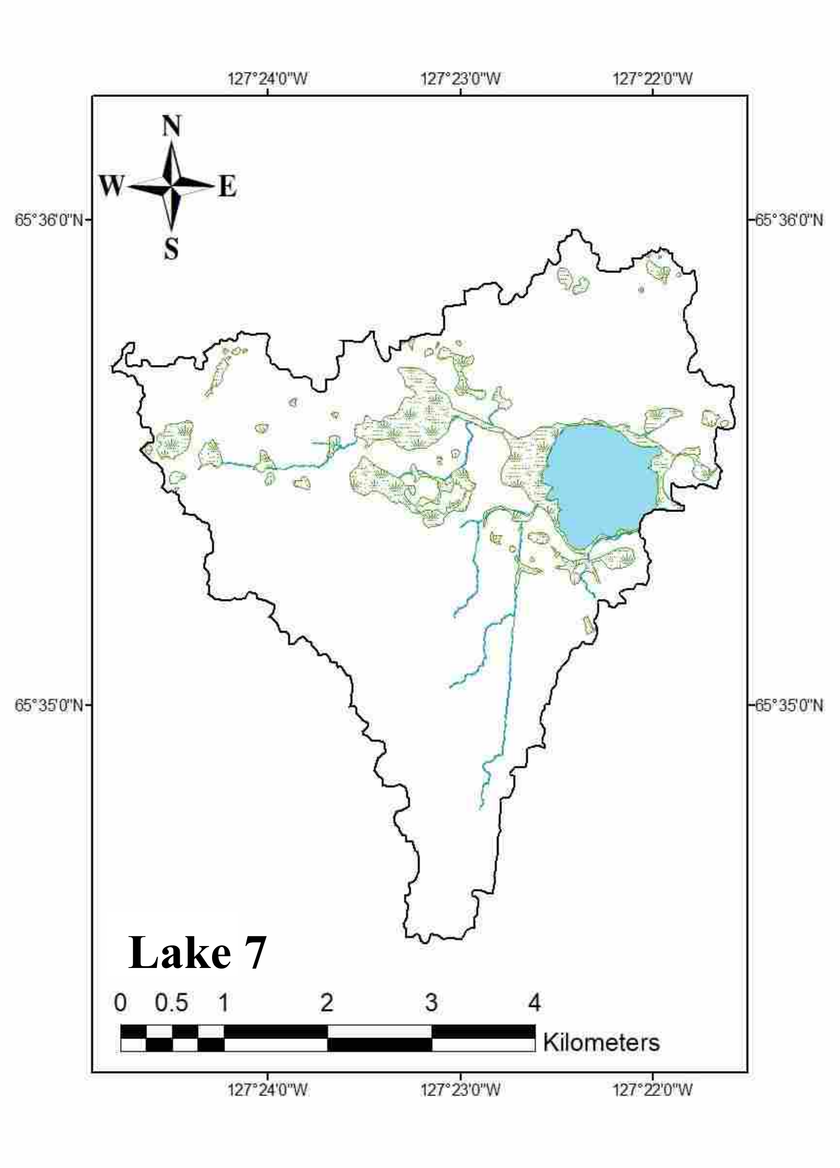

Figure S10: Lake 7 water catchment, hydrological inflow (blue outline), and wetland area (green outline).

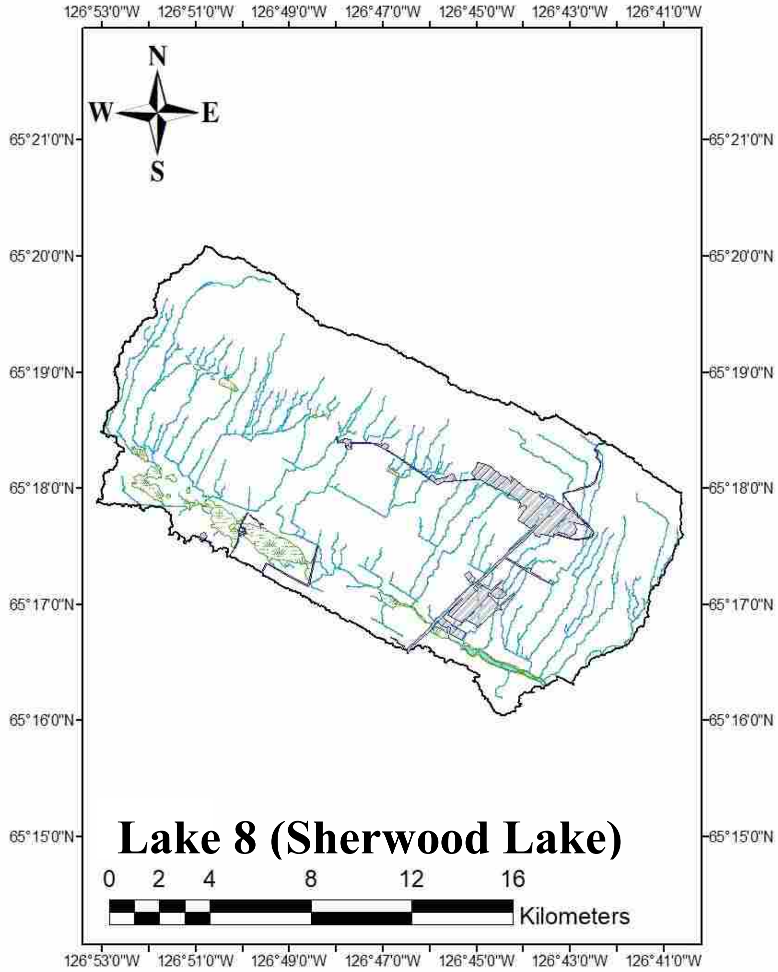

Figure S11: Lake 8 (Sherwood Lake) water catchment, hydrological inflow (blue outline), wetland area (green outline), and anthropogenic development (purple and grey outline).

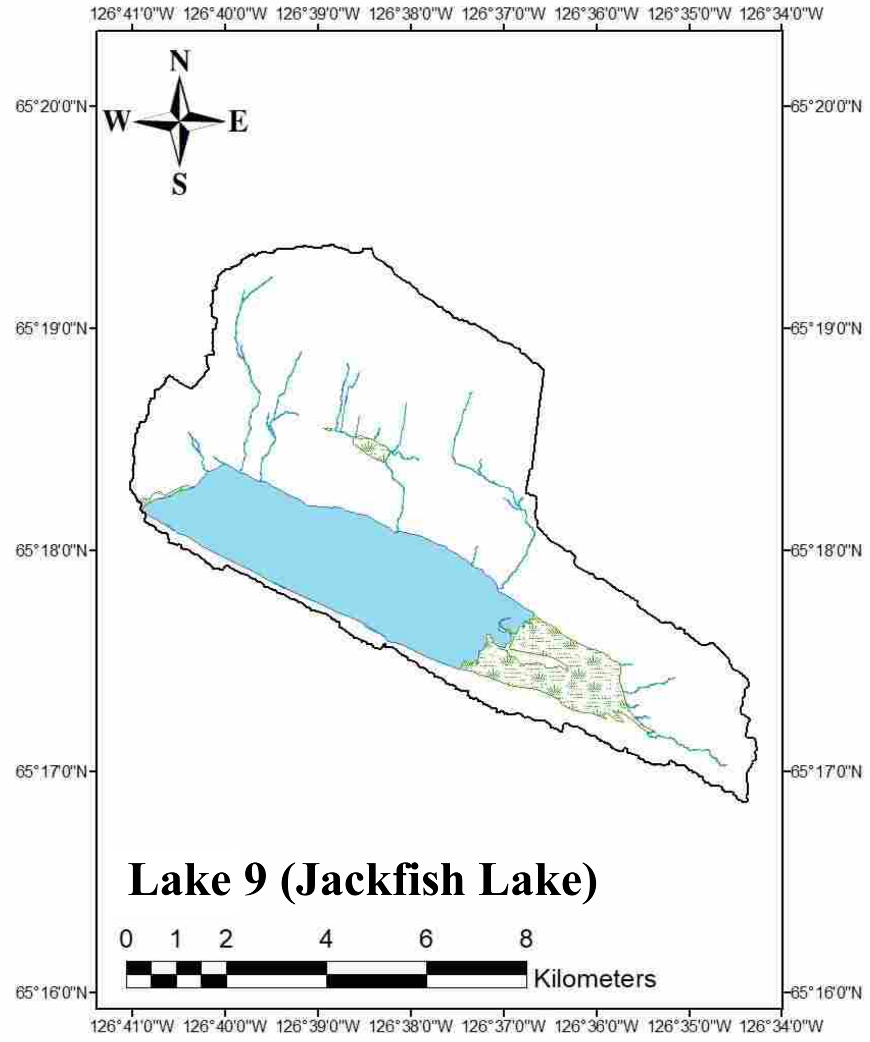

Figure S12: Lake 9 (Jackfish Lake) water catchment, hydrological inflow (blue outline), and wetland area (green outline).

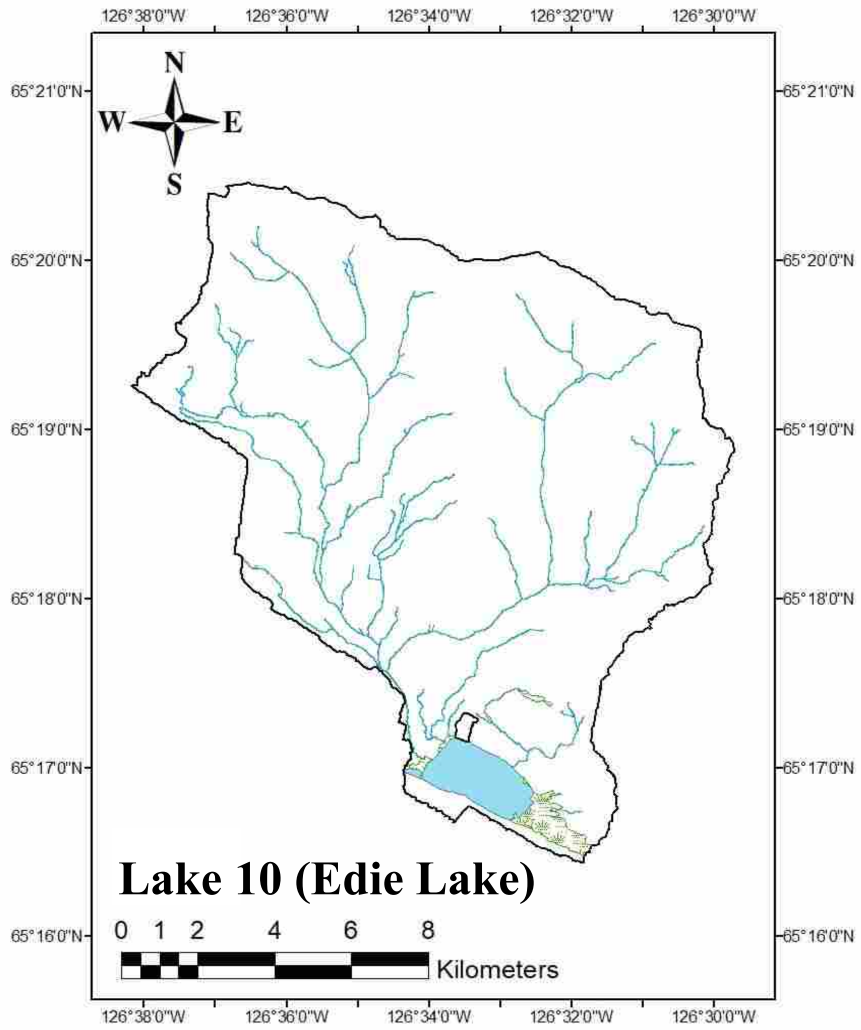

Figure S13: Lake 10 (Edie Lake) water catchment, hydrological inflow (blue outline), and wetland area (green outline).

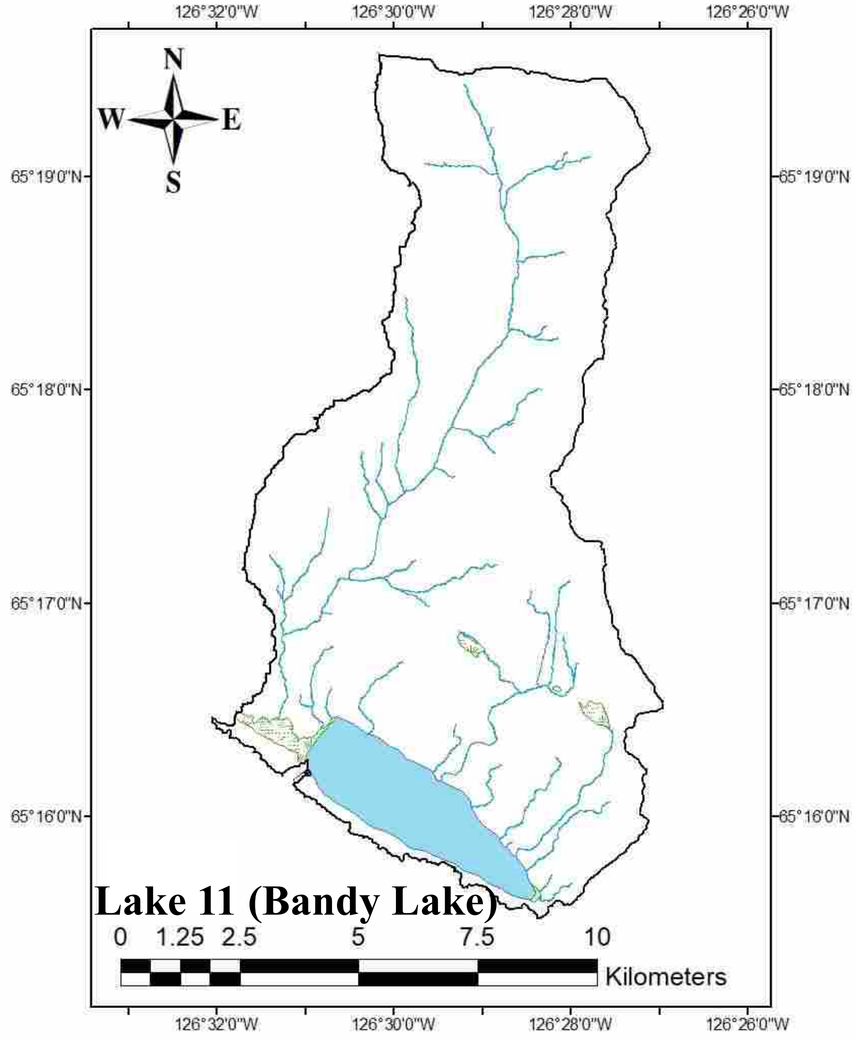

Figure S14: Lake 11 (Bandy Lake) water catchment, hydrological inflow (blue outline), and wetland area (green outline).

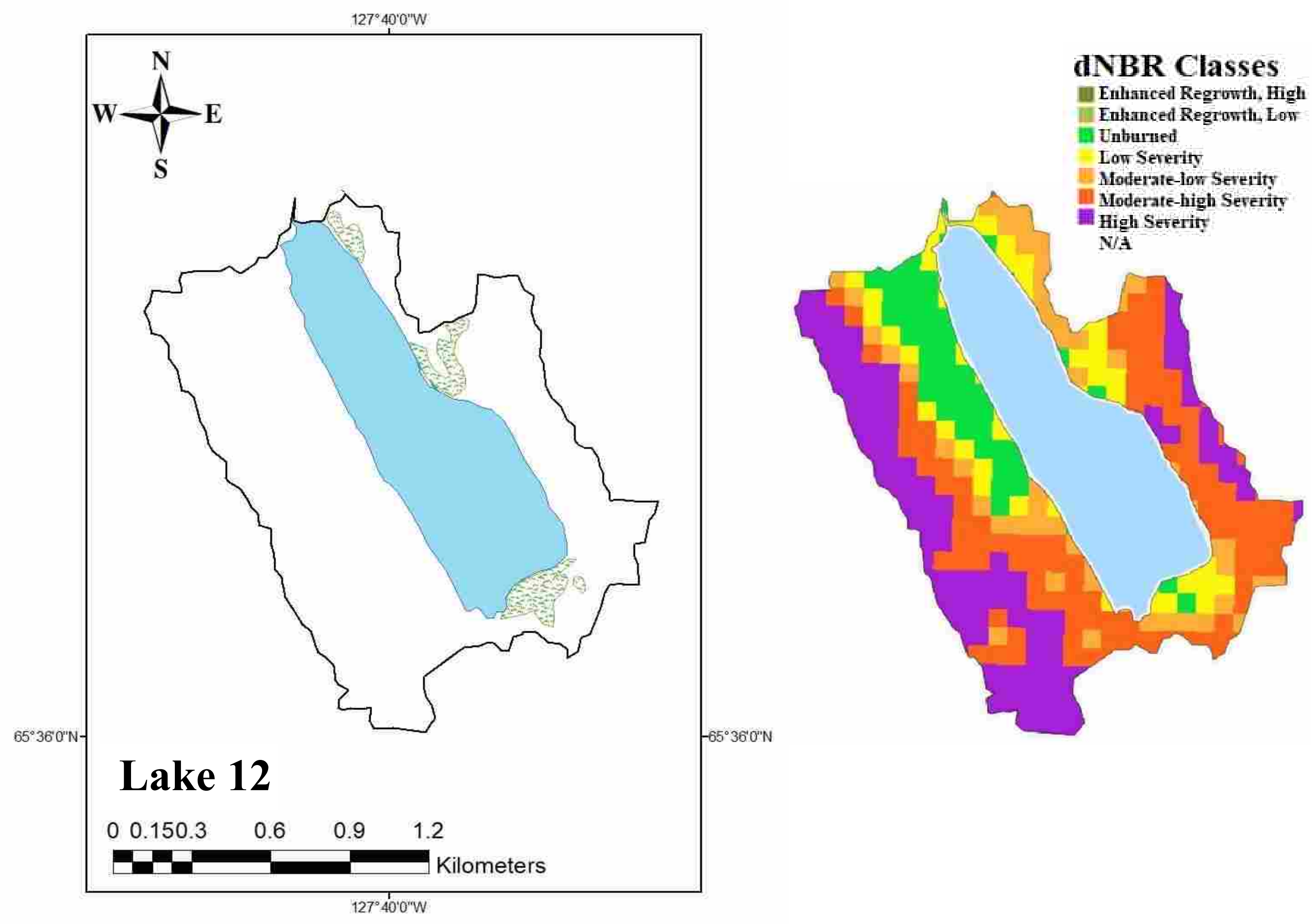

Figure S15: Lake 12 water catchment, hydrological inflow (blue outline), wetland area (green outline), and areas affected by fire scaring.

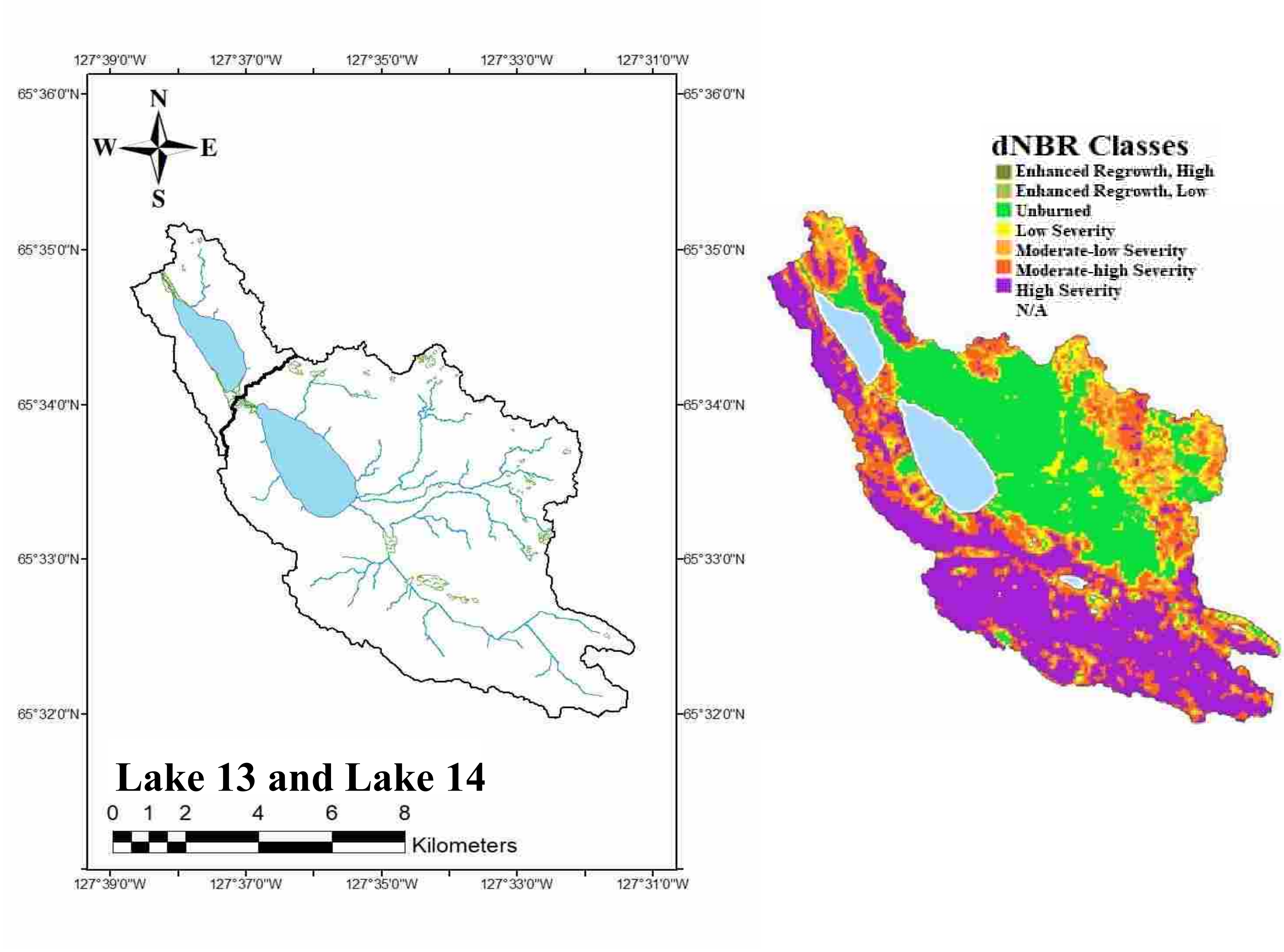

Figure S16: Lake 13 and Lake 14 water catchment, hydrological inflow (blue outline), wetland area (green outline), and areas affected by fire scaring.

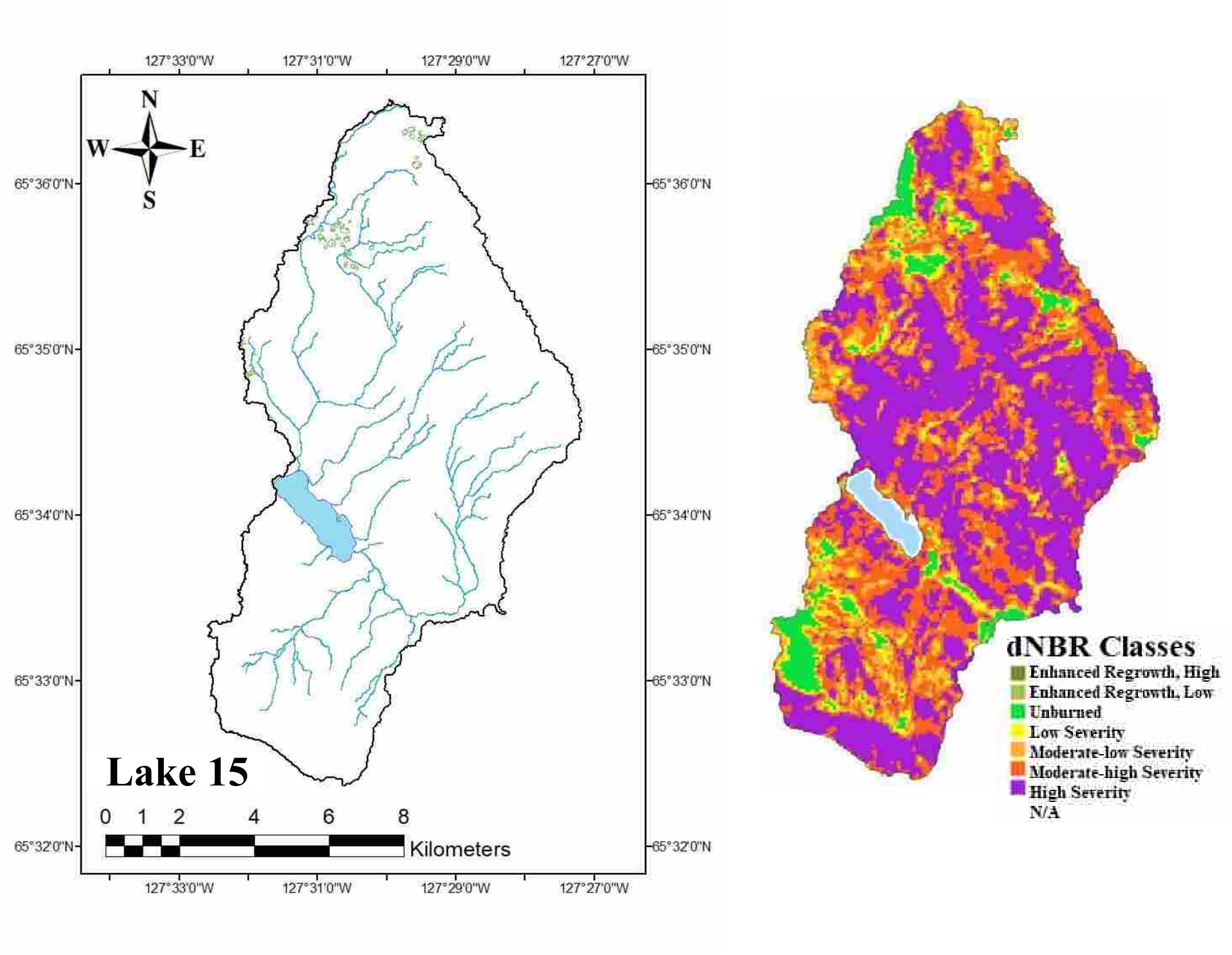

Figure S17: Lake 15 water catchment, hydrological inflow (blue outline), wetland area (green outline), and areas affected by fire scaring.

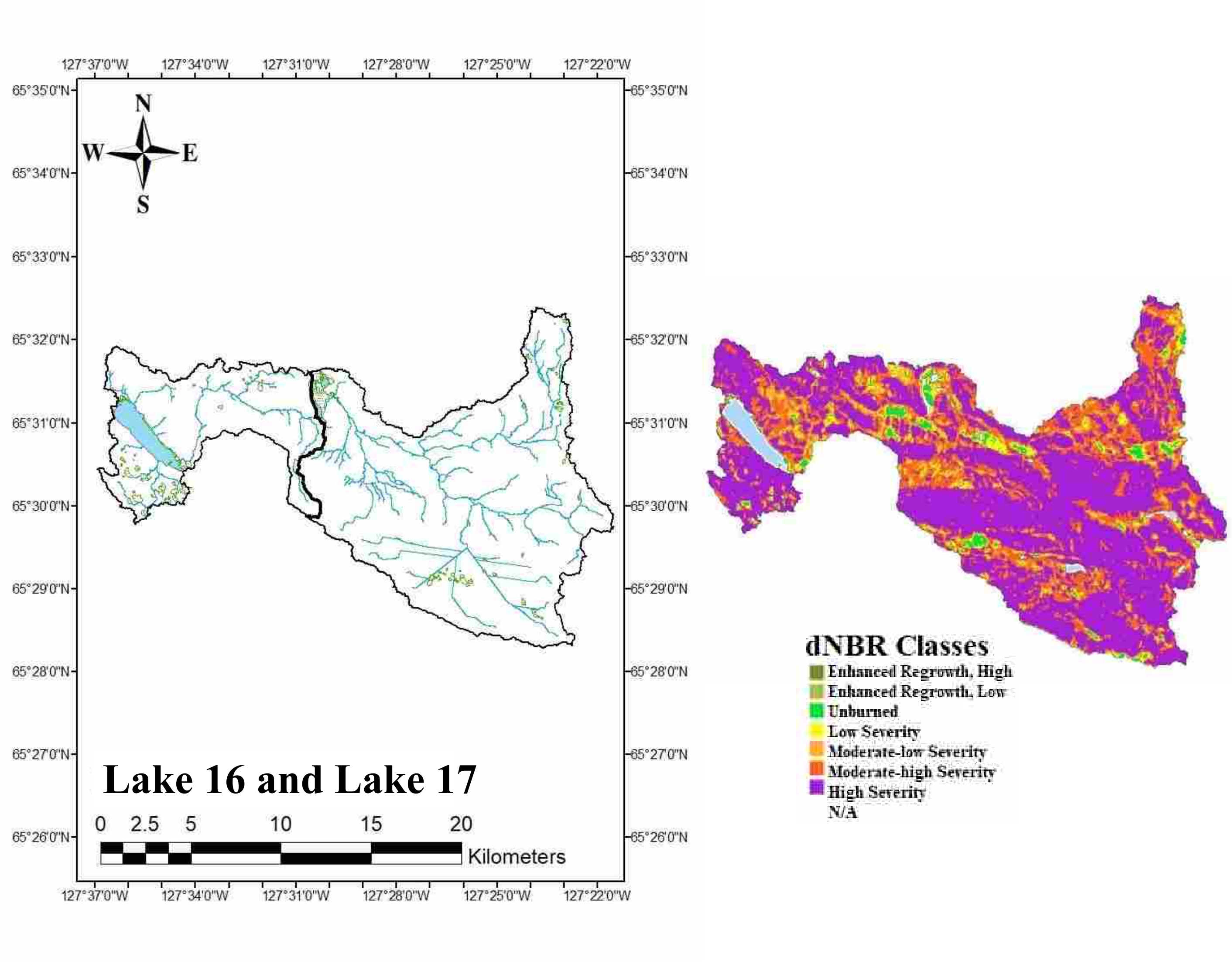

Figure S18: Lake 16 and Lake 17 water catchment, hydrological inflow (blue outline), wetland area (green outline), and areas affected by fire scaring.

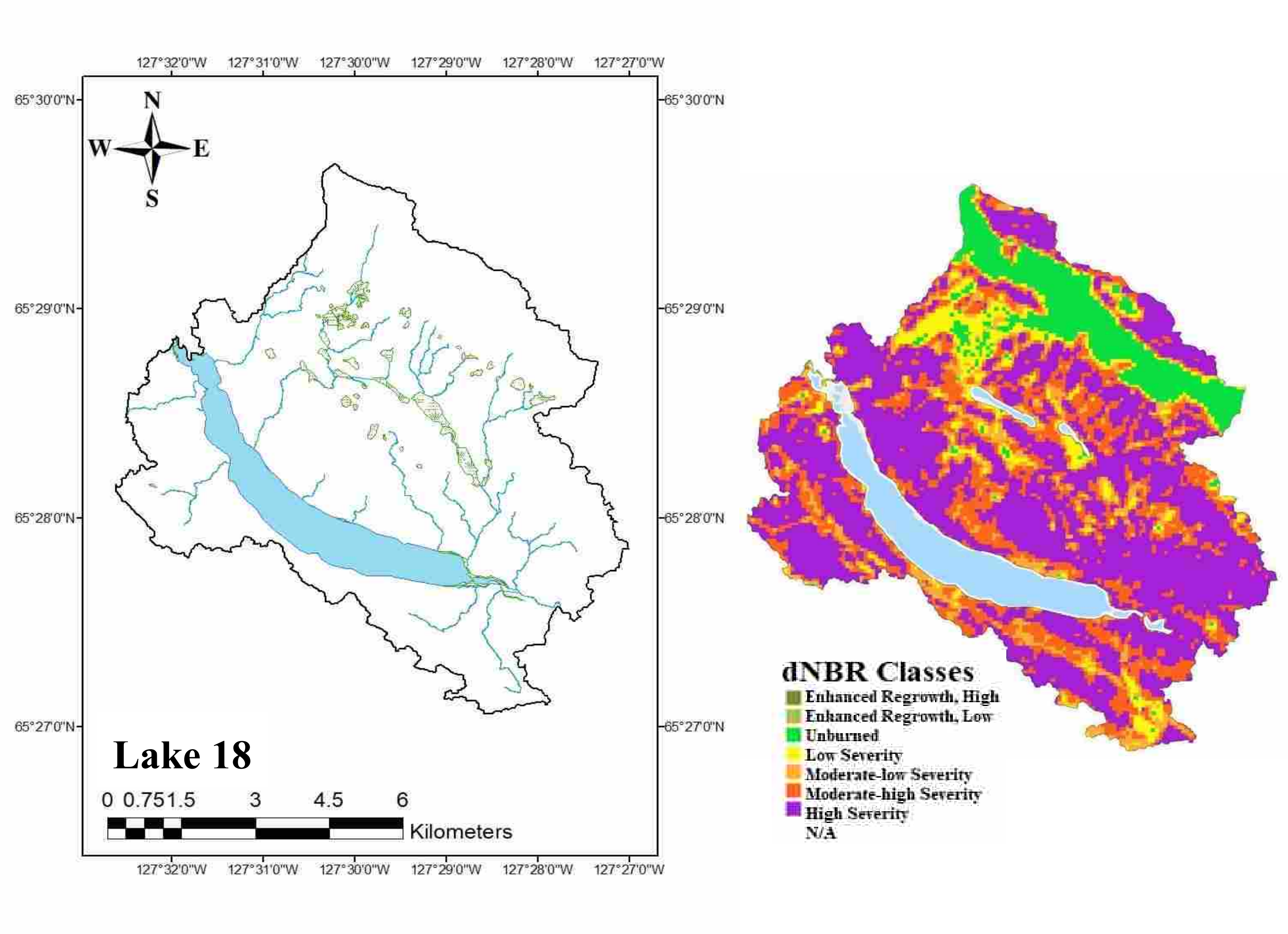

Figure S19: Lake 18 water catchment, hydrological inflow (blue outline), wetland area (green outline), and areas affected by fire scaring.

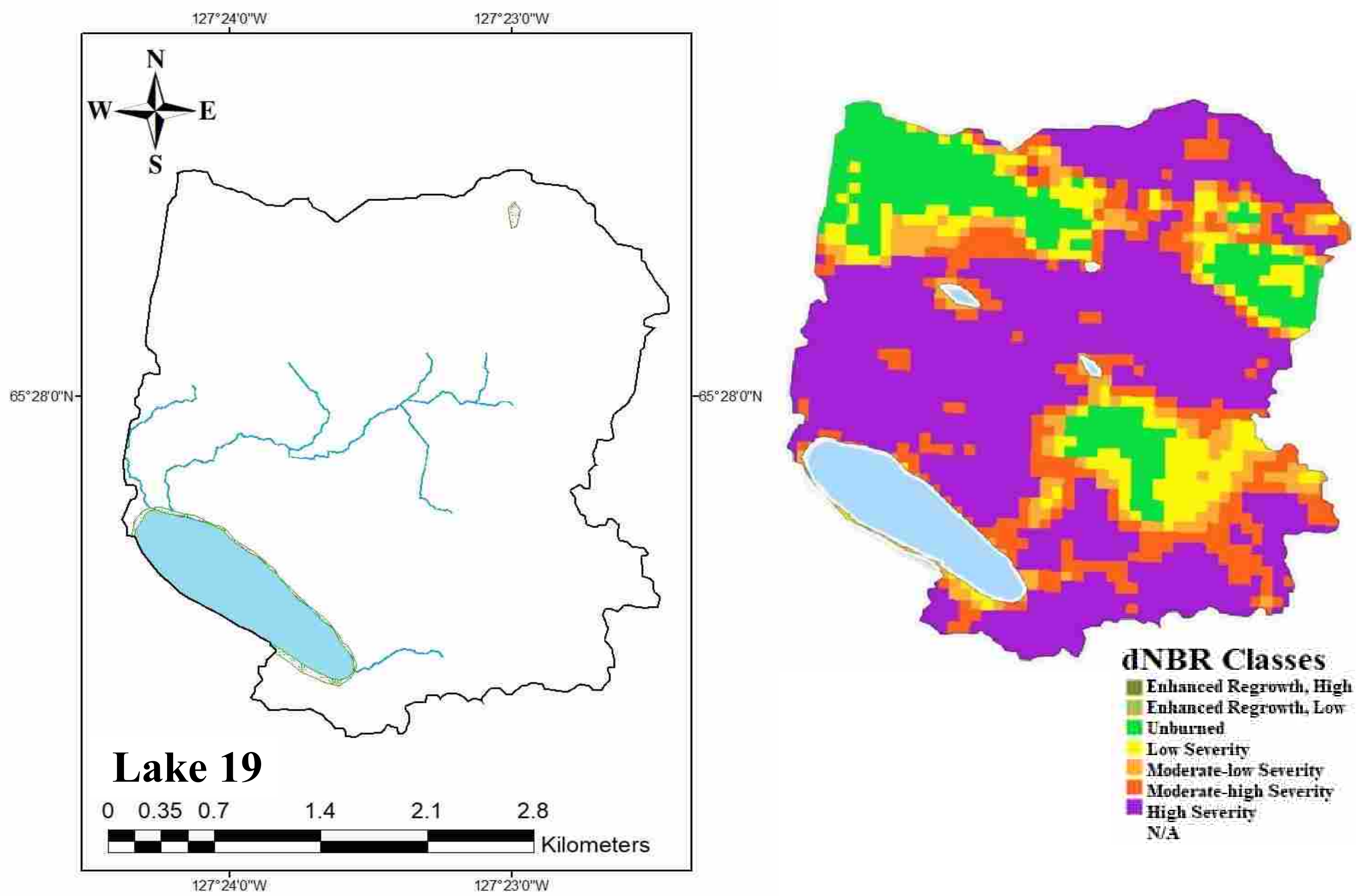

Figure S20: Lake 19 water catchment, hydrological inflow (blue outline), wetland area (green outline), and areas affected by fire scaring.

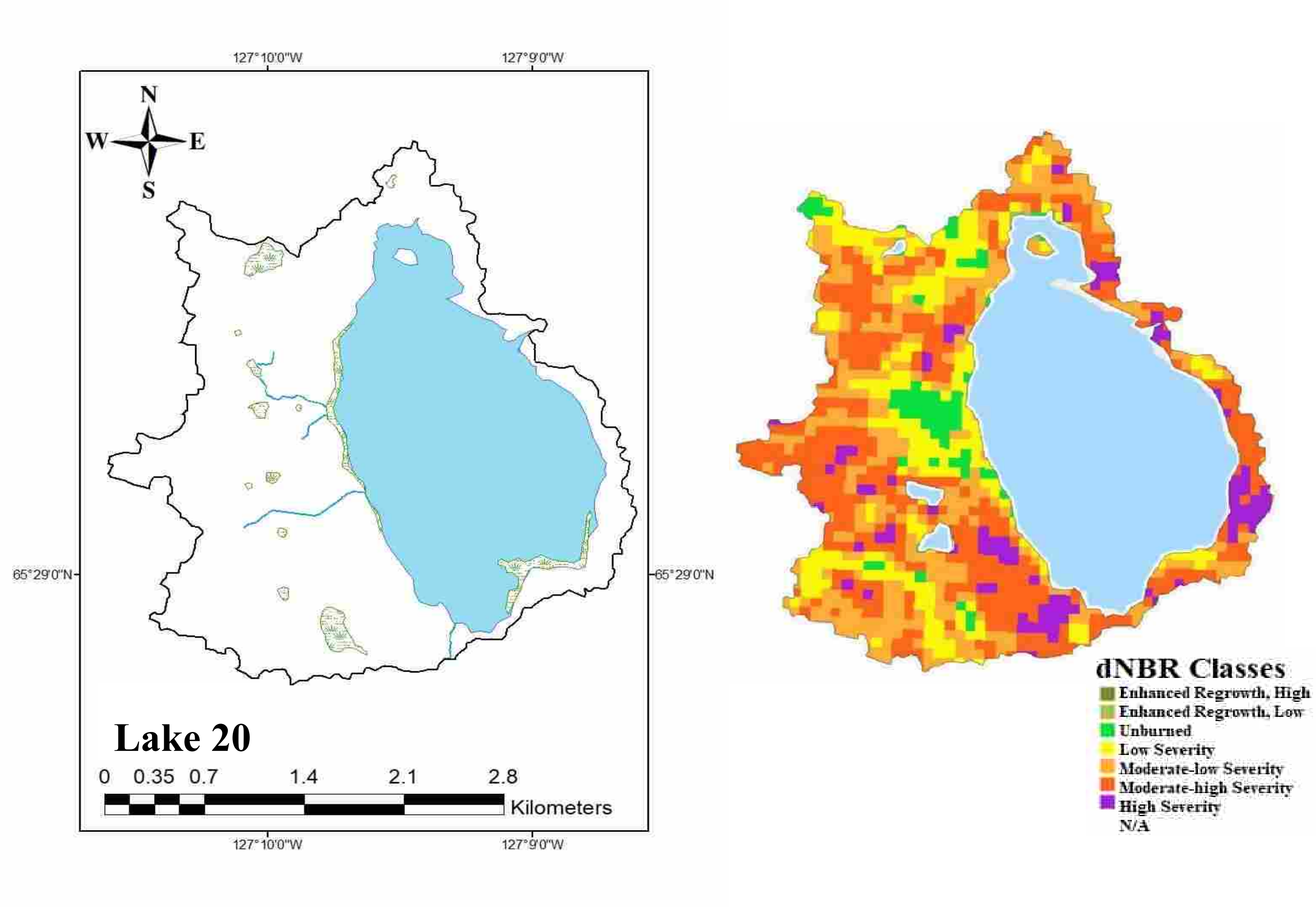

Figure S21: Lake 20 water catchment, hydrological inflow (blue outline), wetland area (green outline), and areas affected by fire scaring.