“When we went onto the land,
the elders taught me to not drink from a lake if there were no insects,
because that lake was dead.”

- Inuvialuit community member,
Personal communication, 2017

THE IMPACTS OF RECENT WILDFIRES ON STREAM WATER QUALITY AND MACROINVERTEBRATE ASSEMBLAGES IN SOUTHERN NORTHWEST TERRITORIES, CANADA

By

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Abstract

High-latitude regions are currently undergoing rapid ecosystem change due to increasing temperatures and modified precipitation regimes. Since 2012, the Northwest Territories (Canada) has been experiencing severe drought and wildfire seasons. In 2014 alone, fires within the Northwest Territories consumed over 3.4 million hectares of forested land; 1.4 times larger than the national yearly average for Canada. Wildfire is one of the most important agents influencing age structure and composition of forest stands, as such, it is a critical factor in ecosystem dynamics. The impacts of wildfire on terrestrial systems garner more attention compared to aquatic habitats. This is especially true when considering aquatic ecosystems, specifically sub-arctic streams, where the impact of fires on stream ecology and chemistry are relatively understudied. Freshwater ecosystems, such as lakes and streams, are relied upon by northern communities for their cultural significance and economic and environmental goods and services they produce, including country foods.

This study examines the impact of recent wildfire on freshwater streams within the North Slave, South Slave, and Dehcho regions of the Northwest Territories (Canada) through analysis of their water chemistry and benthic macroinvertebrate assemblages. Benthic macroinvertebrates, the macroscopic organisms living within/on the substrate of streams, were sampled following methodologies outlined by the Canadian Aquatic Biomonitoring Network (CABIN). Biological indices (e.g. Species Richness, Shannon Diversity Index) were calculated and compared statistically to determine relationships regarding benthic diversity and abundance. Results of this study suggest that recent wildfires cause short-term perturbations in water quality, such as increases in dissolved aluminum, TSS and turbidity. In addition, results indicate slight structural changes in invertebrate communities of streams within burned catchments (impacted) compared to those in unburned catchments (control), including increased richness and abundance of primary consumers and their predators.

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Table of Contents

Title Page…………………………………………………………………….. **i**
Abstract……………………………………………………………………. **ii**
Acknowledgements………………………………………………………… **iv**
Table of Contents…………………………………………………………... **v**
List of Tables………………………………………………………………. **vii**
List of Figures……………………………………………………………… **viii**
List of Acronyms…………………………………………………………. **x**

Chapter 1 **Introduction**……………………………………………………… **1**
- 1.1 Study Context……………………………………………………... **1**
- 1.2 Thesis Organization……………………………………………….. **5**

Chapter 2 **Literature Review**………………………………………………... **6**
- 2.1 The Climate and Environment of the Northwest Territories………… **6**
  - 2.1.1 Climate……………………………………………………….. **6**
  - 2.1.2 Vegetation……………………………………………………. **8**
  - 2.1.3 Glacial History……………………………………………….. **9**
  - 2.1.4 Geomorphic Setting (Permafrost)……………………………. **10**
- 2.2 Impacts of Climate Change on Disturbances within the Boreal Forest……………………………………………………………… **13**
- 2.3 Natural Drivers of Disturbance Regimes in the Boreal Forest Biome. **13**
- 2.4 Factors Influencing the Natural Variability of Streams……………... **19**
- 2.5 Impacts of Wildfire on Lotic Systems……………………………….. **20**
  - 2.5.1 Stream Morphology………………………………………….. **20**
  - 2.5.2 Stream Chemistry……………………………………………. **23**
  - 2.5.3 Stream Ecology………………………………………………. **29**
- 2.6 Methodological Approaches: CABIN Protocol……………………... **36**
- 2.7 References

Chapter 3 **The Impacts of Recent Wildfires on Stream Water Quality and Macroinvertebrate Assemblages in Southern Northwest Territories, Canada** **53**
- 3.1 Introduction………………………………………………………….. **53**
- 3.2 Description of Study Site……………………………………………. **56**
  - 3.2.1 Environmental Characteristics of the Study Regions………... **56**
  - 3.2.2 Recent Wildfires……………………………………………... **61**

3.3 Methodology………………………………………………………… **63**
  3.3.1 Site Selection……………………………………………… **63**
  3.3.2 Water Chemistry…………………………………………... **63**
  3.3.3 Benthic Macroinvertebrates……………………………….. **64**
  3.3.4 Macroinvertebrate Analysis……………………………….. **65**
  3.3.5 Data Preparation…………………………………………... **66**
  3.3.6 Data Analysis……………………………………………… **68**
3.4 Results……………………………………………………………….. **69**
  3.4.1 The Cameron River (CR-01), North Slave Region, NWT…... **69**
  3.4.2 The South Slave and Dehcho Regions, NWT……………….. **78**
3.5 Discussion…………………………………………………………… **89**
  3.5.1 Increases in Allochthonous Inputs May Prompt the Proliferation of First-order Consumers………………………. **89**
  3.5.2 Stream Nutrient Concentrations and Primary Productivity…... **92**
  3.5.3 Wildfire Impacts on the Abundance and Richness of Invertebrate Groups…………………………………………... **93**
3.6 Conclusion…………………………………………………………… **95**
3.7 References…………………………………………………………… **97**

Chapter 4 **Discussion and Conclusions** **102**

4.1 Changes in Water Quality…………………………………………… **102**
4.2 Changes in Trophic Relationships…………………………………… **105**
4.3 Research Limitations and Future Recommendations………………... **106**
4.4 References **108**

List of Appendices………………………………………………….. **109**
Appendix……………………………………………………………. **110**

List of Tables

2.1 Annual temperature and precipitation values for several communities within the Northwest Territories. Locations represent a latitudinal gradient with Inuvik and Fort Smith representing northern and southern climates, respectively. Data is for the period 1981-2010 (Environment Canada, 2017).

2.2 Various water quality parameters and their general response to wildfire.

2.3 Major functional groups found within freshwater ecosystems and their behaviors. Note that the example taxa included will exhibit these behaviors within their aquatic larval stages. Adult specimens will demonstrate alternate behaviors (Monterey Peninsula Water Management District, 2004; West Virginia Department of Environmental Protection, 2016).

3.1 Location, burn status, and general catchment characteristics for streams included within this study.

3.2 Attributes of fires included within this study, local climate conditions near the time of their detection and 30-year climate normals. Information regarding fires is summarized from statements issued by the Government of the Northwest Territories on its website and social media accounts. Climate data is acquired from the nearest Environment Canada climate station with climate normals available for the period 1981-2010.

3.3 Pettitt's change point test statistics for Cameron River historical water quality data. No significant changes are detected in the data during or after the Reid Lake fire (July 2016).

3.4 Pearson correlation coefficients ($\alpha \leq 0.05$) between years since fire/size of fire and water quality variables for streams located in the Dehcho and South Slave regions, NWT. Only the significant correlation coefficients were included. P values are included in brackets.

3.5 Taxonomic classifications, functional feeding methods, and descriptive statistics for families demonstrating significant differences in abundance between control and impacted streams.

3.6 Pearson correlation coefficients for functional feeding groups and fire size. Only significant ($p \leq 0.05$) are presented below.

4.1 Water quality variables and their associated drinking water guidelines (Health Canada, 2017). Parameters that were found to be the most different between control and impacted streams are bolded.

List of Figures

2.1 The distribution of permafrost zones across Canada (Smith, 2011). Modified from Heginbottom et al. (1995).

2.2 Relationships between climate change and natural and anthropogenic factors influencing wildfire (Herawati et al., 2015).

2.3 Major macroinvertebrate taxa and their relative tolerances to declining water quality. Freshwater systems with high dissolved oxygen, high pH, cooler temperatures, and suitable habitat may contain all major groups of taxa. Impaired streams and lakes will contain macroinvertebrate groups tolerant to lower dissolved oxygen, pH, and warmer temperatures (namely leeches, worms, snails, and certain fly larvae).

2.4 Conceptual model of the potential direct impacts of wildfire on benthic invertebrate taxa. Declines in overall water quality are the product of sediment input and changes in water temperature, which occur through the processes of combustion, overland flow, and atmospheric deposition.

2.5 Methodology for in-stream sampling of benthic macroinvertebrates. Sampling commences downstream within the riffle, and continues upstream in a zig-zag fashion for a maximum of 3 minutes to standardize sampling effort.

3.1 Map of the study region in Southern Northwest Territories.

3.2 Changes in TDS, $_{\text{dissolved}}$Al, and turbidity relative to the Reid Lake fire (vertical red line). TDS and $_{\text{dissolved}}$Al increase immediately following the fire, while an increase in turbidity followed a large precipitation event in September 2016

3.3 Comparison of relative abundance and richness for major taxa groups present in the Cameron River between 2015 and 2016.

3.4 Comparison of relative abundance and richness for functional feeding groups present in the Cameron River between 2015 and 2016. The ‘Other’ category includes Macrophyte-Herbivores, Omnivores, Piercers, and unclassified taxa.

3.5 Box and whisker plots illustrating the means and standard deviation of water quality parameters between control and impacted streams. The mean for each treatment is indicated by the red crosshair. The horizontal line within the box represents the median value while the edges of the box are the upper and lower quartiles. The whiskers represent the highest and lowest values, respectively. Outliers are represented by small circles. Only Alkalinity and $_{\text{dissolved}}$Al are found to be significantly different between impacted and control streams.

3.6 Principal component plot illustrating correlations between abundance measures and water chemistry variables for streams located within the Dehcho and South Slave regions. Strong associations are observed among turbidity, TSS, and total species abundance. There is no clear separation of sites amongst either the primary or the secondary axis. Control and impacted sites are represented by black circles and red triangles, respectively.

3.7 Principal component plot illustrating correlations between richness measures and water chemistry variables for streams located within the Dehcho and South Slave regions. There is no clear separation of sites amongst either the primary or the secondary axis. Control and impacted sites are represented by black circles and red triangles, respectively.

List of Acronyms

Al	Aluminum
ANOVA	Analysis of variance
As	Arsenic
Au	Gold
Be	Beryllium
BOD	Biological oxygen demand
cA	Continental Arctic air mass
Ca	Calcium
CABIN	Canadian Aquatic Biomonitoring Network
$CaSO_4 \bullet 2H2O$	Gypsum
Cl	Chloride
CN^-	Cyanide
CO_2	Carbon dioxide
CPOM	Coarse particulate organic matter
Cu	Copper
DFA	Discriminant function analyses
DO	Dissolved oxygen
DOC	Dissolved organic carbon
DON	Dissolved organic nitrogen
EPT	Ephemeroptera+Plecoptera+Tricoptera
Fe	Iron
FFG	Functional feeding groups
FPOM	Fine particulate organic matter
HCO_3^-	Bicarbonate
HNO_3	Nitric acid
K	Potassium
Li	Lithium
LWD	Large woody debris
Mg	Magnesium
Mn	Manganese
mP	Maritime polar air mass
Na	Sodium
NH_3	Ammonia
NH_4^+	Ammonium
NO_3^-	Nitrate
NO_3+NO_2_N	Nitrate and nitrite as nitrogen
NTU	Nephelometric turbidity unit
Pb	Lead
PCA	Principal component analysis
PO_4^{3-}	Phosphate
PP	Primary producers
PyC	Pyrogenic carbon
RA	Relative abundance
RCA	Reference condition approach

Si	Silica
SO_4^{2-}	Sulphate
TDS	Total dissolved solids
TOC	Total organic carbon
TP	Total phosphorous
TSS	Total suspended solids
yBP	Years before present
Zn	Zinc

Chapter One

Introduction

1.1 Study Context

Changes in climate factors (e.g. increasing temperatures) are impacting natural systems, especially those located at higher latitudes (ACIA, 2005). Increasing temperatures and changing precipitation regimes due to anthropogenic-induced climate change are expected to alter the distribution of vegetation zones and the frequency and severity of disturbance events, such as wildfire, in higher latitude regions (McCoy and Burn, 2005). Across Canada's boreal region, the frequency and intensity of wildfires is on the rise, with current levels of total area burned exceeding the 10-year national average for the period 2004-2014 (Natural Resources Canada, 2015).

Wildfires are a typical feature of Canada's boreal region due to the high amounts of insolation received as a result of long days and highly combustible organic litter produced from the canopy (Perera and Buse, 2014). Wildfire is also one of the most important agents influencing age structure and composition of forests. Therefore, it is a critical factor in ecosystem dynamics (Rapanoela et al., 2015).

The Northwest Territories are renowned for their water-rich landscapes. Lakes, rivers, and streams make up over 13% of the territories' total area. These resources are highly valued for their intrinsic and spiritual properties and are celebrated by indigenous communities for their life-giving attributes. For example, community members in Kakisa (NWT), a town affected by wildfires in 2014 and 2015, describe their livelihood in the following manner:

> "This deeply traditional outpost is set amongst blazing fireweed and towering timber evergreens on the shores of vast Kakisa Lake, just beside the source of the Kakisa River. We make our living from these waters and the surrounding woods, fishing for pickerel, trapping, and hunting" (Northwest Territories Tourism, 2016)

From a systematic approach, these ecosystems provide numerous amenities that can be divided into provisioning, supporting, regulating, and cultural services, according to the benefits derived from them. In the context of aquatic ecosystems, provisioning services include the production of consumable or economic goods such as food and freshwater. Regulating services are processes which humans and other organisms derive benefits from, such as water filtration, flood control, and climate regulation (Millenium Ecosystem Assessment, 2005). Cultural services include intangible benefits that are dependent on individual values and beliefs. Finally, supporting services can be identified as processes that support the continuous production of all ecosystem services, including photosynthesis and nutrient cycling (Millenium Ecosystem Assessment, 2005). The ability for these services to continuously provide goods and benefits can be temporarily or permanently impacted by disturbances (Blumstein and Thompson, 2015). In addition, understanding how the quality of freshwater resources is affected by perturbations in the environment is important as declining water quality resulting from environmental change can potentially impact community health. For instance, increased sediment input to rivers and lakes resulting from increased erosion in the surrounding catchment may decrease the effectiveness of water treatment processes (Emelko and Sham, 2014). Monitoring these ecosystems is warranted in order to 1) mitigate potential ecosystem impacts, and 2) maintain the benefits derived from their services.

Northwest Territories, Canada, contains approximately 800,000 km^2 of forest, in addition to fresh water resources including streams, rivers and lakes. In recent years (2012 to current), many regions within the Northwest Territories have been experiencing drought. Lower-than-average precipitation levels, combined with warmer air temperatures, are thought to have largely contributed to the severe wildfire season of 2014 (Olesinski and Brett, 2016). This was the most destructive fire season in recent record (1970-2015) in terms of total area burned (National Forestry Database, 2017). The lack of precipitation also contributed to low water levels and discharge in watercourses across the territory. Water levels dropped so low that, in areas such as the Snare River basin, yields from hydroelectric generating stations were significantly reduced (Northwest Territories Power Corporation, 2015).

In the wake of these recent environmental changes, federal agencies have acknowledged the need for increased hydrological monitoring and data collection (Government of the Northwest Territories, 2016). Currently, there are a number of hydrological monitoring stations and community-based monitoring programs across the Northwest Territories that provide historical and real-time hydrometric and water quality data including discharge, water level, and sediment loading. However, there is a lack of data and research available focused on changes in ecological structure of streams and lakes in response to recent disturbances such as drought and wildfire.

Benthic macroinvertebrates (small bodied organisms living within or upon the substrate of freshwater ecosystems) are commonly employed as bio-indicators of ecosystem change in freshwater monitoring studies. Benthic invertebrates are advantageous in this regard as their collection and processing can be inexpensive and they are often abundant among freshwater ecosystems. In the Northwest Territories, benthic invertebrates have been used as indicators of change in aquatic settings to evaluate the impacts of mining activities (Thomas and Heath, 2012), retrogressive thaw slumps (Mesquita et al., 2008; Chin et al., 2015), mercury bioaccumulation (Swanson, 2016), and oil pipeline installation (Young and Mackie, 1991), as well as for various other water-quality monitoring projects.

This study analyzes benthic invertebrate assemblages from control and impacted stream catchments within the North Slave, South Slave, and Dehcho regions of the Northwest Territories. The objective of this thesis is to determine the impacts of recent wildfires on benthic invertebrate communities and water quality in streams located in the southern regions of the Northwest Territories. Specifically, this study seeks to determine if there are major differences in the community composition and trophic structure of streams within burned catchments (impacted) versus non-burned catchments (control). In addition, this thesis analyzes the recovery of water quality in the Cameron River following a large fire near Reid Lake in 2016. This work represents the first study that analyzes wildfire impacts on both the water quality and ecology of sub-Arctic streams using benthic macroinvertebrates for the North Slave, South Slave, and Dehcho regions of the Northwest Territories. Understanding how common disturbances, such as wildfire, affect freshwater ecosystems is important since the frequency and severity of wildfire are expected to increase due to changes in global climate change.

1.2 Thesis Organization

This thesis is divided into four chapters. The first chapter offers a brief introduction to climate change, and its anticipated and observed impacts on high latitude ecosystems. This chapter also includes a description of recent climatic changes within the study area, and how benthic macroinvertebrates have been used in similar studies within the region. Chapter one provides context for the research question, which is included at the end of chapter one. Chapter two is a literature review of available information related to this research and is broken up into six sections: 1) the climate and environment of the Northwest Territories, 2) impacts of climate change on disturbances within the boreal forest, 3) drivers of disturbance regimes in the boreal forest biome, 4) factors influencing the natural variability of streams, 5) impacts of wildfire on lotic systems, and 6) methodological approaches: CABIN protocol. Chapter three is presented as a stand-alone manuscript and includes detailed descriptions of the study area, methodologies employed in this research, results of this study, and discussion of results. Chapter four includes a broader discussion of this research, including the applications of this work.

Chapter Two

Literature Review

2.1 The Climate and Environment of the Northwest Territories

2.1.1 Climate

Climate across the Northwest Territories is highly variable. Currently, there are 18 recognized level III ecoprovinces/ecozones in the Northwest Territories, which are defined according to regional climate patterns. Two principal factors affecting regional climates include latitude, and regional and global circulation patterns (Ecosystem Classification Group, 2012).

During the winter months, the Canadian climate is largely influenced by the presence of the continental Arctic air mass (cA). This air mass is characterized as being both extremely dry and cold as a result of forming over ice/snow-covered surfaces in Polar regions (Ahrens et al., 2012). As the winter season transitions to spring, decay of the cA air mass over much of the northern territories and Prairie provinces becomes evident as warmer maritime polar air masses (mP) and low-pressure systems from the northwest and Pacific bring greater precipitation and warmer temperatures (Klock et al., 2001). The Northwest Territories experiences cool and wet summers as a result of the retreat of the dry cA air mass and movement of the moist mP air mass further inland (Klock et al., 2001; Ahrens et al., 2012). Annual temperature and precipitation values for the Northwest Territories vary based on factors including latitude, altitude, and topography. Generally, northern regions exhibit cooler and drier regional climates (Table 2.1)

Table 2.1. Annual temperature and precipitation values for several communities within the Northwest Territories. Locations represent a latitudinal gradient with Inuvik and Fort Smith representing northern and southern climates, respectively. Data is for the period 1981-2010 (Environment Canada, 2017).

Location	Latitude (°)	Longitude (°)	Elevation (mASL)	Mean Annual Temperature C°	Total Annual Precipitation (mm)
Inuvik	68.3607	-133.723022	59	-8.2	240.6
Norman Wells	65.2815	-126.837158	65	-5.1	294.4
Yellowknife	65.4540	-114.371788	202	-4.3	288.6
Fort Smith	60.0055	-110.614723	204	-1.8	353.6

All stream sites considered within this study are located within the Taiga Shield High boreal ecoregion (Level IV Ecoregion) of the Northwest Territories. Climate records for this ecoregion are limited; therefore, only general statements can be made about the climate of the broader region. This region generally experiences cool, wet summers with colder, and drier winters (Ecosystem Classification Group, 2008). A lack of precipitation in the winter months is partially due to the positioning of the arid continental Arctic air mass (cA), whereas increased precipitation in the summer months is produced mainly from the passage of cyclonic fronts (Prowse et al., 2009).

2.1.2 Vegetation

The distribution of vegetation types is affected by large scale and regional factors. The boundaries of vegetation zones, or biomes, are predominately determined by moisture availability, temperature, and solar radiation levels (Smith and Smith, 2012). Therefore, the location of biomes reflects the physiological requirements of the plant species that constitute each zone. The landscape of the Northwest Territories can be divided into two biomes (~ analogous to Level I Ecoregions); boreal/taiga forest and tundra.

The boreal/taiga forest, the largest global biome, is characterized by cold-tolerant tree taxa such as black spruce (*Picea mariana)*, white spruce (*Picea glauca)*, pine (*Pinus* spp.), fir (*Abies* spp.), tamarack (*Larix* spp.), poplar (*Populus* spp.), and birch (*Betula* spp.) (Smith and Smith, 2012; Brandt et al., 2013). Shrubs such as willow (*Salix* spp.), Alder (*Alnus spp.*), and heath (*Vaccinium* spp.) along with lichen (*Cladinia spp., Cetraria spp.*), and mosses (*Polytrichium spp., Sphagnum spp.*) are typically present within the understory (Ecosystem Classification Group, 2008). With increasing latitude (and altitude), the closed-crown canopies of the boreal forest give way to lichen woodland. This ecozone is dominated by lichens, including the genus *Cladonia sp.,* and black spruce, which typically make up less than 40% of the total vegetation (Payette, 1992). Plant communities diverse with shrubs and herbaceous species can also be found in areas of the boreal biome where topography or atypical climate conditions result in cooler, and drier temperatures (e.g. alpine settings) (Brandt, 2009). Separating the closed canopy forest of the boreal/taiga and the lichen-rich tundra biome is the forest-tundra ecotone. This transitional ecotone is geographically defined as being 'located south of the treeline and north of the southern limit of upland tundra' (Timoney et al., 1992). Trees growing within this zone may display stunted growth patterns (e.g., krummholz).

The tundra biome exists north of the treeline and encompasses the southern Arctic, and tundra cordillera ecoregions (Level II ecosystems). This biome is characterized by dwarf-shrubs including various shrubs of the heath family (e.g., *Vaccinium spp.),* willow, birch and alder, sedges (Cyperaceae), and bryophytes (Ecosystem Classification Group, 2012). Generally, plant diversity of the tundra is low. Those species that exist at these high latitudes are tolerant to geomorphic disturbances (e.g., cryoplanation, erosion, thermokarst processes), low moisture levels, low temperatures, and reduced incoming solar radiation during the winter months (Smith and Smith, 2012).

2.1.3 Glacial History

North America has experienced several glacial periods during the past 2.5 million years. The last glacial period, also known as the Wisconsinan glaciation, occurred between 80,000-10,000 yBP. During this period, three ice-sheet complexes covered most of Canada and the northern United States. The Northwest Territories was overlain by the largest of the ice-sheets, the Laurentide ice sheet complex (Keewatin sector), in addition to the Innuitian ice sheet in the far north of the Canadian Archipelago (Dyke, 2004). The landscape of the Northwest Territories is marked by a variety of glacial landforms as a result, including moraines, drumlins, and eskers. Some regions, including the Mackenzie River Delta, overlay vast deposits of glacial sediments (e.g., till) that originate from glacial ice advance and retreat (Smith, 1992; Ecosystem Classification Group, 2008). In other areas, thick deposits of lacustrine sediments (e.g., silts and clays) result from the formation of proglacial lakes such as glacial Lake McConnell and Lake Mackenzie (Smith, 1992). It is currently thought that both Great Bear Lake and Great Slave Lake are the remnants of glacial Lake McConnell, which at its maximum extent was estimated to be 210,000 km^2 in size (Smith, 1994).

2.1.4 Geomorphic Setting (Permafrost)

Permafrost (ground material that remains at or below 0°C for two or more consecutive years) is an important component of the global cryosphere (Arctic Climate Impact Assessment (ACIA), 2005). Permafrost distribution is typically characterized by its presence, and is broadly defined as being continuous (100%-90%), discontinuous (90%-50%), and sporadic (50%>) (Fig. 2.1; International Permafrost Association, 2015). Examples of landforms associated with permafrost include palsas, lithalsas, pingos, and polygonal terrain (Yoshikawa and Hinzman, 2003). The presence of underlying permafrost can affect the process of pedogenesis, resulting in the formation of cryosols or gelisols (United States Department of Agriculture, 1999). In Northwest Territories, cryosols are the dominant soils found in the tundra shield, taiga plains subArctic, and taiga shield subArctic ecoregions (Ecosystem Classification Group, 2007; 2008; 2012).

There are a multitude of factors that determine the spatial distribution of permafrost at a regional scale. These factors include climate, local geology, and vegetation (Osterkamp and Jorgenson, 2009). It is currently accepted that increasing global temperatures have increased the rate and spatial distribution of permafrost thaw in high-latitude regions (IPCC, 2014). Thawing permafrost has been attributed to increases in stream flow, thermokarst topography (retrogressive thaw slumps, subsidence etc.), changes in vegetation cover (paludification) and surface water, and the release of stored/regulation carbon (Callaghan et al., 2011; Kokelj and Jorgenson, 2013; Baltzer et al., 2014; Connon et al., 2014). Thawing permafrost releases greenhouse gasses, such as carbon dioxide and methane, which contribute to increases in global temperature via the greenhouse effect. Furthermore, this accelerates permafrost degradation and is referred to as the permafrost carbon feedback system (Schaefer et al., 2014).

The development of mines, hydroelectric facilities, roads, and other forms of infrastructure often lead to increases in ground temperature in both continuous and discontinuous permafrost zones due to disturbance of the active layer (Northwest Territories Environmental Audit, 2005). In the subArctic regions of Canada, wildfires have been found to increase active layer thickness and rates of permafrost thaw through removal of soil organic layers. These impacts are more pronounced in forest ecosystems compared to either the tundra or peatlands (Zhang et al., 2015).

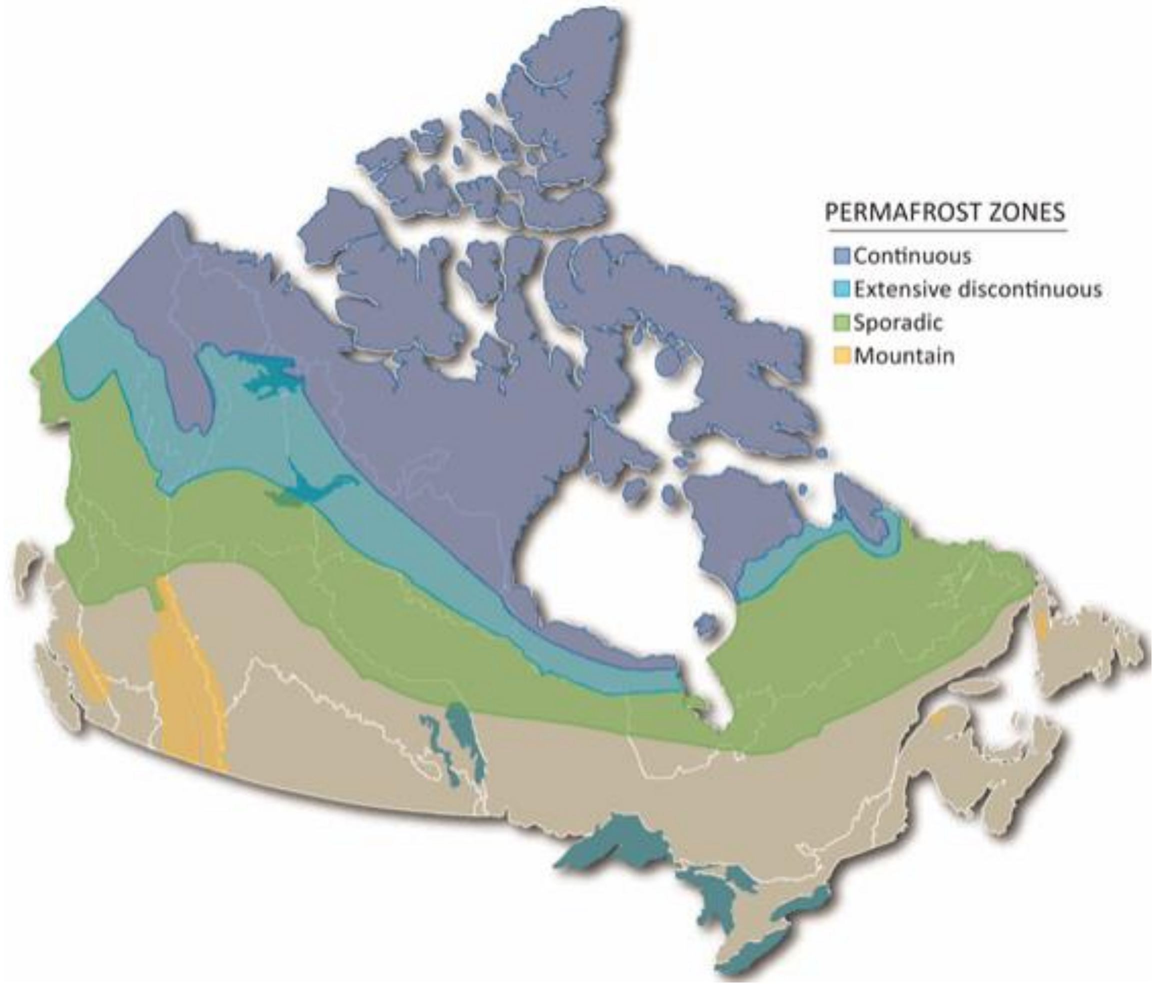


Fig. 2.1 The distribution of permafrost zones across Canada (Smith, 2011). Modified from Heginbottom et al. (1995).

2.2 Impacts of Climate Change on Disturbances within the Boreal Forest

The global climate system is a function of solar and volcanic forcing, ocean and atmospheric currents, atmospheric greenhouse gas composition, surface albedo, and biological processes (e.g., respiration, photosynthesis; Florides et al., 2013). Global climate is consistently in a state of flux, leading to alternating glaciations and periods of warming (Price et al., 2013). However, it is now commonly understood that recent warming is the consequence of anthropogenic contributions to the greenhouse effect (chiefly CO_2 concentrations; IPCC, 2007; Price et al., 2013).

Changes in climate have contributed to the warming of the atmosphere and oceans, loss of continental and sea ice, and a rise in sea levels (IPCC, 2014). Positive feedback mechanisms contributing to the global climate, such as reductions in surface albedo, are accelerating ecosystem change in high-latitude environments (Price et al., 2013). Boreal forests contain over one-third of terrestrial carbon stores, making them critical components of the global carbon cycle, and therefore, regulators of global climate (Price et al., 2013; Mohamed Anas et al., 2015).

2.3 Natural Drivers of Disturbance Regimes in the Boreal Forest Biome

Disturbances are temporary deviations from normal environmental conditions that influence ecosystem change at a broad or local scale. These changes may reduce ecosystem function through modifying the availability of certain resources and services (Turner, 2010). Disturbances may be instantaneous (e.g., storm events) or longer in duration (e.g., runoff from agriculture) and may be a result of either natural or anthropogenic activities. Magnitude and frequency of disturbances are two factors that determine the amount of change that occurs to systems due to disturbances (Miller et al., 2010).

Natural disturbances have been found to exert large influences on ecosystem dynamics within northern forest zones (Venier et al., 2014). Within Canada's boreal region, wildfire, insect infestations, and disease are primary examples of natural disturbances (Kreutzweiser et al., 2013; Brandt et al, 2013). Wildfires are critical for disease regulation within forest stands and heavily influence the species composition of an ecosystem, and therefore biodiversity (Brandt et al., 2013). Natural disturbances are understood to be important factors in maintaining forest ecosystem health, however their influence on a forest stand is largely determined by magnitude and severity (Kreutzweiser et al., 2013).

Wildfires are typically categorized based on intensity, size, seasonality, severity and source of ignition (Miquelajauregui et al., 2016). Fires are classified into groups based on the nature of their burn within a forest stand. Crown fires burn from ground to canopy, and generate the most heat and energy (Natural Resources Canada, 2016). These fires are the most frequently observed fire type within Canada's boreal and are chiefly ignited by lightning strikes (Van Wagner, 1983). Fires restricted to the litter and detritus on the forest floor and forest subsurface are known as surficial burns or ground fire (National Park Service, 2016; Natural Resource Canada, 2016).

The intensity of a fire is equivalent to the rate at which it releases energy per unit time, and is usually measured in kilowatts/metre (Byram, 1959). Fire severity refers to the environmental impacts observed after a burn and is dependent on fire type (e.g., crown fires), and intensity (Miquelajauregui et al., 2016). Measures of severity will vary according to the scope or purpose of a study. For example, severity may be measured as the total amount of surficial organic material consumed in a burn (Barrett and Kasischke, 2013).

There are both large- and small-scale variables that influence the size and location of wildfires, including climate, topography, and fuel (Fig. 2.2; Falk et al., 2011). The magnitude of fire seasons is often represented through comparisons of total forest area consumed. When considering the boreal forest, ‘large’ fire years refers to fire seasons where over 1% of the forest area is combusted (Barrett and Kasischke, 2013).

At millennial timescales, relationships between climate and wildfire frequency are defined by oscillations in orbital and solar parameters (Wanner et al., 2008; McCoy and Burn, 2005; Flannigan et al., 2005). On shorter timescales, increases in atmospheric greenhouse gas concentrations as a result of recent anthropogenic activity have caused increasing temperatures in Arctic regions within the 20th century (Kaufman et al., 2009). Increasing air temperatures are associated with increases in the size of wildland fires (Flannigan et al., 2005). This relationship raises concern as the climate changes. In addition to increases in fire size, warmer temperatures also promote a longer fire season (Flannigan et al., 2005).

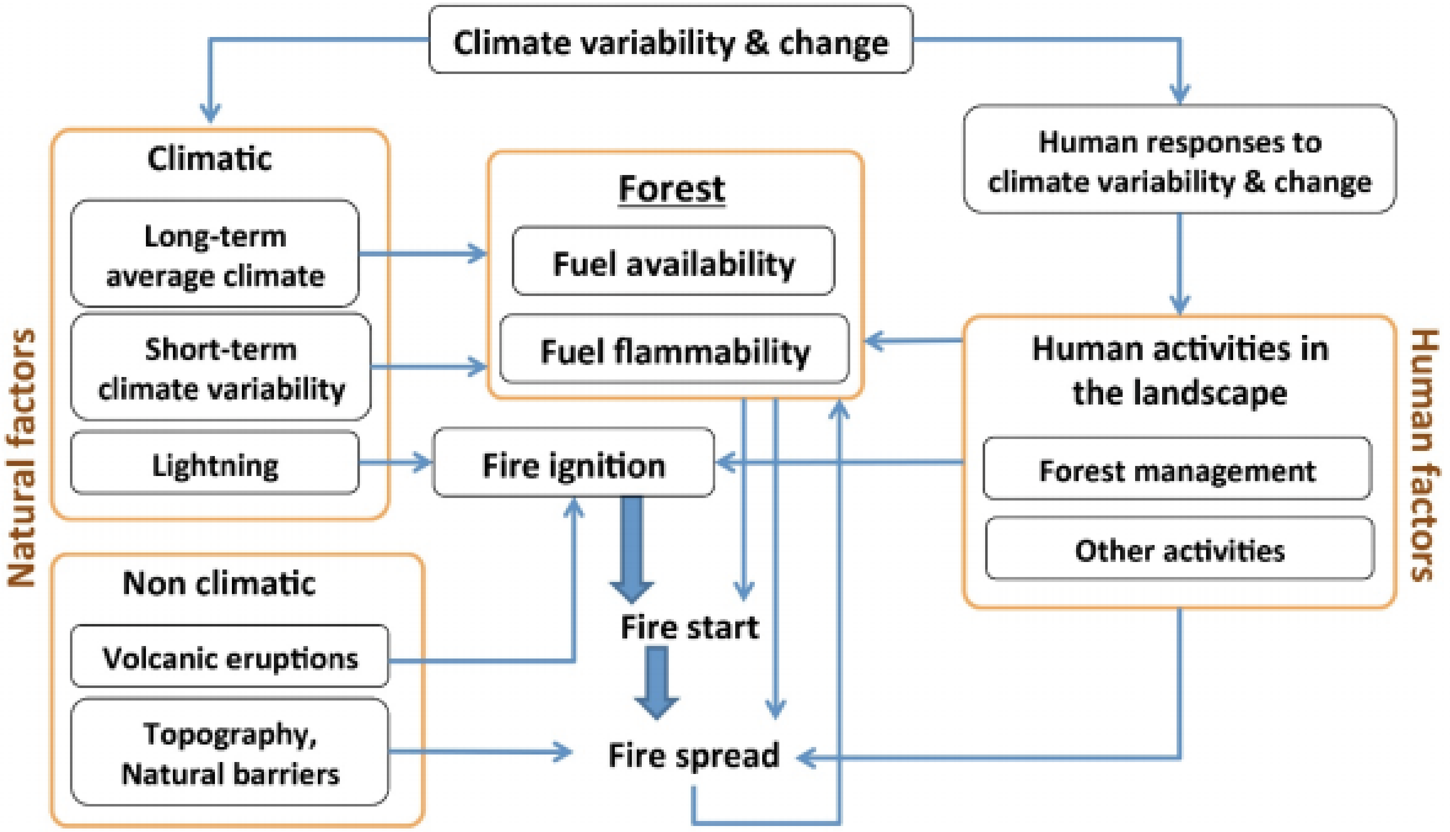


Fig. 2.2. Relationships between climate change and natural and anthropogenic factors influencing wildfire (Herawati et al., 2015).

Paleoecological records, namely charcoal and pollen preserved in lake sediment, indicate that wildfire frequency within the boreal is related to changes in the position of air masses and fluctuating climatic conditions (Carcaillet et al., 2001). During the middle to late Holocene (~2,500 yBP), Canada's eastern boreal was characterized by a hydrological shift which favored reduced precipitation and humidity in summer months, leading to increases in fire frequency (Caracaillet et al., 2001). Between ~3,800 and ~3,150 years BP, the Alaskan boreal experienced increases in precipitation. Unexpectedly, these wetter conditions led to increases in fire frequency (Lynch et al., 2004). Increases in the frequency of lightning strikes was likely the cause of the higher fire activity under these conditions (Lynch et al., 2004). Boreal fire regimes remain heavily influenced by precipitation and lightning events (Fig. 2.2). In the eastern and western forest regions, approximately 40% of wildland fire is a result of lightning strikes occurring under relatively dry conditions (Peterson, 2010). The potential for fire is positively correlated to the number of precipitation-free days in these areas (Peterson, 2010). McCoy and Burn (2005) suggest this can be explained by the fact that large concentrations of dry organic litter provide a highly combustible fuel source and opportunity for ignition.

Fire patterns are also influenced by vegetation structure, which can enhance the possibility of fires transitioning from ground to canopy fires. Some conifers, such as black spruce (*Picea mariana*) have branches along the entire extent of their trunk that act as fire ladders and allow the fire to easily climb from the forest floor to the canopy (Johnson, 1992). Conifer dominated forests tend to have higher burn rates in fire models of the Canadian boreal forest (Bergeron et al. 2004). This is due to the high concentration of flammable oils and resins produced by these conifer groups relative to deciduous species (Senici et al., 2015). In addition, the litter from conifer groups does not decay as readily as organic material produced from hardwoods, which leads to a greater accumulation of combustible fuel on the forest floor (Flanagan and Van Cleave, 1983).

Previously, wildfires within the Canadian boreal biome were considered to be completely stand replacing, however, recent evidence suggests that wildfires may result in more heterogeneous landscape, leaving intact patches and corridors known as ‘residuals’ or ‘remnants’ (Ferster et al., 2016). At a larger scale, vegetation mortality is determined by fire severity, which is intricately linked to several factors including ecoregion type, fuel type, elevation, and soil moisture (Ferster et al., 2016).

Most boreal flora has evolved structures or life strategies which aid in their propagation after a wildfire. These include serotinous cones, root suckers, stem sprouts, and rhizomes (Debano, 1998, Perera and Buse, 2014). Vegetation propagules, which contribute to secondary succession, can arrive from fire residuals mainly from wind transport (Perera and Buse, 2014). In forests of the Mackenzie River Valley, the first groups to colonize burned areas include the liverworts, mosses and shrubs (Larsen, 1980). Early successional woody taxa include alder, willow, birch, and aspen (Perera and Buse, 2014).

Humans have altered fire regimes through modifying the availability of ignition sources and resources such as fuel and moisture. Recreational and industrial activities occurring within the boreal frequently serve as points of ignition for fire events that may result in significant economic and human loss (Flannigan et al., 2008). These human activities, such as resource extraction, are responsible for approximately 62% of the fires that occurred in the Alaskan boreal during the past 50 years (Chapin et al., 2003). On the other hand, processes that act to fragment forest landscapes, such as logging, can reduce the ability of fire to spread, affecting natural fire dynamics. These impacts will be compounded by human-induced climate change, which is expected to increase overall fire frequency within Canadian forests (up to 118% in some areas) (Flannigan et al., 2008). Current wildfire frequency within the Northwest Territories varies due to geographic location, with northwestern regions experiencing increasing wildfire frequencies in comparison to southern locations which have exhibited declining wildfire incidence within the past ~50 years (Natural Resources Canada, 2016). Both climate models and sediment archives from lakes illustrate that increasing temperatures are causing these northern areas to become more vulnerable to increasing fire frequency (Kochtubajda et al., 2006). It should be noted that some areas within the Northwest Territories have experienced the largest increasing annual temperatures in comparison to any other location on the planet since the 1950's. For example, the Mackenzie River Valley has experienced an increase in average annual temperature of 2-3°C (ACIA 2004).

2.4 Factors Influencing the Natural Variability of Streams

Aquatic ecosystems are characterized by abiotic and biotic features of the surrounding landscape. Local geology can influence water chemistry, stream morphology, and the biota found within streams and lakes (Dow et al., 2006). For example, limestone, a soluble sedimentary rock, contributes high amounts of ions (notably calcium) to water chemistry. In addition, the dissolution of limestone can increase stream pH and conductivity, which further impacts biological communities within these systems (Wallace and Eggert, 2009). More resistant bedrock types, such as granite, are less likely to impact stream water chemistry.

The capacity of certain macroinvertebrate taxa to colonize benthic ecosystems is also dependent on the type of sub-strata present; with sandy and silt bottomed streams (typical of limestone rich areas) being unfavorable for taxa of the orders Ephemeroptera, Plecoptera, and Tricoptera (Barnes et al., 2013; Hellmann et al., 2015).

2.5 Impacts of Wildfire on Lotic Systems

Wildfires can have pronounced impacts on aquatic environments. However, these impacts are less studied in comparison to terrestrial ecosystems and warrant further research (Minshall et al., 1989; Bixby et al., 2015). Perhaps this is due to the effects of wildfire on terrestrial landscapes being more readily observable and posing a more immediate threat to infrastructure and human well-being (Bixby et al., 2015). The response of aquatic ecosystems to fire will vary based on the dynamics of the burn and the time that has elapsed since the disturbance.

2.5.1 Stream Morphology

Wildfires influence the morphology of streams through altering both exogenous and endogenous stream features. One of the most visible impacts of fire on stream catchments is the change in total vegetation cover within riparian areas and the surrounding catchment. Riparian vegetation, or the flora located at the land-stream interface, largely influences stream hydrogeomorphology and is an ‘ecosystem engineer’ as it propagates the growth of plant communities through the capture of fluvial-transported sediment and seeds (Gurnell et al., 2012). A readily observable effect of wildfire in catchments is the reduction of this riparian vegetation (Minshall, 1997). Depending on the severity of the burn, combustion of riparian vegetation may lead to increase in stream temperatures as a result of increasing exposure to incoming solar radiation (Minshall, 1997). In addition, the removal of plant roots, which serve as sediment anchors, may result in increased transport of sediment to streams from the surrounding landscape (Lamb et al., 2011). Some studies suggest that post-fire sediment transport to streams may be increased by a factor of 4 (Scott and Van wyk, 1990). This transport can be increased in streams where banks are steeper, resulting in the dry raveling of sediment downslope into the water column. Raveling refers to the failure of individual particles or thin layers of sediment on steep slopes. These processes are compounded by the fact that combustion can reduce the overall cohesiveness of soils (Lamb et al., 2011). Although the reduction of riparian flora may have significant impacts on lotic systems, most impacts are generally short-lived, as this ecosystem has been observed to recover relatively quickly (Minshall, 1997).

The morphology of a stream can also be impacted by movement of trees and other large woody debris (LWD) into stream channels from the surrounding terrestrial environment (Davidson and Eaton, 2015). The rate of LWD transport into stream environments has been found to be higher in burned catchments versus non-burned catchments, however, this increase does not necessarily occur immediately after combustion (Jones and Daniels, 2008). When LWD enters a stream, it may increase the heterogeneity of stream habitats by producing pools, bars, and riffle environments or by altering channel aggradation (Davidson and Eaton, 2015). The time it takes for LWD to impact stream morphology varies based on several factors including the length of time it takes for the LWD to enter the stream, and sediment yield. Typically, the impacts of LWD on stream morphology can be observed 40 years after entering the stream, and are typically the most intense in intermediate sized streams (Jones and Daniels, 2008; Davidson and Eaton, 2015).

Sedimentation, or the movement of sediment particles into an aquatic environment where they are ultimately deposited, is commonly observed in post-fire ecosystems. Increases in sedimentation due to wildfires is generally highest in the first 5 years after the fire event (DeBano et al., 1998). Overland flow (also known as surface runoff), occurs when precipitation exceeds infiltration capacities of soils, or when soils are exceedingly saturated, resulting in the movement of water overland towards the watercourse (Dimitriou, 2011). This process is largely responsible for the movement of sediment into streams in post-fire environments as infiltration rates of soils in burned catchments are reduced through increased hydrophobicity (water-repellency) (Debano et al., 1998). Depending on conditions of the catchment, and the amount of precipitation received, overland flow can produce large debris flows within streams (Staley et al., 2014). Typically, post-fire sediment loading is most profound in lower order streams, where the volume of the stream is less compared to higher-order streams (Minshall, 1997; Minshall, Brock and Varley, 1989). Sediment loading can influence the overall morphology of streams through aggradation, which has been found to cause channel widening and increases in the number of pool-riffle systems (Eaton et al., 2010). In addition, wildfires have also been found to alter normal flow dynamics of streams. Some studies have observed an increase in flow following precipitation events after a wildfire has occurred (Campbell et al., 1977; Klose et al., 2015). Debano et al., (1998) states that peak flow, timing of flow, and base flow of stream networks can also be impacted by fire.

2.5.2 Stream Chemistry

The impacts of wildfire on stream water chemistry are numerous and include changes to physical parameters, ions, and nutrient concentrations. Physical parameters, or the abiotic properties of stream water, include turbidity, total suspended solids (TSS), total dissolved solids (TDS), conductivity, pH, temperature, and dissolved oxygen. These properties largely reflect local environmental conditions in terms of topography, landscape type, and climate (Oliver et al., 2012). Turbidity, or the scattering of light by particulates suspended within the water column, has been commonly observed to increase in fire-affected streams as a result of atmospheric deposition of ash and transport of material into the water column via surface flow (Debano, 1998). Several factors determine the level of turbidity observed in a fire-affected stream, including stream morphology, channel width, water volume, and the severity of the burn (Rhoades et al., 2011). In addition, precipitation events may cause pulses of turbidity as a result of material moving into the stream from the surrounding environment. Some studies have found that precipitation can increase turbidity levels 50-fold in post-fire catchments (Sherson et al., 2015). Turbidity levels resulting from the flushing of the adjacent catchment may remain high for extended periods of time, in some cases upwards of 5 years (Rhoades et al., 2011). Although atmospheric deposition of charcoal does immediately impact water chemistry parameters, the effects are short in duration (Earl and Blinn, 2003).

Stream temperatures may increase as a result of fire if streamside vegetation is removed through combustion. This allows for more incoming radiation to reach the stream, subsequently increasing temperatures within the water column (Minshall, 1989; Klose et al., 2015). Stream temperatures are also influenced by the infiltration of groundwater and stream velocity (Wagner et al., 2014). The latter of these factors has been found to increase in burned catchments (Campbell et al., 1977; Debano, 1998; Klose et al., 2015). Similar to turbidity, water temperatures in a fire-affected stream may not return to pre-fire levels until vegetation within the catchment has recovered (Wagner et al., 2014).

Dissolved oxygen (DO) concentrations are often lower in fire affected streams opposed to unaffected streams due to several processes, including increases in the biological oxygen demand (BOD) (respiration), reduction in canopy cover, and increasing water temperatures (Diemer et al., 2015; Sherson et al., 2015). DO concentrations may be dramatically reduced following precipitation events in burned catchments as a result of increased sediment being transported to the stream. Subsequent increases in turbidity of the water column may reduce primary productivity while increasing BOD (Sherson et al., 2015). Some studies suggest that decreases in DO concentrations within streams is partially due to the oxidation of reduced chemical species within pyrogenic organic matter by available oxygen, such as sulphides, which arrive in a stream through either surface flow or atmospheric deposition (Dahm et al., 2015).

Concentrations of dissolved ions and trace metals may also be impacted by wildfire. Bicarbonate ions (HCO_3-), which are a component of the natural buffering systems of streams, may be transported to streams in additional quantities where they can react with dissolved gasses (CO_2) to become highly soluble salts (Debano, 1998). Base cations, such as magnesium (Mg^{2+}), sodium (Na^+), and potassium (K^+), and other inorganic solutes, have been found to increase in fire affected streams, possibly due to the destruction of the organic layer on the land adjacent to the catchment (Bayley et al., 1992; Bayley and Schindler, 1991; Betts et al., 2009; Diemer et al., 2015). Concentrations of base cations may remain elevated for several years after a fire, but the recovery of vegetation in the surrounding areas may reduce the concentration of ions entering streams (Bayley et al., 1992). Base cations, such as Ca^{2+} are found within plant tissues and enter streams primarily through the leaching of pyrogenic organic matter from the forest floor (Gallaher et al., 2002; Ranalli, 2004).

Several fire studies report elevated anion concentrations, namely sulfate (SO_4^{2-}) and chloride (Cl^-) in burned catchments (Bayley and Schindler, 1991; Ferreira et al., 2005; Betts et al., 2009). Sulfate ions, which are fixed through microbial processes within organic soil horizons, are mobilized through overland flow (Bayley and Schindler, 1991; Ferreira et al., 2005). Chloride ions, produced through the combustion of vegetation, enter soils through leaching and subsequently arrive in streams through infiltration of groundwater (Mast and Clow, 2008). It has been suggested that the combustion of conifer-dominated forests may produce higher concentrations of Cl^- relative to hardwood dominated forest stands (Bayley and Schindler, 1991). Conductivity and pH often increase in post-fire environments, although decreases in pH have been attributed to increasing nitrate (NO_3^-) and sulfate (SO_4^{2-}) concentrations (Williams and Melack, 1997). Streams can become increasingly alkaline through the deposition of ash and charcoal, which can also contribute to greater overall conductivity (Bayley et al., 1992; Dahm et al., 2015).

The effects of wildfire on trace metal concentrations vary (table 2.2). Studies analyzing the changes in stream water chemistry in burned catchments cite increases in iron (Fe), lead (Pb), zinc (Zn), copper (Cu), aluminum (Al), and arsenic (As) (Gallaher et a., 2002; Stevens, 2013; Bodi et al., 2014; Tecle and Neary, 2015). Although forests are large sinks for atmospheric deposition of mercury, the relative concentrations of mercury in post-fire streams does not seem to follow a distinct trend (Gallaher et al., 2002; Driscoll et al., 2013; Stevens, 2013). Cyanide (CN^-), a potential enviro-toxin, may also increase in concentration within disturbed catchments, usually as non-toxic chemical forms (Gallaher et al., 2002; Bodi et al., 2014). Cyanide may arrive in streams through the incorporation of smoke during combustion (Gallaher et al., 2002).

Nutrients, such as nitrate and phosphate, have been found in elevated concentrations within fire affected streams relative to unaffected sites, likely being mobilized by overland flow (Spencer et al., 2003; Hall and Lombardozzi, 2008; Klose et al., 2015). Nitrogen is also easily removed through volatilization during combustion, although the actual amount lost is determined by the severity of the burn and soil moisture (Wright and Bailey, 1982). Changes in nitrogen species within streams following a wildfire are governed by the vegetation within the catchment. For example, the Boreal forest is considered to be poor in available nitrogen, relative to other biomes and may therefore export less nitrogen to streams (Lupi et al., 2013). More specifically, the concentration of nitrogen released to streams during combustion, is governed by the dominant vegetation type within the catchment. Forests dominated by black spruce, typically produce larger nitrogen exports when burned (Betts et al., 2009). Forms of nitrogen generally found to increase within freshwater environments after a burn include ammonia (NH_3), ammonium (NH_4^+), and nitrate (NO_3^-) (Gallaher et al., 2002; Bodi et al., 2014). Combustion of the inorganic forms of nitrogen found within forest litter converts them into plant-usable forms (such as ammonia) through oxidation (Williams and Melack, 1997). Upon being transported to streams, these species may be converted to nitrate through biological oxidation (Willams and Melack, 1997). Decreases in dissolved organic nitrogen (DON) concentrations have observed in streams that were recently burned, have been found to recover in subsequent years (Betts et al., 2009; Diemer et al., 2015).

Phosphate (PO_4^{3-}) concentrations may increase within catchments as a result of wildfire (Spencer et al., 2003; Bodi et al., 2014). Generally, phosphorous is found in two forms within soil; orthophosphate (inorganic phosphate) and organic phosphate (Debano, 1998). Organic phosphate (or phosphate P) is highly reactive with other soil components and is not likely to undergo leaching (Debano, 1998). Therefore, phosphate entering streams occurs through movement of soil into the water column, or through the deposition of ash (Gerla and Galloway, 1998).

Combustion of vegetation releases otherwise trapped carbon from biomass to the atmosphere (in the form of CO_2) and soil (Bodi et al., 2014). Incomplete combustion of organic material results in the formation of pyrogenic carbon (PyC), which may be removed from local carbon stores through atmospheric or overland transport (Bodi et al., 2014). Increases in PyC within watersheds have been observed following large precipitation events, due to overland flow (Cotrufo et al., 2016). PyC may also arrive in streams through groundwater infiltration (Cotrufo et al., 2016). While PyC concentrations typically increase in burned catchments, dissolved organic carbon (DOC) levels have been reported to decrease after wildfire (Betts et al., 2009; Diemer et al., 2015).

Table 2.2 Various water quality parameters and their general response to wildfire.

Parameter Type	Water Chemistry Parameter	General Response of Parameters to Wildfire Relative to Baseline Conditions
Physical	Turbidity/TSS	+
	Temperature	+
	Dissolved Oxygen	-
	pH	+/-
	Conductivity	+
Major Ions	Ca^{2+}	+/-
	Mg^{2+}	+
	K^+	+
	Na+	+
	SO_4^{2-}	+
	Cl^-	+
Nutrients	DOC	-
	DON	-
	NO_3^-	+
	PO_4^{3-}	+
Trace Metals	Al	+
	Cu	+
	Fe	+
	Hg	+/-
	Pb	+
	Zn	+

2.5.3 Stream Ecology

Primary producers (PP), or autotrophs, are organisms that obtain their energy through a complex series of oxidative reactions. These organisms are integral to both terrestrial and aquatic ecological systems as they utilize otherwise biologically unavailable energy sources, such as solar radiation or chemical reactions to produce complex compounds, which can then be acquired by other organisms (Allan and Castillo, 2007). In streams, major groups of primary producers include algae and submerged macrophytes (aquatic plants).

Unfortunately, there is very little information available on the impacts of wildfire on submerged aquatic macrophytes. The combustion of the riparian zone and soils within the surrounding catchment can increase the discharge of streams, reduce infiltration capacity and increase overland flow. Changes in discharge and input of debris from the surrounding catchment can scour stream channels, resulting in damage to the bed and changes in stream morphology (Meyer and Pierce, 2003). This may impact aquatic plant communities as velocity determines species composition and abundance within streams, with a lower number of species occurring in streams that are prone to flooding events (Riis and Biggs, 2003). Stream macrophytes will display morphologies that endure hydrodynamic drag forces imposed by the current (Riis and Biigs, 2003). In addition, increasing turbidity within the water column can reduce overall macrophyte abundance through reducing available light, furthermore, hindering primary productivity (Schutten et al., 2005).

Algae, a polyphyletic group of aquatic primary producers, are important components of aquatic ecosystems as they are primary food sources for consumer organisms and produce oxygen as a by-product of their metabolism. The overall effects of wildfire on stream algal assemblages are ambiguous, with no impact being reported on multiple occasions (Earl and Blinn, 2003; Malison and Baxter, 2010). Several studies do report decreases in algal biomass in burned streams; attributing losses to increasing sediment load (which reduces light penetration), and scouring of stream beds (Beche et al., 2005; Pete et al., 2013; Klose et al., 2015). The morphology of the stream and the nature of the material moving into the stream will determine the overall impact on algal communities (Peat et al., 2013; Klose et al., 2015). Significant changes in diatom (siliceous phytoplankton) assemblages have been observed in burned lotic systems, with assemblages dominated by larger taxa transitioning to those with smaller, adnate species (Earl and Blinn, 2003; Klose et al., 2015). Although none of the literature reviewed reported increases in algal biomass post-fire, potential increases in biomass may occur following pulses of nutrient input. Intense browsing by grazer invertebrate groups may mask increases in algal concentrations (Malison and Baxter, 2010).

Benthic macroinvertebrates refer to the groups of animals that live in aquatic environments for at least a portion of their lifecycle. They belong to many taxonomic groups including annelids, gastropods, bivalves, amphipods, platyzoans, and insects (Wallace and Eggert, 2009). These organisms are essential to ecosystem dynamics in freshwater environments as they are a primary source of food for higher order consumers like fish. Benthic macroinvertebrates are commonly employed in studies focussing on environmental change in aquatic ecosystems (Angeler and Goedkoop, 2010; Glaz et al., 2014; Lewis et al., 2014), because each grouping of organisms demonstrates unique responses to different environmental stressors (Fig. 2.3).

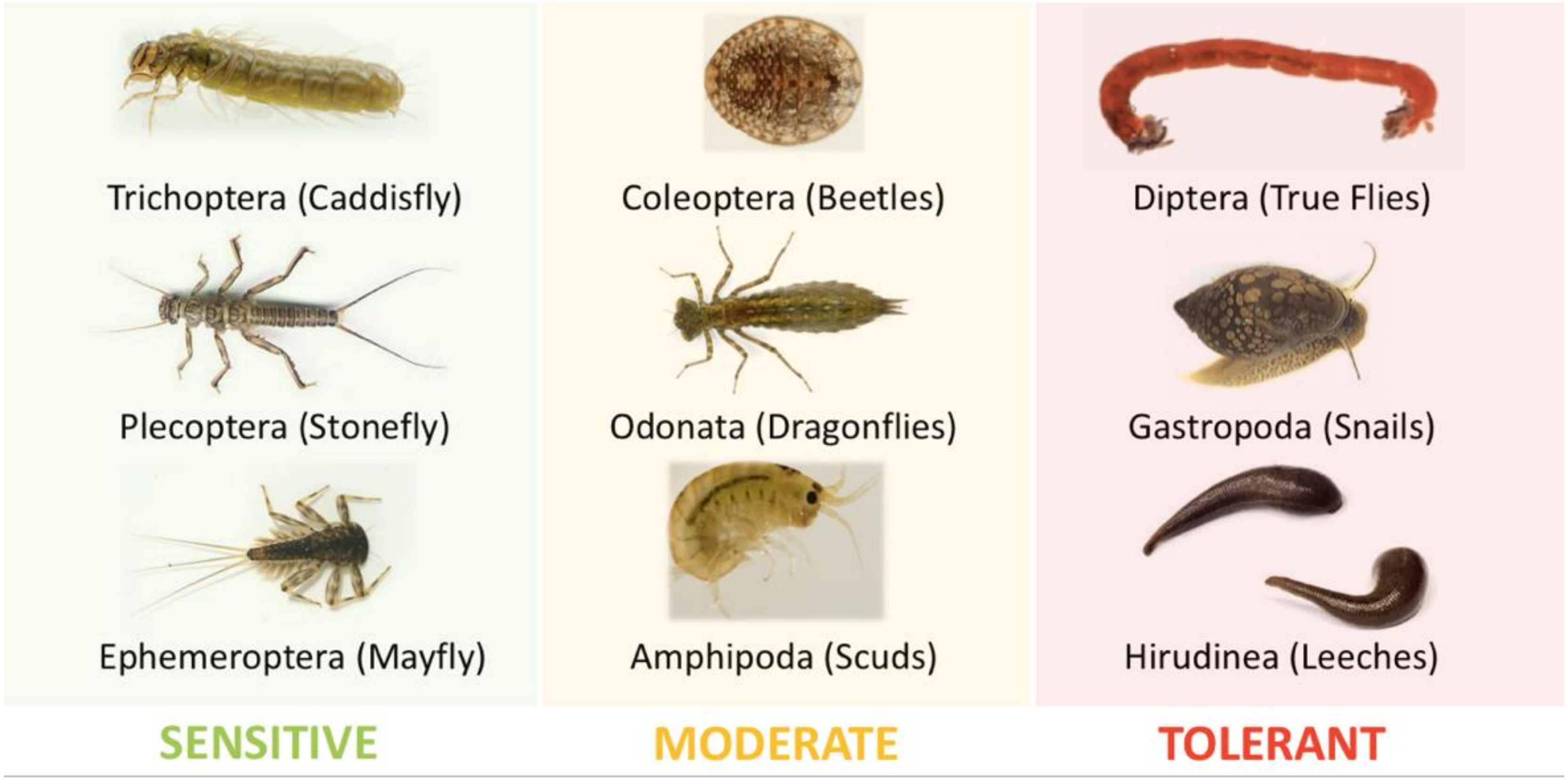


Fig. 2.3 Major macroinvertebrate taxa and their *general* tolerances to declining water quality. Freshwater systems with high dissolved oxygen, high pH, cooler temperatures, and suitable habitat may contain all major groups of taxa. Impaired streams and lakes will likely contain macroinvertebrate groups tolerant to lower dissolved oxygen, pH, and warmer temperatures (namely leeches, worms, snails, and certain fly larvae)

Fire events can affect aquatic ecosystems by promoting increases in total suspended sediment, changing water chemistry, and introducing nutrients into streams (Fig. 2.4). These modifications to the physical and chemical environment subsequently alter stream communities and ecosystem dynamics. Macroinvertebrate populations are impacted more readily by indiscriminately timed inputs of terrestrial material compared to immediate changes to the surrounding landscape after a fire (Arkle et al., 2010). Random hydrological events occurring in the years following fire may disrupt recovering aquatic macroinvertebrate communities by modifying stream morphology, increasing sediment input, and affecting available food sources (Vieira et al., 2004; Arkle et al., 2010).

Organic debris arriving in streams and lakes from terrestrial ecosystems serves as primary food sources for some aquatic organisms. Generally, this material is categorized as either coarse particulate organic matter (CPOM; 1>mm) or fine particulate organic matter (FPOM; 0.45 μm – 1 mm) (Smith and Smith, 2012). Examples of the former category include twigs, leaves, and larger debris from the canopy or adjacent riparian zone. Smaller material, such as detritus, phytoplankton, and the biological compounds they produce, are considered examples of FPOM. In addition, organic material can be classified based on whether it is produced externally and subsequently transported to aquatic systems (allochthonous), or whether it is generated within the stream (autochthonous) (Glaz et al., 2014).

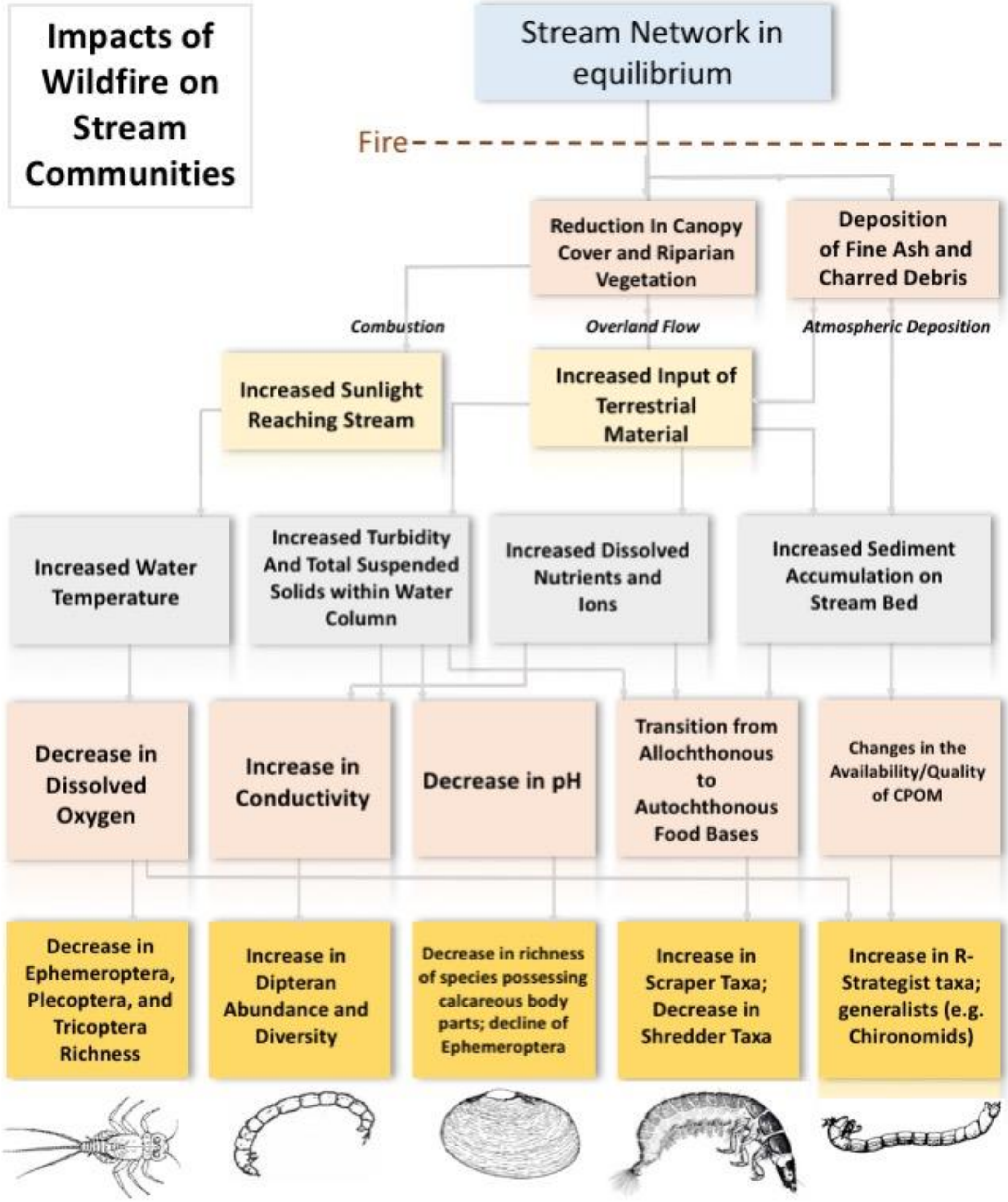


Fig. 2.4. Conceptual model of the potential direct impacts of wildfire on benthic invertebrate taxa. Declines in overall water quality are the product of sediment input and changes in water temperature, which occur through the processes of combustion, overland flow, and atmospheric deposition.

Invertebrate communities within freshwater systems are assigned to 'functional feeding groups' (FFG's) based on their food preference (e.g., CPOM, FPOM), and life habits. FFG's commonly used in freshwater studies include shredders, scrapers, predators, filterers, and gatherers. These classifications are not based on taxonomy, but rather consider the general behaviours of taxa and their modes of feeding (West Virginia Department of Environmental Protection, 2016). General behaviours of major functional groups and example taxa are included below (Table 2.3).

Previous analyses of aquatic macroinvertebrate assemblages have noted shifts from communities bearing taxa sensitive to sediment inputs (e.g., heptageniid mayflies, nemourid stoneflies, and scraper taxa) to communities with increased generalist taxa with high larval dispersal capabilities, especially chironomids (Minshall et al., 1997; Vieira et al., 2004). Decreased canopy cover and riparian vegetation lead to decreases in allochthonous inputs and may result in a decline of shredder taxa (Fig. 2.4; Wallace and Eggert, 2009). Conversely, increased rates of primary productivity resulting from a reduction in shade from the canopy may increase the amount of autochtonous material available for scraper/grazer taxa (Wallace and Eggert, 2009). Generally, there is a positive correlation between the abundance of r-strategist taxa and the magnitude of the disturbance, with larger and more severe fires being associated with increases in chironomids, *Baetis* (mayfly), and Simuliidae (blackfly) (Malison and Baxter, 2010).

Table 2.3. Major functional groups found within freshwater ecosystems and their behaviours. Note that the example taxa included will exhibit these behaviours within their aquatic larval stages. Adult specimens will demonstrate alternate behaviours (Monterey Peninsula Water Management District, 2004; West Virginia Department of Environmental Protection, 2016).

Functional Group	**Behaviour**	**Example Taxa**
Scrapers and Grazers	Consume phytoplankton, detritus, flora, micro-fauna, and biological compounds produced by other organisms (epilithon)	Hydropsychidae (Web-spinning caddisflies) *Psephenu*s (water pennies)
Shredders	Consume CPOM entering system from surrounding landscape. This includes wood, litter and other organic debris.	*Lepidostoma* (caddisflies)
Collectors and Gatherers	Collect FPOM from substrate	*Baetis* (mayflies), Orthocladiinae (midges)
Filterers	Collect FPOM suspended within water column	Hydroptilidae (purse-case caddisflies)
Predators	Secondary consumers	Empididae (dance flies) *Argia* (damselflies)

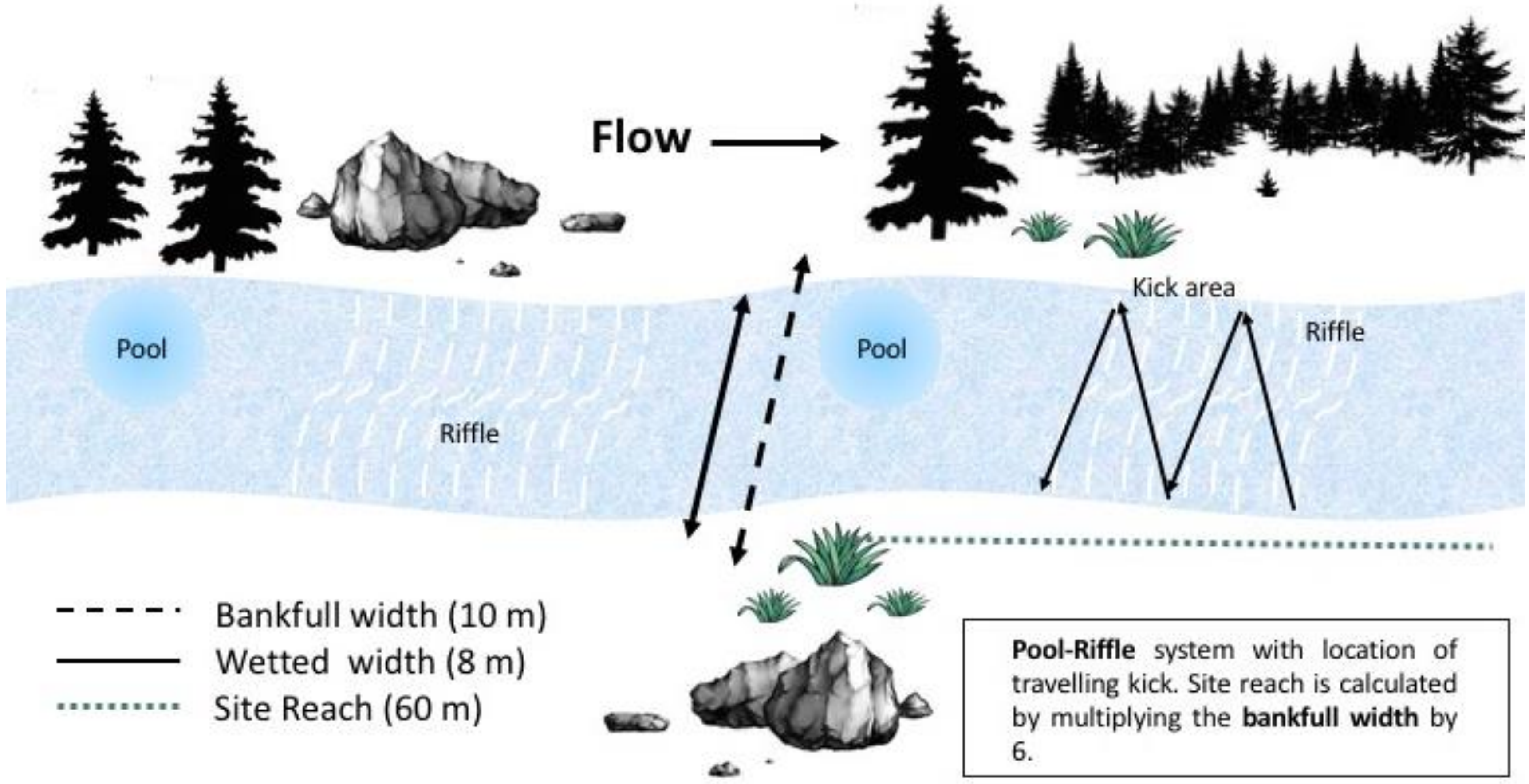
2.6. Methodological Approaches: CABIN Protocol

The Canadian Aquatic Biomonitoring Network (CABIN), officially launched in 2006, provides protocols for the standardized collection and storage of aquatic ecological data (Phillips, 2008). The principal purpose of CABIN is to monitor the ecological health of Canadian aquatic ecosystems while providing a forum for the sharing of data. Although CABIN aims to incorporate multiple proxy types into its protocols, it currently focusses on the collection of benthic macroinvertebrates. Unlike most monitoring exercises which use temporal or spatial replication, CABIN suggests the use of the reference condition approach (RCA) as a means of determining whether differences exist between reference sites and tests sites. RCA, a form of predictive modelling, uses Discriminant Function Analyses (DFA) to determine groups of ecologically similar reference sites within a region that can then be compared to tests sites that should have similar benthic communities (Reynoldson et al., 1997). Deviations of test site data from reference site data observed in ordination space are assumed to be due to environmental disturbances (Reynoldson et al., 1997).

Based on the guidelines of the Canadian Aquatic Biomonitoring Network (CABIN), benthic organisms are collected from riffle habitats within streams. These areas are high-energy environments, thought to contain the highest diversity of macroinvertebrate taxa including stress-sensitive groups such as certain mayflies, caddisflies, and stoneflies (Merritt et al., 1996; Environment Canada, 2011). These sensitive groups are important in environmental impact studies, as their assemblages will reflect the health of the ecosystem at the time of sampling.

Informative water chemistry parameters including pH, temperature, dissolved oxygen, and conductivity are collected using a multi-parameter sonde, while a physical water sample is analyzed for concentrations of nutrients, major ions, and metals. Benthic organisms are collected from streams using a kick net. Sampling commences downstream in riffle habitats and continues upstream in a zig-zag fashion for three minutes (Fig. 2.5). Benthic organisms are removed from the substrate by a ‘kicking’ motion, and are carried into the net/collection tube by the flow of the stream (Environment Canada, 2011).

Macroinvertebrates are generally subsampled from a Marchant box; a case containing 100 cells from which organisms are randomly selected, counted, and identified. This subsampling technique is used to eliminate sampling bias. Organisms are then identified to the lowest taxonomic order possible using identification keys provided by literature and CABIN field manuals (Environment Canada, 2014).



Pool-Riffle system with location of travelling kick. Site reach is calculated by multiplying the **bankfull width** by 6.

Fig. 2.5. Methodology for in-stream sampling of benthic macroinvertebrates. Sampling commences downstream within the riffle and continues upstream in a zig-zag fashion for a maximum of 3 minutes to standardize sampling effort.

2.7. References

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Chapter Three

The Impacts of Rececnt Wildfires on Stream Water Quality and Macroinvertebrate Assemblages in Southern Northwest Territories, Canada

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3.1 Introduction

Climate change is expected to impact high-latitude ecosystems to a greater extent than temperate regions (IPCC, 2014; AMAP, 2017). During the past century, notable changes in temperature and precipitation have been observed across high latitude regions (ACIA, 2005). Since climate is one of the primary drivers of many natural processes, including disturbances such as wildfire, changes in disturbance regimes are expected to occur as temperatures continue to rise and precipitation regimes are altered (IPCC, 2014).

Wildfire is a common feature of the boreal forest, and is essential for maintaining ecosystem health and function (Weber and Flannigan, 1997). The impacts of wildfire on terrestrial ecosystems have been well described (Smithwick et al., 2005; Carcaillet et al., 2006; Nelson et al., 2008; and Seedre et al., 2011), however, the effects of wildfire on aquatic ecosystems have not been investigated as fully, especially those in high latitude regions. In the last decade, the Northwest Territories (Canada) has experienced severe fire seasons, with a total of 3,416,291 ha of forest burned in 2014 alone (approximately 5.6% of total forested land within the territory). The total area burned in 2014 was approximately 1.4 times larger than the national yearly average for all of Canada (2,527,647 ha) for the period 2007-2017 (Natural Resources Canada, 2018; National Forestry Database, 2016).

Benthic macroinvertebrates have been used as indicators of ecosystem change in response to multiple forms of disturbances, including wildfire (Mellon et al., 2008; Arkle et al., 2010; Malison and Baxter, 2010; Oliver et al., 2012; Verkaik et al., 2015). Invertebrate communities have been used to detect changes in allochthonous inputs (Wood and Armitage, 1997; Oliver et al., 2012), stream morphology (Minshall et al., 2001), and nutrients and primary productivity (Malison and Baxter, 2010; Silins et al., 2014). Benthic macroinvertebrates are widely used water quality indicators because they are abundant in freshwater ecosystems, and the ecology of many low-level taxonomic groups is well studied. Benthic macroinvertebrates are mostly sedentary. Therefore, their presence reflects long-term upstream conditions, but communities may also be shaped by random stochastic events (e.g. large rainfall events). In addition, sample collection is relatively straightforward and inexpensive, which is advantageous when working in high latitude environments where remote and adverse conditions are frequently encountered. Benthic macroinvertebrates are even more useful when coupled with data regarding environment and water quality, providing an informative toolset for discerning the potential causes of ecosystem change in aquatic systems.

This study investigated the short-term impacts of recent wildfires (occurring within 2 years of sampling date) on the physical and ecological condition of streams found in the North Slave, South Slave, and Dehcho regions of the Northwest Territories, Canada (Figure 3.1). Two separate studies were conducted as part of this research. Specifically, we used pre-and post-fire water chemistry data for the Cameron River (North Slave Region), which was affected by the Reid Lake fire in July 2016, to understand how wildfire coupled with local climate factors, such as total monthly precipitation, impacted water quality in high-latitude streams. Two benthic invertebrate samples collected before and after the Reid Lake fire were also compared to discern changes in invertebrate community structure. In addition, a control-impact study design was used to analyze differences in invertebrate community structure and functional ecology between impacted (affected by recent fires) and control (unaffected by recent fires) streams within the Dehcho and South Slave regions. This is the first study to examine the effects of wildfire on benthic invertebrate ecology for these sub-Arctic regions. Many streams included within this study did not previously have environmental, water quality, or ecological data available. Therefore, this study aimed to provide preliminary insights regarding how stream invertebrate communities respond to environmental changes induced by recent wildfire events within a northern setting. Specifically, this research aimed to answer the following questions:

1) Are there observable differences in benthic invertebrate community metrics (ecological measures describing community structure) between impacted streams and control streams?
2) Do benthic invertebrate communities within impacted streams have higher proportions of taxa tolerant to changes in water quality such as Chironomidae (non-biting midges) + Baetidae (mayflies), + Simuliidae (blackflies) (R-strategists), Gastropoda (freshwater snails), and Bivalvia (freshwater clams)?
3) Do benthic invertebrate communities within impacted streams have higher proportions of taxa belonging to functional feeding groups that primarily consume fine particulate organic matter (FPOM) and detritus, such as collector-filterers, collector gatherers and scrapers?

3.2. Description of Study Site

The study area is located in the southern Northwest Territories, Canada. Study sites were selected within the North Slave, South Slave, and Dehcho regions (Fig. 3.1, Table 3.1)

3.2.1 Environmental Characteristics of the Study Regions

The North Slave region is situated within the Taiga Shield Level II ecoregion and is characterized by cool and dry climatic conditions. The most extensive climate records available for this region are for the capital city of Yellowknife (Environment Canada, 2017).

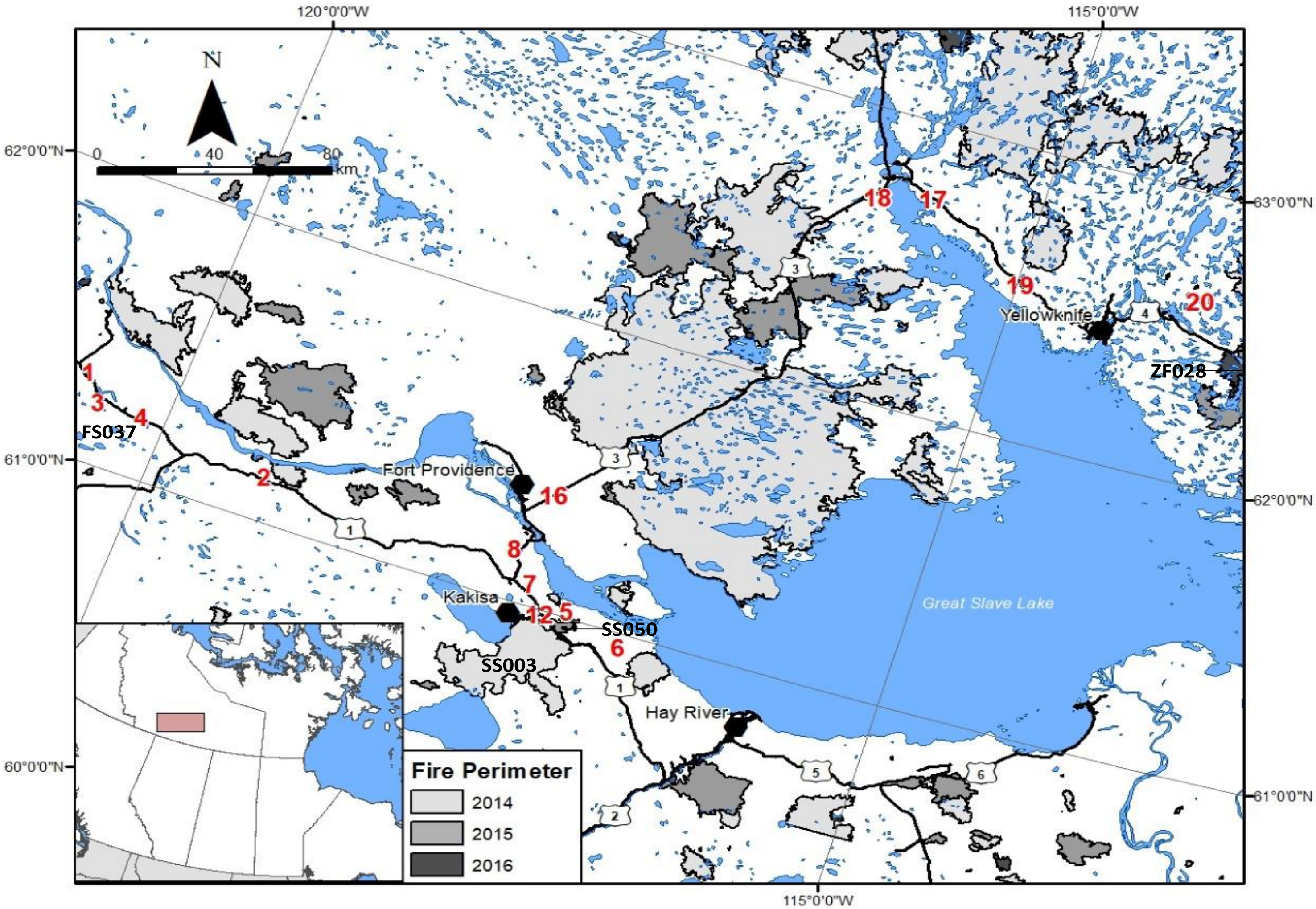

Figure 3.1. Map of the study region in Southern Northwest Territories.

Table 3.1. Location, stream condition, and general catchment characteristics for streams included within this study.

[1]Site No.	Site Code	Site Name	Year Sampled	Lat./Long.	Condition	Catchment	Catchment Size (km^2)	Bedrock Type
1	FSC-01	UN	2016	61.35750/-119.2422	Control	UM – Jean Marie	20,189.91	Sedimentary
2	FSC-02	Wallace Creek	2016	61.31556/-120.3275	Control	UM– Jean Marie	20,189.91	Sedimentary
3	FSB-01	UN	2016	61.32222/-120.0189	Impacted	UM– Jean Marie	20,189.91	Carbonate
4	FSB-03	Trout River	2016	61.37528/-117.1247	Impacted	UM – Jean Marie	20,189.91	Sedimentary
5	KC-02	UN	2016	60.99000/-116.7158	Control	GSL – Outlet	2,116.30	Sedimentary
6	KC-03	McNallie Creek	2016	60.95556/-117.3942	Control	GSL– Outlet	2,116.30	Sedimentary
7	KVB-01	UN	2016	61.08000/-117.5717	Impacted	Kakisa	15,508.04	Sedimentary
8	KVB-02	UN	2016	61.17944/-117.4317	Impacted	Kakisa	15,508.04	Sedimentary
9	KVB-03	UN	2016	61.06278/-117.3850	Impacted	Kakisa	15,508.04	Carbonate
10	KBU-02	UN	2016	61.19778/-117.5717	Impacted	GSL – Outlet	2,116.30	Sedimentary
11	KBU-03	UN	2016	61.08861/-117.2671	Impacted	GSL – Outlet	2,116.30	Sedimentary
12	KU-01	UN	2015	60.99425/-117.4179	Control	Kakisa	15,508.04	Sedimentary
13	KR-01	Kakisa River	2015	60.94183/-117.5157	Impacted	Kakisa	15,508.04	Carbonate
14	CH-01	Chikilee Creek	2015	61.14000/-117.5170	Control	GSL – Outlet	2,116.30	Sedimentary
15	WS-01	Wolfskull Creek	2015	61.14072/-117.4469	Control	GSL – Outlet	2,116.30	Sedimentary
16	BF-01	Bluefish Creek	2015	61.39814/-115.7245	Control	Horn	20,753.75	Sedimentary
17	ST-01	Stag Creek	2015	62.74358/-116.0888	Control	Snare	15,840	Granitoid
18	MO-01	Mosquito Creek	2015	62.6962/-114.9701	Control	GSL-North Arm	7,983.47	Evaporite
19	BOC-01	Boundary Creek	2015	62.53472/-113.7667	Control	Yellowknife	18,777.78	Granitoid
20	CR-01	Cameron River	2016	62.63417/-120.4278	Impacted	Yellowknife	18,777.78	Granitoid

[1]Site number corresponds to numbering on Figure 3.1.

UN=No formal name available, UM=Upper Mackenzie, GSL=Great Slave Lake

Thirty-year climate normals for Yellowknife (1981-2010) show that winters are typically cold and dry. January is the coldest month with a mean air temperature of -25.6°C. On average, the largest amount of snow falls during the month of November (36.5 mm). Snowpack depth begins to accumulate in November, decrease rapidly in March, and is generally absent by May (Environment Canada, 2017). Summers for this region are normally cool and wet. July is the warmest month with an average air temperature of 17°C. July is the also the wettest month, with an average of 40.8 mm of precipitation falling as rain (Environment Canada, 2017).

The North Slave region is dominated by coniferous forest, consisting of jack pine (*Pinus* banksiana), white spruce (*Picea* glauca), black spruce (*Picea* mariana), and aspen (*Populus* sp.). Jack pine is abundant on granitic rock outcrops that are found throughout the North Slave. Black spruce is more abundant in low-lying areas where soils are thicker and more organic. White spruce is more abundant with increasing proximity to the northern treeline. Paper birch (*Betula* papyrifera) and Dwarf birch may be common in recently burned areas (Ecosystem Classification Group, 2008).

The dominant soil types across the North Slave study region are mainly brunisols with organic cryosols and gleysols occurring in some areas (Ecosystem Classification Group, 2008). The North Slave region is located within the Slave geological province (Archaen-Neo-Proterozoic) (Galloway et al., 2015). Bedrock of this area is largely composed of granites and gneisses and is relatively rich in minerals bearing elements such as silica (Si), lithium (Li), beryllium (Be), and gold (Au). Sedimentary rocks form a minor component of the underlying geology. Large evaporite deposits, notably gypsum ($CaSO_4 \cdot 2H_2O$), are found in western parts of the North Slave region.

Study sites located in South Slave and Dehcho regions fall within the Taiga Plains ecoregion. In comparison to the Taiga Shield ecoregion, climate is slightly warmer and wetter with mean annual temperature and total annual precipitation of -4ºC and 300 mm, respectively (Ecosystem Classification Group, 2007). Fort Simpson is the closest climate station with normals available for the period 1981-2010.

Thirty-year climate normals for Fort Simpson (1981-2010) indicate that January is the coldest month with a mean air temperature of -24.2°C. On average, the largest amount of snow falls during the month of November (33.4 mm). Similar to climate records for Yellowknife, historical records for Fort Simpson indicate that snowpack depth begins to accumulate in November, decrease rapidly in March and April, and is generally absent by May (Environment Canada, 2017). Summers in Fort Simpson are slightly warmer and wetter in comparison to Yellowknife. July is the warmest month with an average air temperature of 17.4°C. July is the also the wettest month, with an average of 61.1 mm of precipitation falling as rain (Environment Canada, 2017).

Within the South Slave and Dehcho regions, the topography is more subdued and more mesic conditions result. Given the more mesic conditions, black spruce and tamarack (*Larix* laricina) are common tree species throughout the vast low-lying portions of the South Slave and Dehcho regions (Ecosystem Classification Group, 2007). Soil types within the Dehcho and South Slave regions are variable and dependant on local geology, although gleysols and brunisols are common. The Dehcho and South Slave regions are part of the interior platform geological province (Paleozoic) and contain extensive lacustrine deposits produced from the formation of glacial Lake McConnell approximately 10,000 yBP (Smith, 1994; Galloway et al., 2015). The majority of the streams sampled in these regions are underlain by fossiliferous carbonate bedrock.

3.2.2 Recent Wildfires

The study area experienced severe burns during the 2014, 2015 and 2016 wildfire seasons. The largest fire in 2014, SS003, burned immediately to the east of the community of Kakisa, consuming ~77,600 ha (Fig. 3.1). In 2015, fires SS049/SS050 (the Mill's Lake fire complex) occurred east of Kakisa along the Mackenzie Highway and Highway 1 (Fig. 3.1). These fires were much smaller in size (~3300 ha burned). The most recent fires examined in this study occurred during the summer of 2016. Fire ZF028 burned ~7500 ha of forest in the Reid Lake area, approximately 50 km east of Yellowknife (Fig. 3.1). One-year prior to ZF028, a smaller fire (ZF014) occurred within the same area. Fire FS037 was located 10 km south of the Mackenzie Highway and Highway 1 near Fort Simpson, and was ~1600 ha in size. In general, fires included within this study occurred during warmer and drier conditions relative to the 30-year climate normals for the respective areas (Table 3.2). Fire SS003 occurred under slightly wetter settings compared to climate normals recorded for the nearby hamlet of Hay River (Table 3.2). At the time fire SS003 was detected, mean daily air temperatures were approximately 3°C higher than climate normals for the area (Table 3.2).

Table 3.2. Attributes of fires included within this study, local climate conditions near the time of their detection and 30-year climate normals (1981-2010). Climate data is acquired from the nearest Environment Canada climate station with climate normals available for the period 1981-2010 (Environment Canada, 2017).

Fire	Date Fire Detected	Size of Fire (ha)	Mode of Ignition	*Mean Daily Air Temp. (°C)	*Mean Daily Air Temp. (°C) 1981-2010	*Total Monthly Precip. (mm)	*Mean Monthly Precip. (mm) 1981-2010
SS003[1]	May 28th, 2014	77600	Lightning	8.0	5.4	33.3	23.3
SS049[1] SS050	July 29th, 2015	3300	N/A	17.5	16.1	40.6	43.3
FS037[2]	July 18th, 2016	1600	N/A	18.4	17.4	46.4	61.1
ZF028[3]	July 14th, 2016	7500	Suspected Person	18.3	17.0	15.4	40.8

* Mean air temperature and precipitation values were calculated for the month and year the fire was detected (ex. for ZF-028, monthly averages were based on data collected during the month of July). Climate information is obtained from Environment Canada climate stations in [1]Hay River (NWT), [2]Fort Simpson (NWT), and [3]Yellowknife (NWT). N/A = Source of ignition not available

3.3 Methodology

3.3.1 Site Selection

Study streams were identified using geospatial hydrology data obtained from the *Roads, Rivers, and Waterbodies* database (Natural Resources Canada) . Initially, streams were selected based on their proximity to major highways and recent burn scars. Fire perimeter data, indicating the spatial extent of past fires, was also obtained from the Geomatics Division of the Government of the Northwest Territories. Fire perimeter data were combined with hydrology data to produce maps indicating the location of potential control and impacted streams. For impacted streams, only those located in recently burned catchments (burned in 2014, 2015, and 2016) were considered. Control streams were not impacted by fire (i.e., no streams traversed recently burned areas, and no fires occurred in headwater catchments of study streams within 10 years of the sampling date). With the exception of FSB03 (Trout River), all streams were either first or second order. To avoid possible influence from roads, sampling took place upstream from any road where possible. Stream sampling was conducted during August of 2015 and 2016 (Table 3.1).

3.3.2 Water Chemistry

Four stream water samples were collected per site for analysis of general physical parameters (including alkalinity, conductivity, TSS, TDS, turbidity, and pH), nutrients, total dissolved metals, and trace elements. One of four samples collected from each site was treated immediately with nitric acid (HNO_3) to prevent precipitation of metals into their oxide components and adsorption to container walls prior to total metal concentration analysis. Water samples were stored in a cool, dark place until their submission to the Taiga Environmental Laboratory in Yellowknife, Northwest Territories. A YSI multi-parameter sonde was used to record water temperature, pH, conductivity, and salinity at each sampling site. Equipment malfunction prohibited the collection of dissolved oxygen measurements at the study sites.

3.3.3. Benthic Macroinvertebrates

Benthic macroinvertebrate samples were collected following the protocols of the Canadian Aquatic Biomonitoring Network (CABIN) (Environment Canada, 2011). General characteristics of the surrounding landscape, such as topography and land use, were recorded for each site. Various stream characteristics, such as habitat types, percent canopy coverage, percent macrophyte coverage, total streamside vegetation, dominant streamside vegetation and percent periphyton coverage were recorded for each stream.

Collection of macroinvertebrates took place in riffle habitats. A d-shaped kick net with attached dolphin bucket (collection tube) was used for sampling. Sampling commenced downstream within the sample reach. The sample reach was calculated by multiplying the bank-full width of the stream by a factor of 6 (Environment Canada, 2011). Macroinvertebrates were dislodged from the substrate by kicking the streambed for three minutes in a zigzag fashion while moving upstream through the riffle. Stream flow acted to carry displaced macroinvertebrates into the net and collection tube. Standardizing travelling kicks of three minutes ensured consistent sampling effort between sites. Large debris such as stones and branches were thoroughly rinsed over the kick net and discarded. Contents of the net and collection tube were rinsed over a sieve and placed into 500mL sample jars with 99% isopropanol as preservative. Sample jars were labelled and sealed for transport to Cordillera Consulting Laboratories (Summerland, British Columbia) for taxonomic identification.

Bank-full width and wetted width were recorded following invertebrate collection. Bank-full width was determined by identifying areas of exposed roots and substrate resulting from abrasion during high flow. The difference between the depth of stagnation and flowing water depth (ΔD) was determined at each site using the velocity head-rod ruler method (Wilm and Storey, 1944) and used to calculate velocity as follows:

$$\text{Velocity (m/s)} = [2(\Delta D/100) * 9.81] \quad \text{(Eq. 1)}$$

Equation from Environment Canada (2011)

To characterize stream substrate, 100 randomly selected pebbles/rocks were measured along their intermediate axis (perpendicular to longest axis) and recorded. Interstitial substrate size was visually assessed and assigned a valued from 0-9, with 0 representing organic cover/silt and 9 representing bedrock.

3.3.4. Macroinvertebrate Analysis

Macroinvertebrate samples were analyzed by Cordillera Consulting Laboratories following CABIN laboratory methods (Environment Canada, 2014). Generally, macroinvertebrates were removed from original preservative and placed in >70% ethanol until identification (Environment Canada, 2014). Prior to sorting, samples were rinsed of preservative using a 400-µm sieve. Samples were evenly distributed into subsamples using a Marchant Box (Marchant, 1989). For each sample 300 individuals were randomly subsampled from a minimum of 5% of the total sample. If over 50% of the sample was required to reach 300 organisms, then the whole sample was sorted and identified

(Environment Canada, 2014). To ensure accurate sorting of sample, sorting efficiency was calculated using the following formula:

%Sorting efficiency=(1-(Organisms missed/Total Organisms Found))*100 (Eq. 2)

Equation from Environment Canada (2014)

CABIN requires all taxonomic identifications be made to the family level at minimum with a minimum of 95% sorting efficiency required. For quality assurance purposes, 10% of the total sample was re-identified to ensure <5% identification error rate (Environment Canada, 2014).

3.3.5. Data Preparation

Statistical analysis was conducted under the assumption that streams with comparable environmental and physicochemical properties should host similar benthic macroinvertebrate communities. It was also assumed that major differences in assemblages within these streams are due to deviations from normal environmental conditions (Hall and Lombardozzi, 2008).

Twenty water chemistry variables consistently fell under the detection limit and were removed from the data set. These variables included orthophosphate (as phosphorus), fluoride ions, nitrite (as nitrogen), $_{\text{total}}$antimony, $_{\text{total}}$beryllium, $_{\text{total}}$cadmium, $_{\text{total}}$chromium, $_{\text{total}}$cesium, $_{\text{total}}$lead, $_{\text{total}}$selenium, $_{\text{total}}$silver, $_{\text{total}}$thallium, $_{\text{dissolved}}$beryllium, $_{\text{dissolved}}$cadmium, $_{\text{dissolved}}$cesium, $_{\text{dissolved}}$lead, $_{\text{dissolved}}$silver, $_{\text{dissolved}}$thallium, $_{\text{dissolved}}$vanadium, and $_{\text{dissolved}}$zinc. No analysis was conducted on these parameters. Pearson correlation tests were conducted on the remaining water chemistry parameters to determine autocorrelation amongst variables. Following the methods of Chin et al. (2016), two variables attaining a pairwise r-value $-0.7 < r > 0.7$ were considered to be autocorrelated. The variable having fewer strong correlations with other variables were removed. When variables had similar amounts of strong correlations with other variables, the variable deemed more ecologically significant was retained. Ecological significance was based on findings of other studies focussing on the response of stream macroinvertebrates to ecosystem change (Hall and Lombardozzi, 2008; Lento et al., 2013; Chin et al., 2016). This reduced the number of water chemistry variables included within the analysis from 77 to 19.

Direct comparison of water chemistry and benthic assemblages required comparable environmental conditions at both impacted and control streams. Impacted streams were located within the South Slave and Dehcho regions and shared similar environments with control streams in these areas. Evaluation of bedrock geology, surficial geology, soil type, and dominant vegetation revealed major differences in these water quality variables between streams located in the North Slave region and those located in the Dehcho and South Slave regions. Ordination plots constructed from water quality variables illustrated large differences between streams located in the North Slave versus the other regions. As a result, 4 streams in the North Slave region were removed from the analysis, leaving 7 control streams and 5 impacted streams for comparison.

Water chemistry parameters, environmental variables, and biological metrics were tested for normality using the Shapiro-Wilk test in *XlStat* statistical software (Addinsoft, Inc., 2017). Variables that did not follow a normal distribution were transformed using the square root and $\log_{10}(x+1)$ functions to improve normality where possible. To compare control and impacted sites in ordination space, data were normalised through conversion of data to z-scores.

Site location, reach, and substrate were recorded where these observations were made. Percent canopy coverage, percent macrophyte coverage, and periphyton thickness (mm) were initially recorded as ranges and later converted to ranks assigned by the CABIN protocol (Environment Canada, 2014). Several ecological metrics (Hilsenhoff biotic index, Shannon Weiner diversity index, species richness, total abundance, relative abundance and richness of major taxonomic groups, and the relative abundance and richness of major functional feeding groups) were calculated using invertebrate data. These metrics were chosen based on their ability to detect environmental perturbations in studies using benthic macroinvertebrates as biological indicators (Hall and Lombardozzi, 2008; Lento et al., 2013; Wallace et al., 2013; Chin et al., 2016).

Temporally continuous water chemistry data for the Cameron River (CR-01) were prepared in a similar manner to data collected from streams within the Dehcho and South Slave regions. Water chemistry data for the Cameron River was collected on a monthly basis (January to December) by the members of the Government of the Northwest Territories using multiple grab samples. Although water quality data for the Cameron River is available from 1983 to present, only data for the period 2013-2016 are analyzed for this study. Variables that consistently fell under the detection limit were or were autocorrelated with other variables were removed. Pearson's r was calculated and two variables attaining pairwise r-values of $-0.7< r > 0.7$ were scrutinized and the less ecologically significant variable was removed.

3.3.6. Data Analysis

Annual variability in water chemistry, precipitation, and hydrological data for the Cameron River (CR-01) between January 2013 and December 2016 (n=48) were assessed using descriptive statistics. Pearson correlation coefficients were calculated for water chemistry variables, precipitation, and hydrological data to determine how water chemistry was influenced by climate and flow regimes. Pettitt's (1979) change-point analysis was performed on all water quality variables in order to determine possible changes in the probability distributions of water quality variables due to fire. There is a lack of invertebrate samples available for the Cameron River, therefore, environmental data and metrics calculated from benthic macroinvertebrate data collected in 2015 and 2016 were directly compared and described based on their differences.

Following the methods of Hall and Lombardozzi (2008), benthic macroinvertebrate metrics, water chemistry, and environmental data were grouped together as either control or impacted streams to create a control-impact study design. Eight environmental variables, 19 water quality parameters, 25 ecological metrics, and family level abundance were statistically compared using a one-way ANOVA or Mann-Whitney test ($\alpha=0.05$). Tests were chosen based on whether data for variables met key assumptions for each test (e.g. normal distribution, variable independence). Pearson's r was calculated for metrics and environmental data, metrics and water quality data, and metrics and fire measures (time since fire (years), fire size (km^2)).

3.4 Results

3.4.1 Cameron River (CR-01), North Slave Region, NWT

Measured environmental and reach characteristics did not differ in 2015 and 2016, with the exception of periphyton thickness. In 2015, periphyton coverage of stream substrate was between 0.5 and 1.0mm thick. The following year, the periphyton mat had decreased in thickness (<0.5mm). Water level and discharge were higher in August 2016

(1.85m, 2.14 $m^3 s^{-1}$) compared to August 2015 (1.66m, 0.371 $m^3 s^{-1}$). Both measurements were recorded at the time of invertebrate sampling.

On average, August was the wettest month (mean=35.77 mm) for the entire study period (January 2013 to December 2016) while the least precipitation was measured during the month of March (mean=5.75mm). Mean monthly stream discharge recorded during this period was also variable with the maximum and minimum discharge occurring July (3.097 $m^3 s^{-1}$) and April (1.329 $m^3 s^{-1}$), respectively. Mean water level was 1.851m (±0.091m).

Daily discharge was significantly correlated with total As ($_{total}As$) (r=0.420, p=0.003), TOC (r=0.312, p=0.031), alkalinity (r=-0.478, p=0.001), $_{dissolved}Al$ (r=0.446, p=0.009), $_{dissolved}Mn$ (r=-0.424, p=0.014), and $_{dissolved}Li$ (r=-0.494, p=0.004). Monthly total precipitation was also significantly correlated with turbidity (r=0.386, p=0.007). No significant correlations were found between daily discharge, monthly total precipitation and water quality variables.

Pettitts change point analysis (1979), performed on water quality data sets for the Cameron River, indicated single change points in 7 parameters including $_{total}As$ (K=312, p=0.002), $_{total}Li$ (K=276, p=0.037), $NO_3+NO_2\_N$ (K=533, p=<0.0001), TOC (K=358, p=0.002), TP (K=271, p=0.044), $_{dissolved}U$ (K=184, p=0.002), and daily discharge (K=484, p=<0.0001)). No significant change points were detected in the data during or after the Reid Lake fire event in July 2016 (Table 3.3).

Table 3.3. Pettitt's change point test statistics for Cameron River water quality data for the period 2013-2016. There were no significant changes detected in the data during or after the Reid Lake fire (July 2016).

Parameter	Pettitt's K	t	Change Point	p-value
Total As	312	15	March 12, 2014	0.002
Total Li.	276	16	April 14th, 2014	0.037
TP	271	19	July 23rd, 2014	0.044
TOC	358	20	August 26th, 2014	0.002
Daily discharge	484	22	October 22nd, 2014	<0.0001
Dissolved U	184	24	December 15th, 2014	0.002
NO_3+NO_2_N	533	28	April 9th, 2015	<0.0001

Several water quality variables exhibited large peaks which occurred synchronously with high precipitation following the Reid Lake fire in July 2016, including turbidity, TDS, and ${}_{\text{dissolved}}\text{Al}$ (Fig. 3.2). Between January 2013 and December 2016, mean turbidity for the Cameron River was 0.554 ± 0.320 NTU. In September 2016, two months following the Reid Lake fire, turbidity values reached 2.06 NTU (Fig. 3.2). This is more than 3σ above the mean turbidity value and represents $<0.1\%$ of the population for approximately normal distributions. This was the maximum turbidity value recorded during this time period. Values returned to a mean of ~0.45 NTU by October (Fig. 3.3). Concentrations of ${}_{\text{dissolved}}\text{Al}$ increased immediately following the Reid Lake fire and by August 2016 reached their maximum concentration of 13.3 µg/L. The August maximum ${}_{\text{dissolved}}\text{Al}$ concentration was more than 3σ higher than the mean for the 2013-16 period (3.19 µg/L). Like turbidity, ${}_{\text{dissolved}}\text{Al}$ concentrations decreased rapidly after the fire event and by September 2016, ${}_{\text{dissolved}}\text{Al}$ concentrations had decreased to ~1.8 µg/L. Peak values of TDS were observed during July of 2016 (69 mg/L). However, these values decreased to mean values by August 2016 (54 mg/L), and never exceeded 2σ from the mean (50.17 mg/L). Turbidity values appeared to increase simultaneously with monthly precipitation values (Fig. 3.2). However, heavy precipitation events recorded in August of 2014 did not correspond with major increases in turbidity like those observed after a large precipitation event in September of 2016, following the Reid Lake fire (Fig. 3.2).

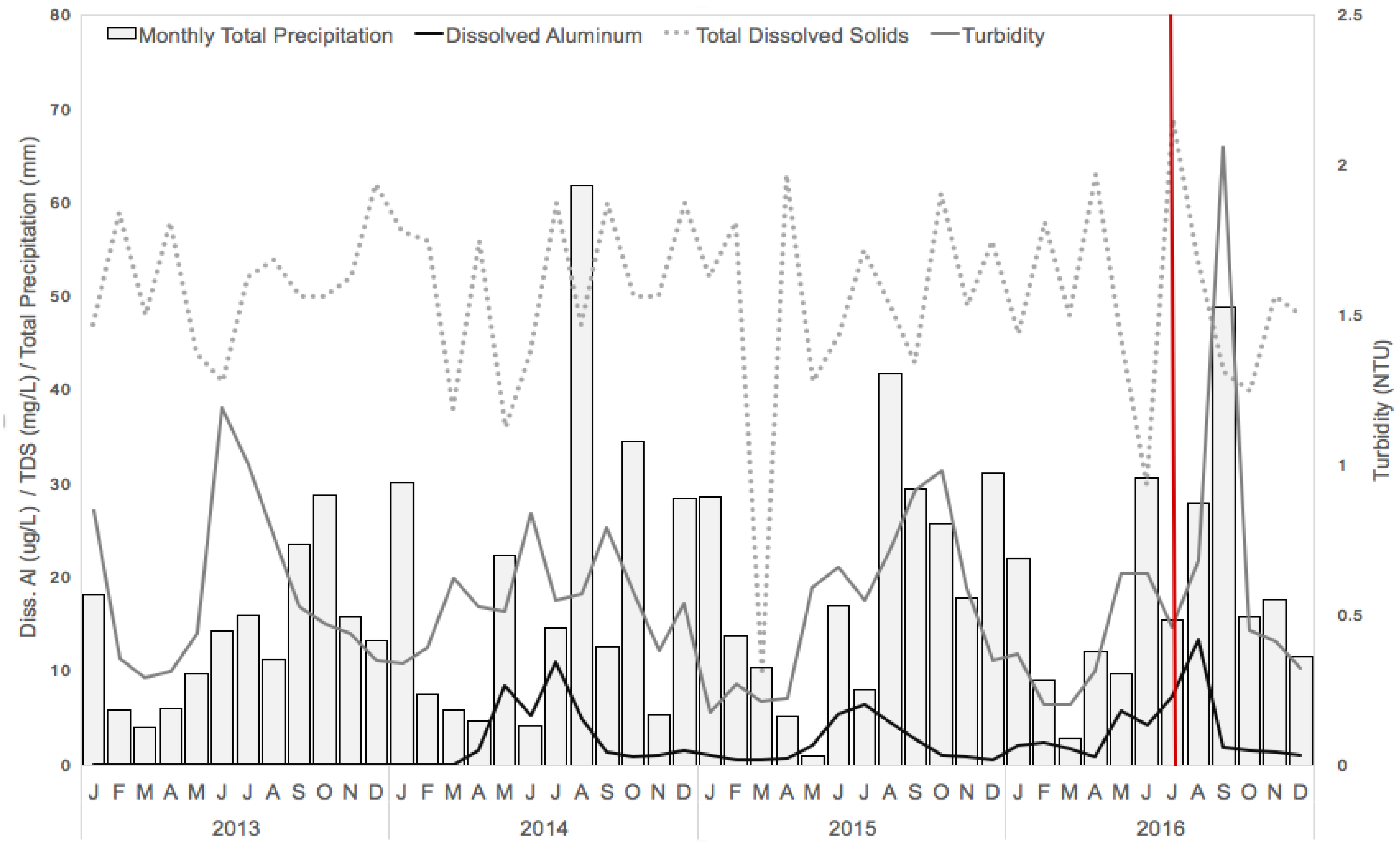

Fig. 3.2. Changes in TDS, ${}_{\text{dissolved}}$Al, and turbidity relative to the Reid Lake fire (vertical red line). TDS and ${}_{\text{dissolved}}$Al increased immediately following the fire, while an increase in turbidity followed a large precipitation event in September 2016.

Benthic macroinvertebrate communities only slightly differed between 2015 and 2016 (Fig. 3.3, Fig. 3.4). Total macroinvertebrate abundance was higher in 2016 compared to 2015, with corrected abundances of 5359 and 1553 total organisms, respectively. In regard to diversity, Shannon-Weiner diversity indices indicated the benthic community in 2016 was marginally less diverse in comparison to 2015 (H_{2016}=1.39, H_{2015}=1.42). The Hilsenhoff biotic index calculated for both years indicates a slightly larger proportion of 'hardy' or 'disturbance-resistant' taxa relative to sensitive taxa in the year following the fire (Hilsenhoff Biotic Index_{2015}=5.35, Hilsenhoff Biotic Index_{2016}=5.77).

The largest differences in relative abundance were observed in the orders Diptera (true flies) and Ephemeroptera (Mayflies; Fig. 3.3). In 2015, Diptera constituted 42% of the invertebrate community, but their relative abundance increased to 52% in 2016. Analysis of changes in abundance of low-level taxonomic groups reveals that the dipteran families Chironomidae (non-biting midges) and Simuliidae (black flies) experienced the largest changes in abundances between 2015 and 2016. Specifically, the abundance of Simuliidae increased from 83 individuals to 1638 individuals in 2016. The Simuliidae family was the most abundant in the 2016 community sample compared to Orthocladiinae chironomids in 2015.

The relative abundance of Ephemeroptera was lower in 2016 compared to 2015 (12% and 6%, respectively), even though their actual abundance increased. Although no other major differences in relative abundance between communities were observed, large increases in actual abundance were detected in Tricoptera (caddisflies) (681 individuals), Gastropods (snails) (233 individuals), and Bivalvia (clams) (318 individuals) between 2015 and 2016.

The relative abundance of both collector-gatherers and collector-filterers were substantially different between sampling years. The collector-gatherers decreased in relative abundance from 32% in 2015 to 16% in 2016 (Fig. 3.4). Family level taxonomic analysis reveals that there is an appreciable increase in the abundance of individuals belonging to the annelid family Naidiidae (92 individuals) between 2015 and 2016. This is the only group amongst any of the collector-gatherer taxa that displays a large difference in abundance between the two study years.

The collector-filterers increased in relative abundance from 2015 to 2016. Collector-filterers relative abundance in 2015 was 29% compared to 52% in 2016 (Fig. 3.4). This difference in relative abundance of collector-filterers is attributed to increases in the families Hydropsychidae (Tricoptera) (271 individuals), Simuliidae (Diptera) (1545 individuals), and Pisidiidae (Bivalvia) (208 individuals). The major increase in relative abundance of collector-filterers in 2016 is likely the result of the growth of the Simuliidae population.

Major taxa groups included within this analysis demonstrated higher richness indices in 2015 compared to 2016, with the exception of the oligochaetes, which increased in richness by a factor of 3 (Fig. 3.4). With the exception of the shredder taxa, there are no large changes in the richness of functional feeding groups. There were four shredder taxa within the 2015 benthic community, whereas only 2 taxa were identified in 2016 (Fig. 3.4)

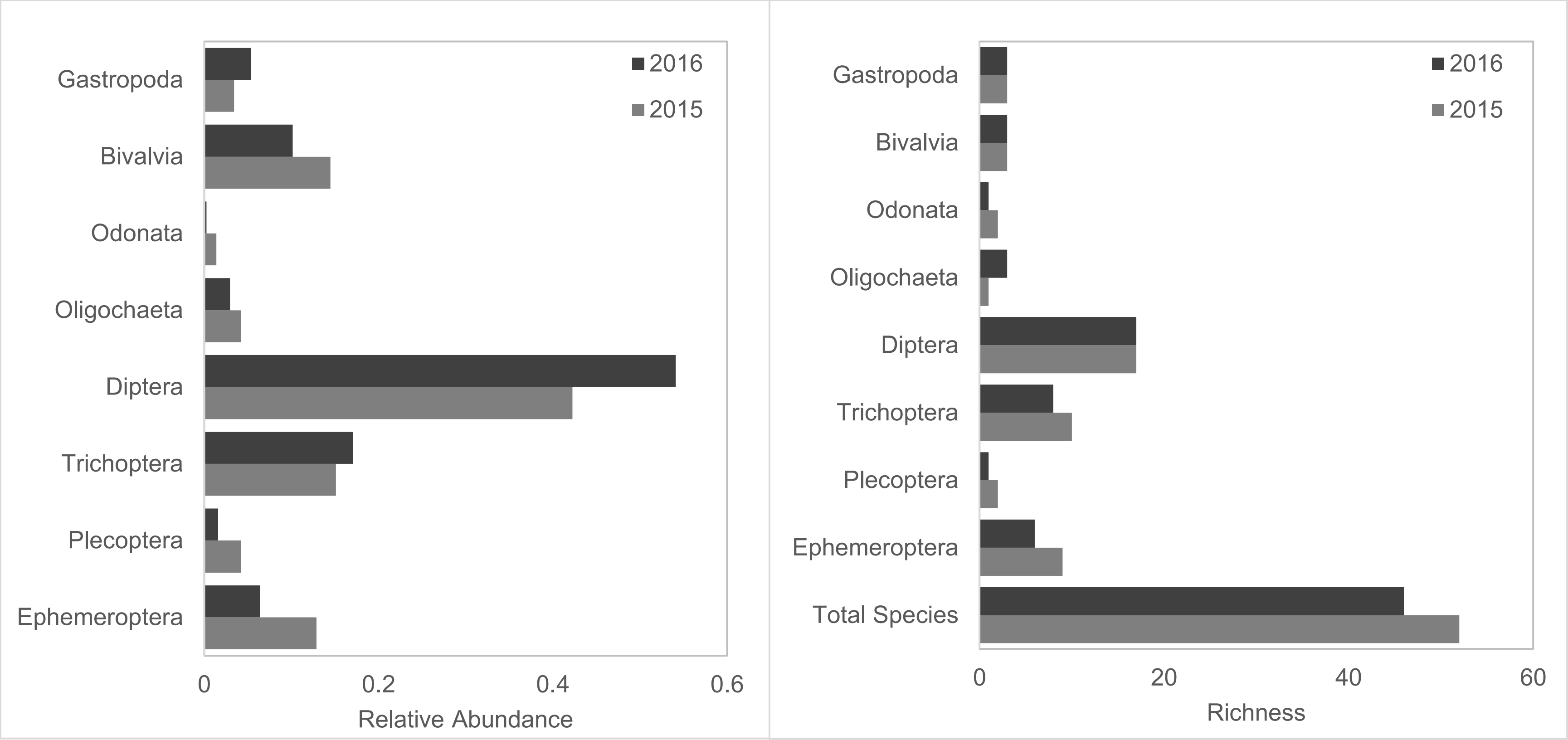

Fig. 3.3 Comparison of relative abundance and richness indices for major taxa groups present in the Cameron River between 2015 and 2016.

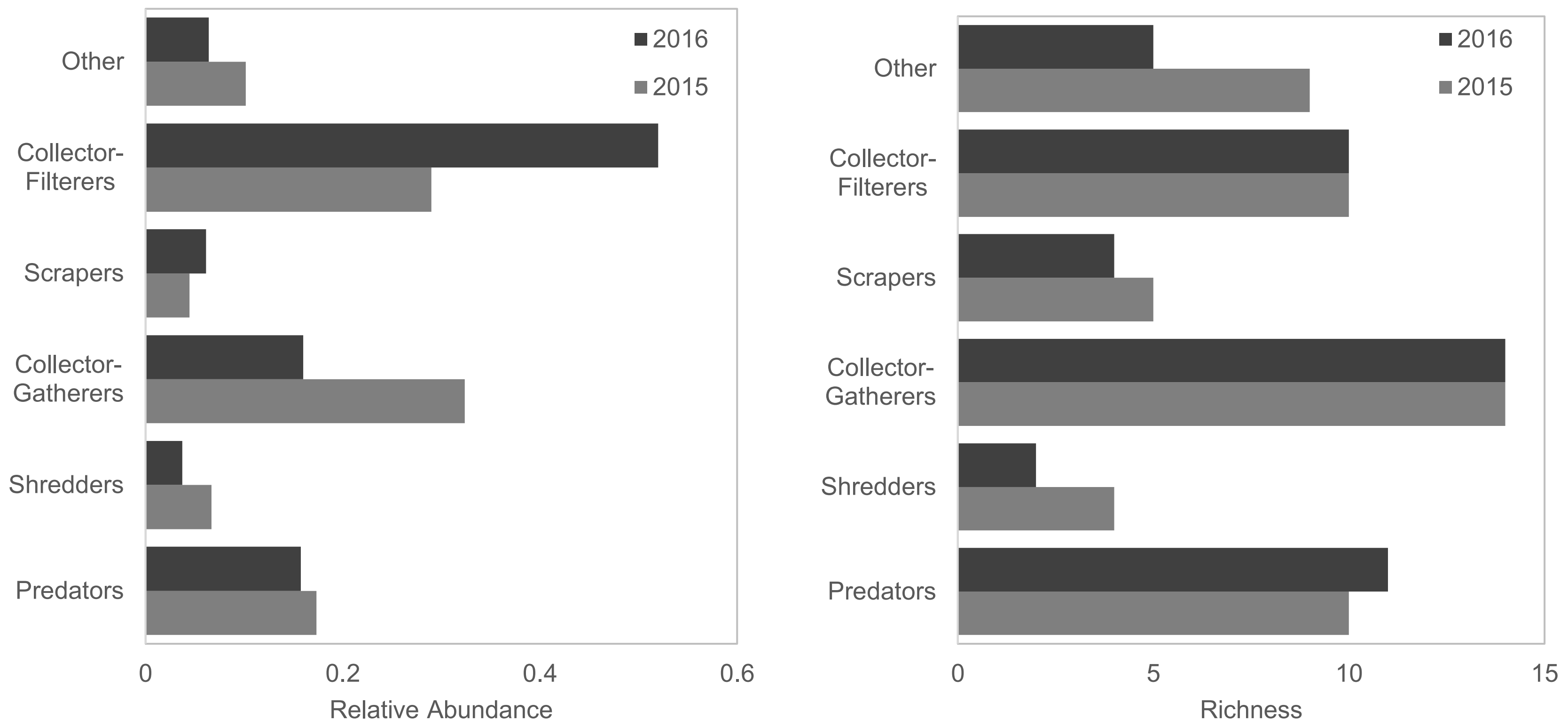

Fig. 3.4. Comparison of relative abundance and richness indices for functional feeding groups present within the Cameron River between 2015 and 2016. The ‘other’ category includes macrophyte-herbivores, omnivores, piercers, and unclassified taxa.

3.4.2. South Slave and Dehcho regions, NWT

Reach characteristics and stream velocity did not vary greatly between control (n=7) and impacted streams (n=5) in the South Slave and Dehcho regions. Wetted width and bank-full width exhibited the most variation between control and impacted streams, however, FSB03 (Trout River) has a relatively large channel width (~53.51m) and likely skewed the means of these parameters. Mann-Whitney (non-parametric) tests indicated significant ($p<0.05$) differences in the distributions of macrophyte coverage categories between control (median=1) and impacted (median=0) sites (U=20.5, p=<0.0001). ANOVA tests detected significant differences in the means of log-transformed interstitial substrate values between control (mean=2.4±1.35) and impacted (mean=3.83±1.39) sites (F=5.380, p=0.046).

Mean alkalinity, TSS, turbidity, and ${}_{\text{dissolved}}\text{Al}$ varied the most amongst control (n= 7) and impacted sites (n=8) with respect to water quality parameters. On average, mean alkalinity values were lower for impacted streams (163.2 ± 56.2 mg/L) in comparison to control streams (231.0 ± 24.2 mg/L). Sample means for TSS, turbidity, and ${}_{\text{dissolved}}\text{Al}$ concentrations were greater in impacted streams, with the largest difference observed in TSS concentrations (control=3.4 ± 1.1 mg/L; impacted=16.3 ± 17.6 mg/L). Analysis of variance (ANOVA) tests conducted on water chemistry variables showed significant differences in the means (p-value<0.05) for alkalinity (F=9.655, p= 0.008) and ${}_{\text{dissolved}}\text{Al}$ (F=9.176, p=0.010) (Fig. 3.5). Mann-Whitney tests showed significant differences in the distributions of TSS values between control and impacted streams (U=9.5, p=0.011).

Pearson's *r* was calculated to determine potential correlations between water quality parameters and burn characteristics (size of burn and years since burn), and indicated significant correlations ($p<0.05$) for 7 water quality variables including TOC, conductivity, pH, NO_3^- as N, K^+, $_{\text{dissolved}}$As, and $_{\text{dissolved}}$Cu (Table 3.4). Burn size is positively correlated with $_{\text{dissolved}}$As (r=0.895, p=0.001) and negatively correlated with pH (r=-0.755, p=0.050). Years since the fire is most strongly correlated with $_{\text{dissolved}}$As (r=0.907, p=0.005) and TOC (r=-0.789, p=0.035).

Slight differences in biological metrics are observed between impacted (n=5) and control (n=7) streams. Mean corrected abundances of impacted and control streams are 3832±2749 individuals and 2417±848 individuals, respectively. High standard deviations reflect larger variation in corrected abundances within impacted streams (Min=1077 individuals, Max=7560 individuals) compared to control streams (Min=1393 individuals, Max=3965 individuals). Mean Shannon diversity index calculated for each group indicated that diversity is marginally higher within impacted streams ($H_{\text{impacted}}=1.31\pm0.11$) compared to control streams ($H_{\text{control}}=1.18\pm0.16$). Comparison of mean Hilsenhoff Biotic Indices indicated higher proportions of disturbance-sensitive organisms present within impacted streams (4.77±1.04) opposed to control streams (5.72±0.82), although both means suggest fair stream water quality. No significant differences in the means or distributions of these indices were detected by ANOVA or Mann-Whitney statistical tests ($\alpha=0.05$).

Table 3.4. Pearson correlation coefficients (α≤0.05) between years since fire/size of fire and water quality variables for streams located in the Dehcho and South Slave regions, NWT. Only the significant correlation coefficients were included.

Water Quality Variable	Years Since Fire	P-value	Size of Fire (km^2)	P-value
TOC	0.895	0.006	0.954	0.001
Conductivity	0.870	0.011	0.755	0.050
pH	-0.771	0.042	-0.755	0.050
NO_3^- as N	0.805	0.029		
K^+	0.881	0.009	0.822	0.023
$_{\text{dissolved}}$As	0.907	0.005	0.957	0.001
$_{\text{dissolved}}$Cu	-0.789	0.035		

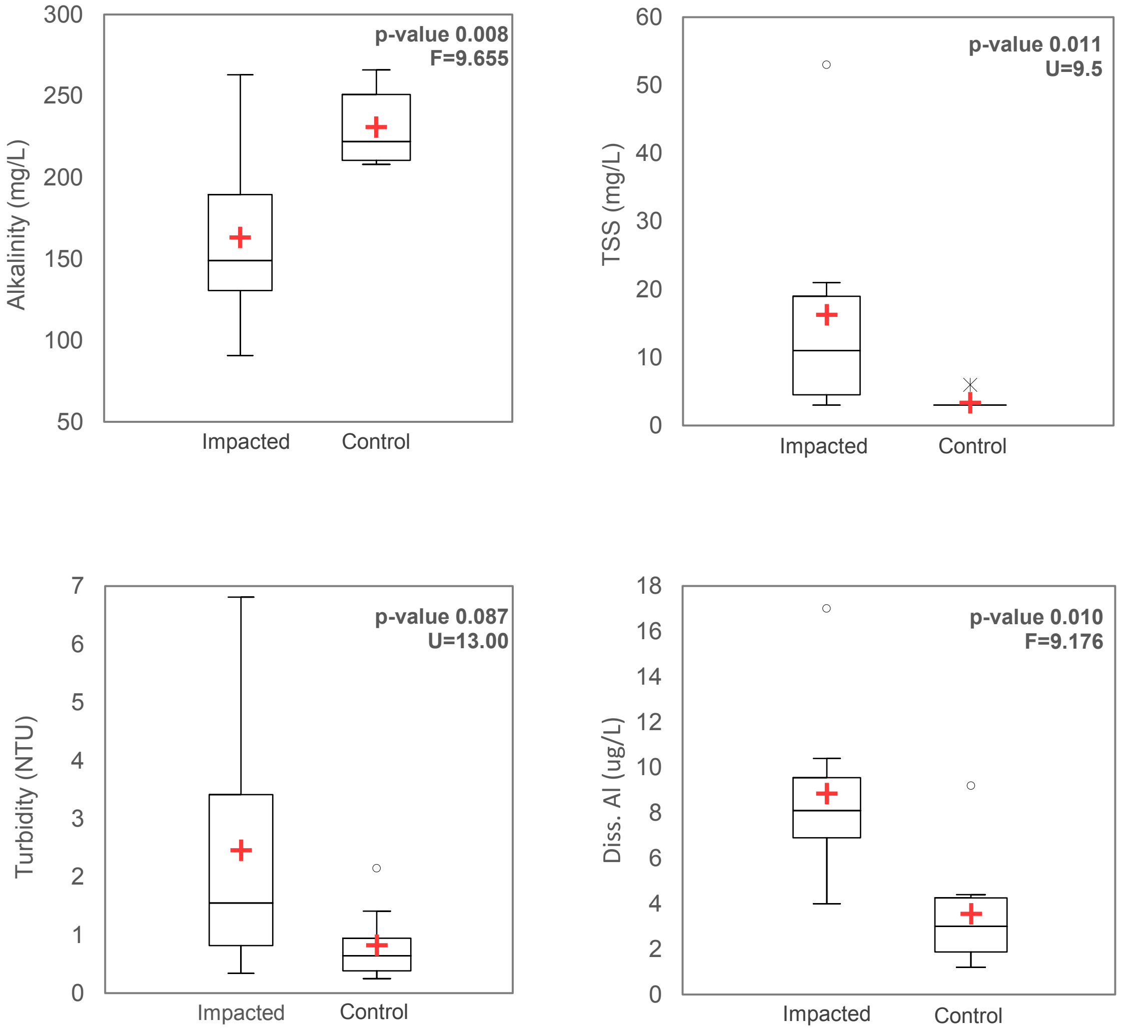

Fig. 3.5. Box and whisker plots illustrating the means and standard deviation of water quality parameters between impacted and control streams. The mean for each treatment is indicated by the red crosshair. The horizontal line within the box represents the median value while the edges of the box are the upper and lower quartiles. The whiskers represent the highest and lowest values, respectively. Outliers are represented by small circles. Alkalinity, $_{\text{dissolved}}$Al, and TSS are found to be significantly different between control and impacted streams.

The largest differences in mean relative abundance of major taxa groups was observed in Tricoptera ($RA_{control}$= 0.02±0.03; $RA_{impacted}$=0.06±0.04) and Oligochaeta ($RA_{control}$=0.04±0.05; $RA_{impacted}$=0.01±0.01). There was a high degree of variance in the relative abundances of all taxonomic groups in both impacted and control streams. The relative abundance of Odonata, Bivalvia, and Tricoptera was relatively low at all sites, regardless of stream condition. The largest difference in mean relative abundance of major functional feeding groups was observed in predators ($RA_{control}$=0.12±0.07; $RA_{impacted}$=0.18±0.05) and the collector-filterers ($RA_{control}$=0.15±0.07; $RA_{impacted}$=0.10±0.09). The relative abundance of scraper and shredder groups was similar at both control and impacted streams. Statistical tests did not indicate significant differences in the means of functional feeding groups.

Species richness was higher in impacted streams relative to control streams (Species $\text{Richness}_{control}$=36.70±9.7; Species $\text{Richness}_{impacted}$=45.80±9.1). Of the major taxa groups, the greatest difference in richness was observed in Tricoptera (Tricoptera $\text{Richness}_{control}$=3.29±2.43; Tricoptera $\text{Richness}_{impacted}$=5.40±2.30), however, no major differences in richness was observed in any taxonomic groups. In terms of functional feeding group richness, the richness of collector-gatherers was appreciably higher within impacted streams (collector-gatherers $\text{richness}_{control}$=12.86±4.60; collector-gatherers $\text{richness}_{impacted}$=18.20±5.17). No significant differences were observed for any of the richness indices calculated.

Statistical tests performed on family-level taxonomic abundances indicated significant differences in the means and distributions of 17 families in control and impacted streams (Table 3.5)

Table 3.5. Taxonomic classification, functional feeding methods, and descriptive statistics for families that demonstrated significant differences in abundance between control and impacted streams.

Family	Classification	FFG	Control		Impacted		Test Statistic	P-value
			Mean Abundance	St. Dev.	Mean Abundance	St. Dev.		
Chironomidae	Diptera	UN	16.86	11.07	4	3.54	F=6.143	0.033
Leptophlebiidae	Ephemeroptera	CG	1.42	1.51	13.4	16.56	F=9.251	0.012
Baetidae	Ephemeroptera	CG	8.14	21.12	12.6	13.36	U=5.5	0.040
Perlodidae	Plecoptera	P	0	0	0.2	0.45	U=14	<0.0001
Glossosomatidae	Tricoptera	UN	0	0	0.4	0.89	U=14	<0.0001
Leptoceridae	Tricoptera	UN	0.14	0.38	0.2	0.45	U=16.5	<0.0001
Limnephilidae	Tricoptera	CG	0	0	0.2	2.17	U=10.5	<0.0001
Staphylinidae	Coleoptera	P	0	0	0.2	0.45	U=14	<0.0001
Culicidae	Diptera	UN	0.57	1.51	0	0	U=20	<0.0001
Dixidae	Diptera	UN	0.86	2.27	0	0	U=20	<0.0001
Tipulidae	Diptera	SH	0	0	0.2	0.45	U=14	<0.0001
Aeshnidae	Odonata	UN	0	0	0.2	0.45	U=20	<0.0001
Coenagrionidae	Odonata	P	1.14	3.02	0	0	U=20	<0.0001
Planorbidae	Gastropoda	SC	1.57	4.16	0	0	U=20	<0.0001
Erpobdellidae	Annelida	P	0.14	0.38	0	0	U=20	<0.0001

FFG=Functional Feeding Group; UN=Unclassified; CG=Collector-Gatherer; P=Predator; SH=Shredder; SC=Scraper

The strongest significant correlations between biological metrics and environmental parameters was observed for Bivalvia richness and average stream depth (*r*=0.864, p=0.001) and Oligochaeta richness and wetted width (*r*=-0.860, p=0.001). Macrophyte coverage, which was found to have significantly different distributions within impacted and control streams, was significantly correlated with Odonata relative abundance (*r*=0.727, p=0.017), Odonata richness (r=0.745, p=0.013), and scraper richness (*r*=0.688, p=0.028). Interstitial substrate was significantly correlated with predator relative abundance (*r*=0.640, p=0.046).

Pearson correlation tests were performed for biological metrics and water quality values demonstrating large or significant differences between control and impacted streams. TSS, which had significantly higher mean concentrations within impacted streams, was positively correlated with total species abundance (*r*=0.825, p=0.001). Turbidity, which was found to be higher within impacted streams, was also significantly correlated with total species abundance (*r*=0.679, p=0.015), and Diptera richness (*r*=0.584, p=0.046). ${}_{\text{dissolved}}$Al, which had significantly different means between the test groups, is negatively correlated with Hilsenhoff's Biotic Index (*r*=-0.704, p=0.011), and positively correlated with Shannon's diversity index (*r*=0.583, p=0.047), species richness (*r*=0.703, p=0.011), Plecoptera richness (*r*=0.625, p=0.036), and predator richness (*r*=0.606, p=0.049). Alkalinity and Ephemeroptera richness were negatively correlated, suggesting that within the sample population, as alkalinity concentrations increased overall mayfly richness decreases (*r*=-0.580, p=0.048) (Fig. 3.7).

Principal component analyses (PCA) biplots constructed from water quality variables, environmental variables and community abundance metrics illustrated no visible separation of impacted and control streams along either the first or second axis, which explained 42.2% of variance within the data (Fig. 3.6). Negative associations were observed between relative abundances of infaunal taxa, such as the oligochaetes and bivalves, and environmental variables representing substrate size (Fig. 3.6). Positive associations were observed Plecoptera relative abundance and $_{diss}Ni$, $_{diss}Al$, in addition to Bivalvia relative abundance + Gastropoda relative abundance and TCu. Plots constructed from water quality variables, environmental variables and community richness metrics illustrated no visible separation of impacted and control streams along either the first or second axis, which explained 41.54% of variance within the data (Fig. 3.7). Species richness and Shannon diversity were strongly positively associated with richness of Ephemeroptera, Plecoptera, and Tricoptera (Fig. 3.7). Positive associations are also observed between variables reflecting increased allochthonous input (i.e. TSS, turbidity), and the richness of Diptera and collector-filterers (Fig. 3.7).

No significant correlations were identified between community metrics and time since fire (Table 3.6). However, strong correlations were detected between fire size (km^2) and several metrics, including collector-gatherer relative abundance, collector-gatherer richness, predator richness, and scraper relative abundance (Table 3.6). Scraper relative abundance is the only metric negatively correlated with fire size (r=-0.959, p=0.010).

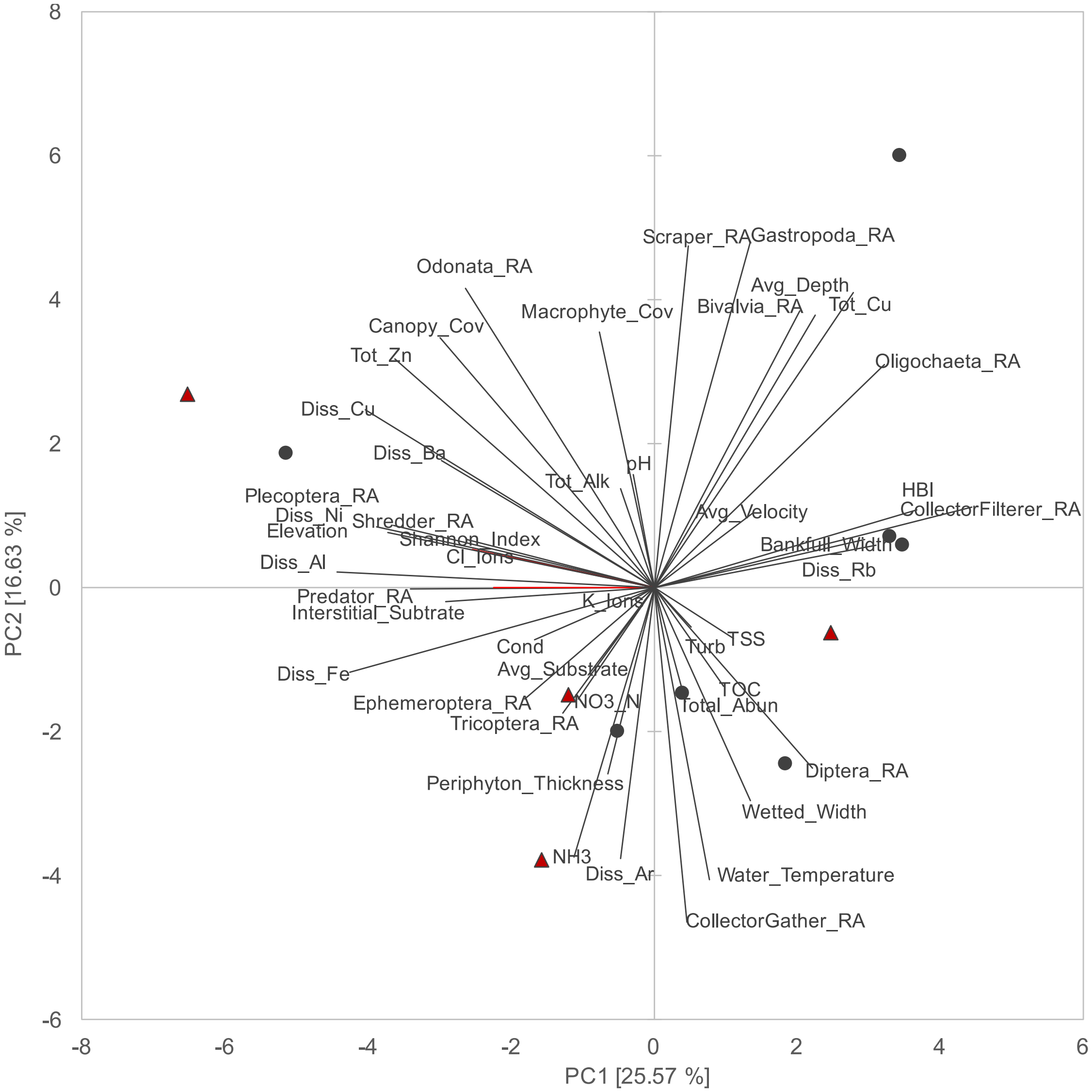

Fig. 3.6. Principal component plot illustrating associations between abundance measures and water chemistry variables for streams located within the Dehcho and South Slave regions. Strong associations are observed among turbidity, TSS, and total species abundance. There is no clear separation of sites amongst either the primary or the secondary axis. Control and impacted sites are represented by black circles and red triangles, respectively.

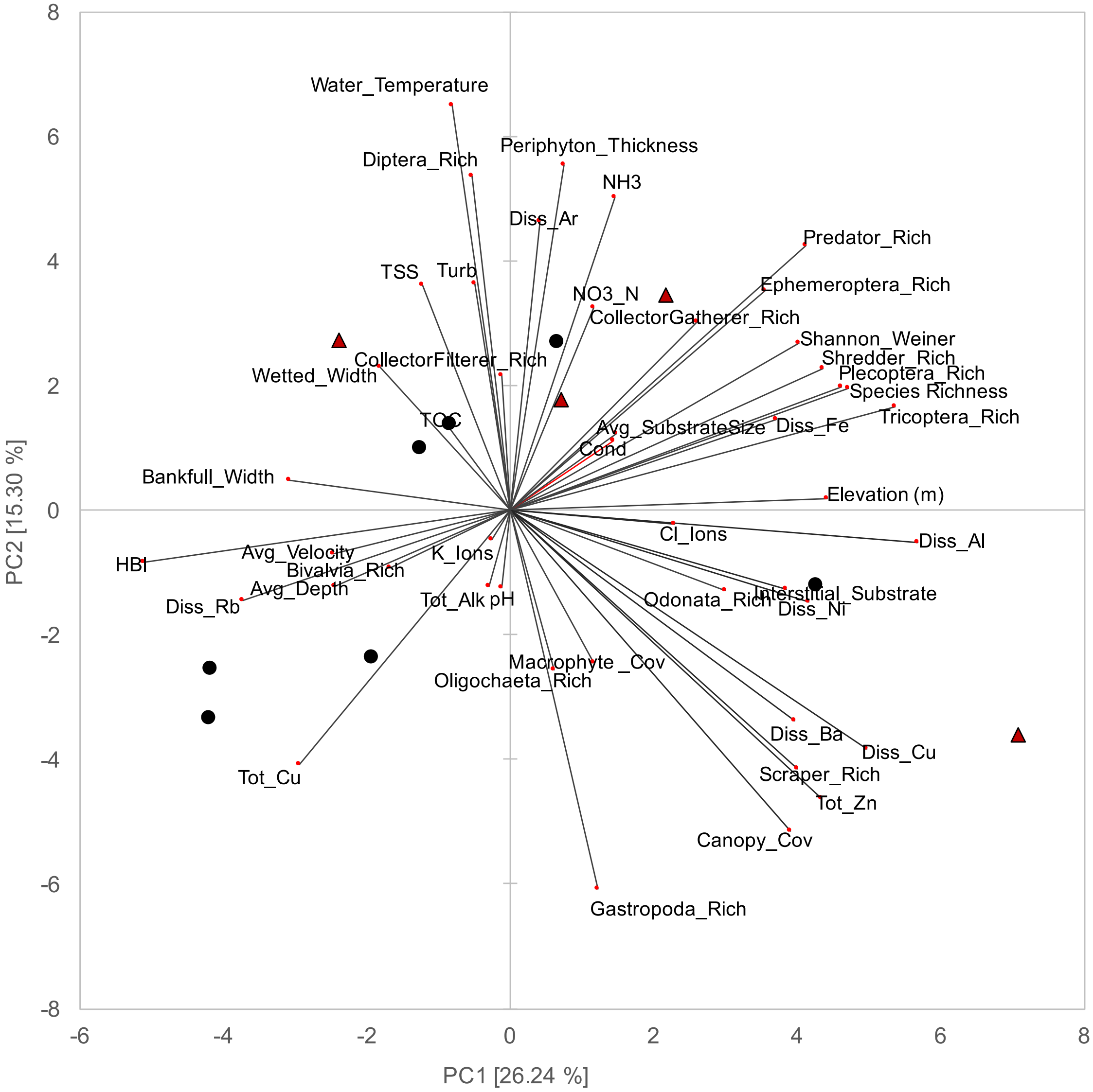

Fig. 3.7. Principal component plot illustrating associations between richness measures and water chemistry variables for streams located within the Dehcho and South Slave regions. There is no clear separation of sites amongst either the primary or the secondary axis. Control and impacted sites are represented by black circles and red triangles, respectively.

Table 3.6. Pearson correlation coefficients for functional feeding groups and fire size. Only significant (p≤0.05) are presented below.

Metrics	Size of burn (km^2)	
	Pearson's *r*	p-value
Collector-Gather RA	0.968	0.007
Collector-Gatherers Richness	0.954	0.012
Predator Richness	0.913	0.030
Scraper RA	-0.959	0.010

3.5 Discussion

This study focused on the effects of recent wildfires on streams located in the North Slave, South Slave, and Dehcho regions of the Northwest Territories using water quality parameters and benthic macroinvertebrates. Results suggested that recent wildfires that have occurred in the study regions had impacts on water chemistry and community structure of local streams. Continuous measurements of water parameters from the Cameron River, which was impacted by the Reid Lake fire in summer 2016, recorded large ‘pulses’ in several stream water chemistry variables in the months immediately following the fire (Fig. 3.2). These ‘pulses’ occurred simultaneously with increases in total monthly precipitation. These parameters included ${}_{\text{dissolved}}$Aluminum, TSS, and turbidity and reflected inputs of terrestrial material arriving in the stream from the adjacent environment due to the removal of vegetation by the fire and ash deposition.

3.5.1. Increases in allochthonous inputs may prompt the proliferation of first order consumers

The ‘pulse-like’ increases in water quality variables did not occur simultaneously and may be suggestive of different mechanisms from which allochthonous material was transported to the Cameron River following the Reid Lake fire. An increase in TDS concentrations was observed immediately following the fire, approximately two months before peak turbidity and TSS values occurred (Fig. 3.2). Turbidity and TSS values increased in September 2016 following the fire in association with increases in total precipitation (Fig.3.2). This reflected the combined impacts of vegetation loss due to the fire and subsequent catchment erosion of sediment via overland flow following large precipitation events. Increases in dissolved ions and nutrient concentrations within streams have been reported following wildfire because of input from overland flow, atmospheric deposition of ash, leaching into groundwater, and diffusion of smoke (Spencer and Hauer, 1991; Earl and Blinn, 2003; Son et al., 2015). Significantly higher values of TSS and ${}_{\text{dissolved}}\text{Al}$, as well as large increases in turbidity, were also observed during point measurements within impacted streams in the South Slave and Dehcho regions (Fig. 3.7).

Previous studies have indicated changes in the relative abundance and diversity of different functional feeding groups as a result of fluctuations in inputs of FPOM and CPOM (Wood and Armitage, 1997; Oliver et al., 2012; Buendia et al., 2013). Increases in the abundances of filtering and burrowing taxa, such as Hydropsychidae, Simuliidae, Bivalvia, and Oligochaeta, in the Cameron River following the Reid Lake fire may be a result of increased availability of fine particulate organic matter, reflected by amplified TSS and turbidity values. However, a greater number of samples would be required to confirm these assumptions as water levels and discharge within the Cameron River also varied between 2015 and 2016. Variations in discharge could have driven some of the differences observed in the benthic community metrics between the two study years. For example, Simuliidae larvae generally prefer fast moving waters (Meissner et al., 2009), and the abundance of Simuliidae larvae were higher in 2016 when water level and discharge of the Cameron River were also higher compared to 2015. It is reasonable that Simuliidae, which have relatively high-fecundity, would have higher relative abundance in an impacted stream relative to an undisturbed stream environment. However, the lack of replicate samples for this stream prevents any statistical tests from being conducted and makes conclusions about the proliferation of Simuliidae within the Cameron River in 2016 difficult to interpret.

Mean invertebrate abundance was higher for impacted streams in the Dehcho and South Slave regions and the Cameron River following fire events. This is consistent with other studies that found increases in macroinvertebrate abundances and shifts in community structure following high-intensity fires in response to changes in the amount and quality of allochthonous material entering streams (Malison and Baxter, 2010; Verkaik et al., 2015). Positive correlations between TSS, turbidity, and species abundance supports the assumption that greater inputs of particulate organic matter can increase the abundance of invertebrate groups such as the collector gatherers and their predators within streams traversing through burned catchments.

An increase of collector-filterer taxa was observed within the Cameron River in 2016 that was not detected in impacted streams within the Dehcho and South Slave regions (Fig. 3.3). The collector-gatherers demonstrated greater richness and abundance in streams flowing through recently burned areas compared to streams where recent fires were not observed. The families Leptophlebiidae, Baetidae, and Limnephilidae, which exhibit collector-gatherer feeding behaviors, had significantly larger abundances in impacted streams compared to control streams (Table 3.5). These observations and findings are similar to those of Oliver et al. (2012) and Ramchunder et al. (2013) and may be related to functional feeding strategies. The collector-gatherers are opportunistic feeders and can adapt to changes in the availability of food resources better than other functional feeding groups, including the shredders and scrapers (Minshall, 2003; Oliver et al., 2012). Positive correlations between collector-gatherer richness and relative abundance with fire size supports the conclusion that collector-gatherers are better adapted to thrive in post-fire landscapes as opposed to other functional feeding groups (Table 3.6).

3.5.2. Stream nutrient concentrations and primary productivity

Recent wildfires did not appear to impact stream nutrient concentrations (TOC, NO_3^-, NH_3) in either the Cameron River or streams within the Dehcho and South Slave regions. Previous studies indicate that wildfire can have variable impacts on nutrient concentrations, with some studies reporting increased nutrient concentrations after wildfires (Earl and Blinn, 2003; Ramchunder et al., 2013; Klose et al., 2015; Son et al., 2015). In general, boreal forest systems are N-limited (Lupi et al., 2013) and uptake of nitrogen by autotrophs is constrained by its limited availability or slow cycling processes. This, coupled with the relatively low volatilization point for nitrogen (Debano, 1990), may suggest that nutrient exports to streams resulting from wildfire could be far less for northern areas compared to those reported for temperate locations. The lack of change in periphyton cover may also suggest that there was no change in nutrient concentrations following the fire event. However, periphyton cover was recorded on a visual basis and offers limited insight with regards to potential changes in stream productivity. Low scraper abundance and richness within impacted samples also suggests minimal changes in primary productivity within test groups. Fluxes in nutrient concentrations due to wildfires are generally short in duration (Earl and Blinn, 2003). Therefore, it is possible that the single point-in-time method used to collect water chemistry samples in this study was inefficient at measuring brief changes in nutrient concentrations. Primary productivity in streams traversing burned catchments can be reduced through increased input of fine-sediment resulting from catchment erosion and post-fire flood events. Increases in suspended particles within the water column reduces the amount of sunlight available for photosynthesis (Klose et al., 2015). In addition, increased macroinvertebrate abundance can result in increased periphyton consumption (Malison and Baxter, 2010).

With respect to aquatic macrophytes, non-parametric statistical tests revealed significant differences in macrophyte cover between control and impacted streams. It is plausible that higher concentrations of suspended particulates, reflected by greater TSS and turbidity values, may have reduced macrophyte cover in impacted streams by reducing light penetration through the water column (Schutten et al., 2005). Care must be taken when drawing conclusions about macrophyte cover data as these values were initially recorded as broad percent ranges. Therefore, large variation in actual macrophyte cover may have existed between streams receiving the same rank.

3.5.3. Wildfire impacts on the abundance and richness of invertebrate groups

Invertebrates of the orders Ephemeroptera, Plecoptera, and Tricoptera (EPT) are commonly described as sensitive to ecosystem disturbances and have been found to decline in abundance within burned catchments (Minshall, 2003). However, large variations in functional feeding strategies, fecundity, and tolerance to changes in water quality exist at lower taxonomic levels within these orders. EPT relative abundance and richness was greater within impacted streams compared to control streams. It is possible that generalist groups within these orders, such as the collector-gatherers, responded to greater food availability resulting from additional allochthonous input in the form of FPOM from the surrounding burned catchment. Leptophlebiidae and Baetidae (Ephemeroptera), which both exhibit collector-gatherer feeding mechanisms, had significantly higher abundances in impacted streams within the Dehcho and South Slave regions relative to control streams (Table 3.5). Similar findings were cited by Brown et al. (2013). Minshall (2003) attributed high-ingestion rates of Leptophlebiidae as a mechanism for overcoming poor-quality food resources found within stream ecosystems traversing burned catchments. Families belonging to the orders Tricoptera (Glossosomatidae, Leptoceridae, Limnephilidae) and Plecoptera (Perlodidae) also exhibited significantly greater abundances within impacted streams and suggests that certain members of these 'sensitive' orders are able to persist in stream environments impacted by wildfire (Table 3.5).

Higher relative abundances and richness of epifaunal taxa (e.g. EPT, Odonata) within impacted streams is likely also a function of greater habitat availability. Logarithmic means of interstitial substrate categories were significantly higher within impacted streams, typically falling within the small pebble category. Interstitial substrate within control streams was finer, falling within the coarse sand category. Finer interstitial material may also explain the higher relative abundance of oligochaetes, an infaunal group, within control streams. Although wildfire likely did not affect the size class of interstitial substrate, differences in suitable habitat within control and impacted streams in addition to changes in food availability, may synergistically affect benthic community structure.

In regard to reproductive strategies, the greater abundance of r-strategist taxa (relatively high-fecundity groups), namely Simuliidae, Chironomidae, and Baetidae, observed within the Cameron River following the Reid Lake fire, has also been observed by others in both peatlands (Ramchunder et al., 2013) and Mediterranean climates

(Verkaik et al., 2015). These taxa rely on rapid reproductive turnover and high dispersal methods for repopulation in disturbed ecosystems (Minshall, 2003).

3.6. Conclusion

The results of this study suggested that recent wildfires within the southern Northwest Territories led to short-term impacts to water quality through increased input of fine sediment and allochthonous material due to increased erosion from the surrounding catchment. In addition, differences in the abundance and richness of functional feeding groups between impacted and control sites suggest that wildfire increases the input of FPOM, increasing the richness and abundance of primary consumer groups and their predators.

This was the first study to analyze the impacts of recent wildfires on stream invertebrate assemblages in subArctic regions of Canada. The results of this study highlight the multiple stressors caused by wildfire and the difficulty separating these from concurrent climate stressors such reduced precipitation. This is especially true when considering the Cameron River, where changes in water level and discharge may have caused major differences in stream invertebrate communities. Although many similar studies cite changes in nutrients and primary productivity following wildfire, the results of this study suggest that these changes may not be a major issue for streams traversing stands of nutrient-poor boreal forest. The results of this study also highlight that the impacts of wildfire to stream ecosystems are, at minimum, short-lived, regardless of fire size.

This study underscores the physical and ecological complexity of boreal stream ecosystems, and their resilience to changes resulting from disturbance. The results also highlight the high degree of variation between streams within a relatively small geographic area resulting from differences in underlying geology, soil types, and surrounding vegetation.

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Chapter 4

Discussion and Conclusions

4.1 Changes in Water Quality

Climate change is expected to affect natural disturbance regimes within both terrestrial and aquatic systems. In Northwest Territories, deviations from typical climate patterns, such as extended periods of increased air temperatures and reduced precipitation, are likely contributing to recent severe wildfire seasons. This research investigates the impacts of recent wildfires on smaller order streams within the southern regions of Northwest Territories. While it is tempting to simply discuss the scientific ramifications of this research, it is also important to highlight the potential implications for stream ecosystem functioning and community health. This is especially true when considering the demographics of the study region where isolated communities, such as Kakisa, have a deeper reliance and spiritual connection with the surrounding environment relative to urban areas.

As mentioned in earlier chapters, wildfires indirectly have the ability to impact community health through reducing the quality of water resources. Most countries have water quality standards that specify limitations for chemical and physical water quality variables based on aesthetic preferences and effects on human health. This research concludes that wildfire does impact water quality at short timescales. Specifically, parameters associated with increased terrestrial input, such as turbidity, were the most different between control and impacted streams. Comparisons of water quality parameters included in this study with their respective national drinking water guidelines indicate that turbidity levels exceed limits recommended by Health Canada (2012) (Table

4.1). Turbidity values recorded for the Cameron River following the Reid Lake fire, and the average turbidity value for the Dehcho-South Slave comparison exceeds 2 NTU; which twice the suggested limit of 1 NTU (Health Canada, 2012). Although increased turbidity values do not directly imply increased risk to human health, increased suspended particulates within the water column can promote microbial growth by preventing disinfection methods from properly removing colony-forming units (Health Canada, 2012). Increased turbidity within the water column is also unfavourable for aesthetic reasons, especially where resources are used for recreational purposes. Although community health and well-being are not among the principal objectives outlined in the introduction of this document, they are not exclusive. In fact, the methodologies used this study could be implemented in research focussing on the potential impacts of wildfire on community and ecosystem health in fire-effected regions. This supports the claim made earlier that benthic macroinvertebrates have great utility as indicators of environmental change and can be used in multiple research disciplines.

Table 4.1 Water quality variables and their associated drinking water guidelines (Health Canada, 2017). Parameters that were found to be the most different between control and impacted streams are bolded.

Water Quality Parameter	Drinking Water Guideline	Adverse Impacts to Human Health
Turbidity	**Generally, ≤1.0 NTU***	**Particles may prevent proper disinfection**
Aluminum	**None cited**	**None cited**
Ammonia	None cited	None cited
Arsenic	0.01 mg/L	Carcinogen, neurological effects
Barium	1.0 mg/L	Cardiovascular effects
Chloride	None cited	None cited
Copper	None cited	None cited
Alkalinity	**None cited**	**None cited**
Iron	None cited	None cited
Nitrate as Nitrogen	10 mg/L	Toxic to bottle-fed infants, carcinogen
Selenium	0.05 mg/L	Selenosis, neurological effects
pH	None cited	None cited
Zinc	None cited	None cited

*Dependent on filtering method

4.2 Changes in Trophic Relationships

Shifts in trophic relationships can result from changes in water quality. As previously mentioned, it is important to understand how wildfire may impact these relationships, especially since many communities within the study region depend on the services aquatic systems provide.

This study identified variations in the abundance and functional ecology of consumer invertebrate groups between reference and burned streams. Other studies have cited increased abundance of low-order invertebrate consumers following wildfire (Malison and Baxter, 2010a; Verkaik et al., 2015). These results suggest that wildfire can induce changes to low-level trophic relationships. Aquatic macroinvertebrates are also a primary food source for higher order consumers. The results of this research found that impacted streams within the Dehcho and South Slave regions had greater predator species richness, which may have been a response to a greater abundance of prey taxa. Similarly, Malison and Baxter (2010a), also found a greater emergence of predatory insects from streams within burned versus unburned catchments. Others have cited wildfire-induced changes in the abundance and feeding habits of larger, more complex consumers such as spiders, bats, and fish (Koetsier et al., 2007; Malison and Baxter, 2010b). Given that wildfire induces change at multiple trophic levels within stream ecosystems, it is recommended that future studies examine the impacts of recent wildfires on fauna important to northern community welfare, such as pickerel and other fish species. In close, this research finds that boreal streams within the Northwest Territories most likely respond to environmental changes resulting from recent wildfires. Changes in water quality and stream ecology are intimately linked through complex processes, such as changes in the quality and availability of food resources.

4.3 Research Limitations and Future Recommendations

Several limitations were encountered during this study, but these were generally a product of the remoteness of the study area and adverse climate conditions. These limitations included logistical constraints and low water levels. Therefore, fewer streams were sampled than initially anticipated (>30). One method to mitigate the issue of low sample size is to upload or share data suites via publicly accessible databases for use by other researchers. CABIN, Northwest Territories Discovery Portal, and Mackenzie Data Stream databases allow for the storage and sharing of environmental, biological, and water quality data for the Northwest Territories. Unfortunately, there was no other publicly shared data with similar water quality suites that could be used to increase sample size. However, this just emphasizes the fact that more frequent data sharing and uploading for these remote regions needs to occur.

There are several ways that this research could be developed further. For example, more detailed and quantitative comparison of primary producer communities (aquatic plants, algae, etc.) between control and impacted streams could expand current understanding of how physical and chemical changes within a stream impacts primary consumers. Determining the taxonomy and concentrations of primary producers' present can also provide insight as to why certain proportions or abundances of higher trophic organisms or functional feeding groups exist within the study area. Analysis could also be extended to include higher-order consumers from the adjacent terrestrial environment, such as bats, whom feed off emerging insects from stream environments. Malison and

Baxter (2010) used similar multi-trophic indicators (algae, macroinvertebrates, spiders, and bats) to measure the response of both lentic and riparian ecosystems to recent wildfires within Payette National Forest in Idaho.

This study provides preliminary evidence for the impact of wildfire on streams in the southern regions of the Northwest Territories. This study concludes that wildfires likely causes at least short-term perturbations in both water quality and stream macroinvertebrate communities. In addition, the research presented here highlights the usefulness of benthic macroinvertebrates as bio-indicators of ecosystem change in high-latitude regions in response to natural disturbances such as wildfire.

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List of Appendices

I Transformations applied to non-normal datasets.
II Comparison of descriptive statistics for stream environment data located in the Dehcho and South Slave regions, NWT.
III Comparison of ANOVA statistics for stream environment data located in the Dehcho and South Slave regions, NWT.
IV Comparison of Mann-Whitney statistics for stream environment data located in the Dehcho and South Slave regions, NWT. Significant results (a=0.05) are bolded.
V Comparison of descriptive statistics for water quality of streams located in the Dehcho and South Slave regions, NWT.
VI Comparison of ANOVA statistics for water quality of streams located in the Dehcho and South Slave regions, NWT. Parameters with significantly (a=0.05) different means between treatments display bolded p-values.
VII Comparison of Mann-Whitney statistics for water quality of streams located in the Dehcho and South Slave regions, NWT. Parameters with significantly (a=0.05) different distributions between treatments display bolded p-values
VIII Comparison of descriptive statistics for stream invertebrate community metrics located in the Dehcho and South Slave regions, NWT.
IX Comparison of ANOVA statistics for stream invertebrate community metrics located in the Dehcho and South Slave regions, NWT. Parameters with significantly (a=0.05) different means between treatments display bolded p-values.
X Comparison of Mann-Whitney statistics for stream invertebrate community metrics located in the Dehcho and South Slave regions, NWT. Significant results (a=0.05) are bolded.
XI Comparison of ANOVA statistics for family-level taxonomic groups in streams located in the Dehcho and South Slave regions, NWT. Parameters with significantly (a=0.05) different means between treatments display bolded p-values.
XII Comparison of Mann-Whitney statistics for family-level taxonomic groups in streams located in the Dehcho and South Slave regions, NWT. Parameters with significantly (a=0.05) different means between treatments display bolded p-values.

XIII Description of metrics included used to compare invertebrate communities between control and impacted streams
XIV Raw environmental and reach observations. Due to logistical constraints, only water chemistry was collected from KVB-01, KVB-03, and FSC-02.
XV Raw water quality observations for streams located in the Dehcho and South Slave regions.
XVI Raw benthic invertebrate counts for streams located in the Dehcho and South Slave regions.

Appendix Material

Appendix I

Transformations applied to non-normal datasets.

	Variable	Transformation
Environmental	Average Depth	$\text{Log}_{10}(x+1)$
Water Quality	Interstitial Substrate Category	$\text{Log}_{10}(x+1)$
	NH_3	$\sqrt{x}$
	Cl^-	$\text{Log}_{10}(x+1)$
Ecological Metrics	Total Abundance	$\text{Log}_{10}(x+1)$
	Ephemeroptera RA	$\sqrt{x}$
	Gastropoda RA	$\sqrt{x}$
	Gastropoda RA	$\sqrt{x}$
	Scraper RA	$\sqrt{x}$
	Diptera RA	$\sqrt{x}$
	Leptophlebiidae Abundance	$\text{Log}_{10}(x+1$

Appendix II

Comparison of descriptive statistics for stream environment data located in the Dehcho and South Slave regions, NWT.

Parameter	Control				Impacted			
	Min	Max	Mean	St.Dev	Min	Max	Mean	St.Dev
Water Temp. (°C)	11.82	16.20	13.63	1.74	11.44	17.49	14.50	2.24
Elevation (m)	130.00	246.00	197.71	40.38	216.00	277.00	237.80	26.39
Average Substrate Size (cm)	0.05	18.34	7.93	5.91	6.66	9.51	7.67	1.09
Canopy Coverage	1.00	3.00	1.71	0.76	0.00	4.00	1.40	1.57
Average Depth (cm)	5.08	35.00	16.63	9.052	7.00	172.30	43.86	72.03
Average Velocity (m/s)	0.28	0.64	0.48	0.13	0.08	0.63	0.30	0.24
Interstitial substrate	1.00	3.00	1.57	0.79	2.00	5.00	3.00	1.41
Wetted Width (m)	1.23	53.51	10.54	19.08	1.46	6.63	3.72	1.99
Bank-full Width (m)	1.41	53.51	12.41	18.33	3.40	8.65	5.06	2.33
Macrophyte Cover	0	1	0.57	0.53	0	1	0.4	0.55
Periphyton Thickness	1	5	1.86	1.46	2	4	2.8	1.10

Appendix III

Comparison of ANOVA statistics for stream environment data located in the Dehcho and South Slave regions, NWT.

Parameter	F	p-value	95 % Confidence Interval			
			Control		Impacted	
			Lower	Upper	Lower	Upper
Water Temp. (°C)	0.577	0.465	11.99	15.281	12.556	16.452
Elevation (m)	3.728	0.082	167.856	227.573	202.471	272.129
Avg. Substrate Size (cm)	0.010	0.923	4.038	11.831	3.055	12.275
Canopy Coverage (rank)	0.228	0.643	0.768	2.661	0.285	2.526
Average depth (cm)	0.026	0.876	0.979	1.417	0.915	1.432
Avg. Velocity (m/s)	2.557	0.144	0.329	0.624	0.108	0.499
Interstitial Substrate Category	5.380	**0.046**	0.283	0.506	0.436	0.731

Appendix IV

Comparison of Mann-Whitney statistics for stream environment data located in the Dehcho and South Slave regions, NWT. Significant results (α=0.05) are bolded.

Parameter	U	p-value
Wetted width (m)	24.5	0.285
Bank-full width (m)	27	0.136
Macrophyte Coverage (%)	20.5	**<0.0001**
Periphyton Thickness (%)	11	0.374

Appendix V

Comparison of descriptive statistics for water quality of streams located in the Dehcho and South Slave regions, NWT.

Parameter	Control				Impacted			
	Min	Max	Mean	St.Dev	Min	Max	Mean	St.Dev
NH3	0.006	0.018	0.01	0.0040	0.009	0.018	0.01	0
TOC	14.5	33.4	23.18	6.23	13.1	40.6	26.86	10.69
Alkalinity	208	266	231	24.20	90.7	263	163.24	56.24
Conductivity	396	585	448.88	59.78	187	639	444.43	147.68
TSS	3	6	3.38	1.06	3	53	16.29	17.59
Turbidity	0.25	2.14	0.82	0.65	0.34	6.81	2.45	2.41
pH	7.54	8.34	7.97	0.23	7.44	8.19	7.83	0.24
Cl-	0.23	1.49	0.66	0.42	0.23	1.23	0.65	0.31
NO3-	0.04	0.28	0.12	0.083	0.05	0.21	0.16	0.05
K+	0.2	1.3	0.65	0.34	0.3	1.4	0.80	0.42
Diss. Al	1.2	9.2	3.56	2.57	4	17	8.86	4.13
Diss. As	0.2	0.4	0.28	0.01	0.2	0.6	0.40	0.16
Diss. Ba	33.8	167	66.5	44.10	24.9	216	61.39	69.04
Diss. Cu	0.2	0.6	0.25	0.14	0.2	0.9	0.37	0.28
Diss. Fe	33	173	83.25	47.96	49	158	103.57	40.26
Diss. Ni	0.2	1	0.43	0.24	0.2	0.9	0.60	0.23
Diss. Rb	0.5	1.5	1	0.34	0.5	1.3	0.81	0.32
Tot. Zn	0.4	2.8	0.9	0.82	0.4	4.5	1.26	1.47
Tot. Cu	0.2	2.3	0.73	0.76	0.2	1.5	0.44	0.48

Appendix VI

Comparison of ANOVA statistics for water quality of streams located in the Dehcho and South Slave regions, NWT. Parameters with significantly (a=0.05) different means between treatments display bolded p-values.

Parameter	F	p-value	95 % Confidence Interval			
			Control		Impacted	
			Lower	Upper	Lower	Upper
NH_3	3.167	0.099	0.007	0.013	0.01	0.017
TOC	0.687	0.422	16.617	29.733	19.846	33.868
Alkalinity	9.655	**0.008**	198.818	263.182	128.838	197.647
Conductivity	0.006	0.939	365.237	532.513	355.016	533.842
pH	1.428	0.253	7.793	8.452	7.635	8.019
Cl^-	0.175	0.683	0.313	12.512	-1.835	11.207
NO_3	1.324	0.271	0.064	0.173	0.103	0.22
K^+	0.587	0.457	0.361	0.939	0.491	1.109
DAl	9.176	**0.010**	0.983	6.142	6.099	11.615
DFe	0.776	0.394	49.207	117.293	67.177	139.965
DNi	2.022	0.179	0.243	0.607	0.406	0.794
DRb	1.172	0.299	0.747	1.253	0.544	1.085

Appendix VII

Comparison of Mann-Whitney statistics for water quality of streams located in the Dehcho and South Slave regions, NWT. Parameters with significantly (a=0.05) different distributions between treatments display bolded p-values

Parameter	U	p-value
TSS	9.5	**0.011**
Turbidity	13	0.094
Diss. As	15.5	0.126
Diss. Ba	41	0.152
Diss. Cu	19.5	0.103
Total Zn	23.5	0.614
Total Cu	32	0.646

Appendix VIII

Comparison of descriptive statistics for stream invertebrate community metrics located in the Dehcho and South Slave regions, NWT.

	Control				Impacted			
	Min	Max	Mean	St.Dev	Min	Max	Mean	St.Dev
Hilsenhoff biotic index	4.64	6.58	5.72	0.82	3.49	6.25	4.77	1.04
Shannon-Weiner	0.92	1.37	1.18	0.16	1.15	1.43	1.31	0.11
Species richness	22	48	37	10	26	54	46	6
Total abundance	1393	3965	2418	848	1077	7560	3832	2749
Ephemeroptera RA	0.00	0.39	0.13	0.15	0	0.42	0.15	0.16
Ephemeroptera richness	0.00	6	3.57	2.23	0	11	5.4	3.29
Diptera RA	0.27	0.82	0.54	0.23	0.37	0.79	0.55	0.19
Diptera richness	13	27	19.29	5.53	10	27	19.8	6.76
Oligochaeta RA	0.00	0.11	0.04	0.05	0	0.03	0.01	0.01
Oligochaeta richness	0.00	3	1.14	1.07	0	3	1.4	1.14
Plecoptera RA	0.00	0.27	0.07	0.11	0.00	0.22	0.08	0.104
Plecoptera richness	0.00	4	1.71	1.70	0	4	2.4	1.51
Tricoptera RA	0	0.06	0.02	0.03	0	0.12	0.06	0.04
Tricoptera richness	0	6	3.29	2.43	0	8	5.4	2.3
Gastropoda RA	0	0.33	0.09	0.11	0.01	0.1	0.06	0.03
Gastropoda richness	1	4	3	1.15	2	4	2.6	0.89
Bivalvia RA	0	0.146	0.020	0.055	0	0.094	0.0330	0.046
Bivalvia richness	0	2	0.286	0.756	0	2	0.6	0.894
Odonata RA	0	0.01	0	0.004	0	0.04	0.01	0.017
Odonata Richness	0	2	0.57	0.786	0	2	0.80	0.84
Collector-filterer RA	0.06	0.27	0.15	0.07	0.01	0.24	0.10	0.09
Collector-filterer richness	3	7	4.71	1.50	2.00	6.00	4.20	1.48
Collector-gatherer RA	0.31	0.55	0.45	0.09	0.31	0.68	0.48	0.16
Collector-gatherers richness	6	19.00	12.86	4.60	11	27.00	18.20	5.17
Predator RA	0.05	0.21	0.12	0.07	0.05	0.25	0.18	0.05

Predator richness	5.00	12.00	8.57	2.44	9.00	14.00	10.60	2.07
Scraper RA	0.03	0.25	0.9	0.08	0.06	0.10	0.09	0.01
Scrapers richness	3.00	6.00	4.29	1.38	2.00	8.00	4.40	2.61
Shredder RA	0	0.21	0.10	0.10	0	0.2	0.09	0.08
Shredder richness	0.00	4.00	2.71	1.70	1.00	6.00	3.40	1.82

Appendix IX

Comparison of ANOVA statistics for stream invertebrate community metrics located in the Dehcho and South Slave regions, NWT. Parameters with significantly (a=0.05) different means between treatments display bolded p-values.

			95 % Confidence Interval			
	F	*p-value*	Control		Impacted	
			Lower	Upper	Lower	Upper
Hilsenhoff biotic index	3.139	0.107	4.95	6.49	3.86	5.68
Shannon-Weiner index	2.472	0.147	1.06	1.30	1.17	1.45
Species richness	3.438	0.093	29.67	43.76	37.46	54.14
Total abundance	0.681	0.429	3.153	3.570	3.235	3.727
Diptera RA	0.030	0.866	0.596	0.845	0.588	0.883
Diptera richness	0.021	0.887	14.189	24.382	13.770	25.830
Ephemeroptera RA	0.145	0.711	.0.104	0.475	0.119	0.558
Ephemeroptera richness	1.338	0.274	1.297	5.845	2.709	8.091
Oligochaeta Richness	0.160	0.698	0.218	2.068	0.306	2.464
Plecoptera richness	0.829	0.384	-0.42	4.99	0.80	7.20
Tricoptera RA	4.839	0.052	-0.004	0.049	0.032	0.095
Tricoptera richness	2.302	0.160	1.282	5.290	3.029	7.771
Gastropoda RA	0.028	0.871	0.128	0.370	0.092	0.379
Gastropoda richness	0.417	0.533	2.109	3.891	1.545	3.655
Collector-filterer RA	1.411	0.262	0.088	0.221	0.021	0.179
Collector-filterer richness	0.347	0.569	3.46	5.97	2.71	5.69
Collector-gatherer RA	0.221	0.649	0.347	0.553	0.362	0.605
Collector-gatherer richness	3.563	0.088	8.79	16.93	13.38	23.02
Predator RA	2.006	0.187	0.070	0.178	0.113	0.241
Predator richness	2.268	0.163	6.63	10.51	8.31	12.89
Scraper RA	0.093	0.766	0.193	0.356	0.195	0.389
Scrapers richness	0.010	0.923	2.63	5.94	2.44	6.36
Shredder richness	0.448	0.519	1.24	4.19	1.66	5.14

Appendix X

Comparison of Mann-Whitney statistics for ecological metric data located in the Dehcho and South Slave regions, NWT. Significant results (α=0.05) are bolded.

Parameter	U	p-value
Odonata RA	22	0.444
Odonata richness	20.5	0.545
Plecoptera RA	19.5	0.768
Bivalvia RA	21	0.318
Bivalvia richness	21.5	0.318
Oligochaeta RA	13	0.505
Shredder RA	17	1.000

Appendix XI

Comparison of ANOVA statistics for family-level taxonomic groups in streams located in the Dehcho and South Slave regions, NWT. Parameters with significantly (a=0.05) different means between treatments display bolded p-values.

			95 % Confidence Interval			
	F	*p-value*	Control		Impacted	
			Lower	Upper	Lower	Upper
Chironomidae	6.143	**0.033**	9.396	24.318	-4.828	12.828
Leptophlebiidae	9.251	**0.012**	0.013	0.616	0.596	1.309

Appendix XII

Comparison of Mann-Whitney statistics for family-level taxonomic groups in streams located in the Dehcho and South Slave regions, NWT. Parameters with significantly (a=0.05) different means between treatments display bolded p-values.

Taxa	U	p-value
Baetidae	5.5	**0.040**
Heptageniidae	20	0.747
Capniidae	17	0.636
Chloroperliidae	18.5	0.937
Nemouridae	16	0.836
Perlodidae	14	**<0.0001**
Brachycentridae	16	0.636
Glossomatidae	14	**< 0.0001**
Hydropsychidae	26	0.141
Hydroptilidae	8	0.141
Leptoceridae	16.5	**< 0.0001**
Limnephilidae	10.5	**< 0.0001**
Elmidae	15	0.848
Staphylinidae	14	**< 0.0001**
Ceratopogonidae	13	0.318
Culicidae	20	**< 0.0001**
Dixidae	20	**< 0.0001**
Empididae	13.5	0.495
Simuliidae	23.5	0.394
Tipulidae	14	**< 0.0001**
Corixidae	20	**< 0.0001**
Aeshnidae	20	**< 0.0001**
Corduliidae	16	0.636
Coenagrionidae	20	**< 0.0001**
Gomphidae	16	0.636
Pisidiidae	14	0.318
Planorbidae	20	**< 0.0001**
Erpobdellidae	20	**< 0.0001**
Naididae	20	0.636

Appendix XIII

Description of metrics included used to compare invertebrate communities between control and impacted streams.

Metric	Description	Formula
Species abundance	Total number of species within sample	Σ Abundances for all taxa
Relative abundance**	The contribution of a taxa relative to total species abundance within a sample. Used to demonstrate the rarity/dominance of different taxonomic groups	Taxa abundance/total species abundance *100
Richness	The total number of different taxonomic groupings within the sample	# Different taxa
Hilsenhoff biotic index)	The tolerance of a community to disturbances. Values are based on weighted based on relative abundance of taxa groups and their known tolerance scores.	Hilsenhoff Biotic Index = $[\Sigma (n_i)(a_i)]/N$ n_i = relative abundance of taxa a_i = tolerance value of taxa N= species abundance
Shannon-Weiner diversity index (H')	A measure of diversity based on the richness and evenness of the community	$H' = -\Sigma(p_i)(\ln p_i)$ p_i= richness of taxa (i)/community richness

** Can be reported as either a fraction or a percentage

Appendix XIV

Raw environmental and reach observations. Due to logistical constraints, only water chemistry was collected from KVB-01, KVB-03, and FSC-02.

Site	Sample Year	Condition	Water Temperature (°C)	Elevation (m)	Canopy Coverage (%)	Macrophyte Coverage (%)	Periphyton Thickness (mm)	Average Depth (cm)	Wetted Width (m)	Bank-full Width (m)	Average Velocity (m/s)	Interstitial Substrate Size Class (cm)	Average Substrate Size (cm)
BF-01	2015	UN	12.80	162	0	2	1	31.83	8.90	11.30	0.00	<0.1	5.13
BOC-01	2015	UN	13.25	172	1	1	2	11.33	2.60	6.05	0.00	<0.1	8.44
CH-01	2015	UN	11.82	178	2	0	1	13.67	3.80	5.90	0.42	<0.1	5.54
CR-01	2015	UN	13.86	213	1	1	2	22.90	15.80	16.25	0.66	0.1-0.2	10.29
KR-01	2015	UN	17.74	206	0	1	2	N/A	77.91	77.91	0.44	<0.1	5.77
KU-01	2015	UN	12.59	195	2	1	1	35.00	1.30	9.60	0.53	<0.1	4.76
MO-01	2015	UN	14.96	178	1	0	1	31.20	10.85	10.85	0.83	<0.1	15.00
ST-01	2015	UN	14.81	159	1	2	4	8.70	10.60	21.30	0.15	<0.1	9.96
WS-01	2015	UN	12.07	179	1	1	1	14.33	3.85	4.90	0.54	0.1-0.2	6.35
FSB-01	2016	BU	11.44	277	4	1	2	7.00	1.23	1.41	0.08	3.2-6.4	7.40
FSB-01	2016	BU	11.44	277	4	1	2	7.00	1.23	1.41	0.08	3.2-6.4	7.40

FSB-03	2016	BU	17.49	252	0	0	1	172.30	39.38	53.51	N/A	0.1-0.2	7.56
FSC-01	2016	UN	15.69	130	2	0	2	5.08	7.76	8.18	0.28	0.1-0.2	18.34
FSC-02	2016	UN	-	-	-	-	-	-	-	-	-	-	-
KBU-02	2016	BU	15.21	216	1	1	5	21.00	2.35	3.40	0.30	N/A	7.19
KBU-03	2016	BU	13.46	216	1	0	4	12.00	2.90	3.40	0.63	0.2-1.6	9.51
KC-01	2016	UN	16.20	216	1	0	2	17.00	4.70	6.20	0.64	<0.1	0.05
KC-02	2016	UN	13.03	240	3	1	2	14.10	2.90	3.50	0.58	<0.1	12.46
KC-03	2016	UN	14.04	246	1	1	4	17.20	6.63	8.65	0.35	0.2-1.6	8.05
KVB-01	2016	BU	-	-	-	-	-	-	-	-	-	-	-
KVB-02	2016	BU	14.92	228	1	0	2	7.00	1.46	3.53	0.20	0.1-0.2	6.66
KVB-03	2016	BU	-	-	-	-	-	-	-	-	-	-	-
BF-01	2015	UN	12.80	162	0	2	1	31.83	8.90	11.30	0.00	<0.1	5.13

Appendix XV

Raw water quality observations for streams located in the Dehcho and South Slave regions.

	NH3	TOC	Alkalinity	Conductivity	TSS	Turbidity	pH	Cl-	NO3 as N	K+	Diss. Al	Diss. Ar	Diss. Ba	Diss. Cu	Diss. Fe	Diss. Ni	Diss. Rb	Total Zn	Total Cu
UOM	mg/L	mg/L	mg/L	mS/cm	mg/L	NTU	pH unit	mg/L	mg/L	mg/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
CH-01	0.008	26.7	209	432	3	0.25	7.95	6.4	0.07	0.8	1.5	0.3	40.7	0.2	33	0.4	1.2	1	1.1
KU-01	0.006	25.5	219	421	6	1.41	7.93	1.6	0.08	0.4	2	0.2	46.3	0.2	50	0.4	1	1	2.3
WS-01	0.01	28	208	429	3	0.42	7.88	6.6	0.08	0.8	1.2	0.3	38.9	0.2	42	0.4	1.3	0.8	1.2
KC-01	0.006	19.2	247	441	3	0.27	8.05	1	0.18	0.4	2.6	0.3	53.9	0.2	67	0.4	0.6	0.4	0.2
KC-02	0.01	17.9	263	585	3	2.14	7.92	30	0.28	1.3	9.2	0.3	63.9	0.6	173	1	0.9	2.8	0.2
KC-03	0.018	33.4	225	411	3	0.79	8.17	0.7	0.04	0.2	4.4	0.4	33.8	0.2	115	0.3	0.5	0.4	0.4
FSC01	0.009	20.2	211	396	3	0.74	7.54	3.4	0.17	0.6	3.4	0.2	87.5	0.2	118	0.2	1.5	0.4	0.2

FSC02	0.013	14.5	266	476	3	0.54	8.34	1.6	0.05	0.7	4.2	0.2	167	0.2	68	0.3	1	0.4	0.2
FSB01	0.009	21.6	189	362	3	0.34	7.88	2.9	0.05	0.3	17	0.2	216	0.9	158	0.7	0.5	4.5	0.2
FSB03	0.018	13.1	90.7	187	3	0.89	8.19	0.7	0.13	0.4	7.8	0.3	37.3	0.6	74	0.6	0.5	0.5	0.4
KVB-01	0.018	38.3	149	639	17	4.73	7.83	2.8	0.21	1.3	8.1	0.6	29	0.3	135	0.9	0.8	0.8	1.5
KVB-02	0.018	34.3	135	511	6	2.1	7.64	15.8	0.18	0.8	10.4	0.6	26.9	0.2	134	0.8	0.6	0.9	0.4
KVB-03	0.011	40.6	126	537	53	0.75	7.44	1.8	0.19	1.4	8.7	0.5	24.9	0.2	72	0.5	1.2	1.3	0.2
KBU-02	0.01	20.4	190	379	21	6.81	7.85	3.8	0.18	0.8	4	0.3	38.4	0.2	49	0.2	1.3	0.4	0.2
KBU-03	0.012	19.7	263	496	11	1.55	7.96	5	0.19	0.6	6	0.3	57.2	0.2	103	0.5	0.8	0.4	0.2

Appendix XVI

Raw benthic invertebrate counts for streams located in the Dehcho and South Slave regions.

	CH-01	KU-01	WS-01	FSC-01	KC-01	KC-02	KC03	FSB-01	FSB-03	KVB-02	KBU-02	KBU-03
Ameletus	6	0	0	0	0	0	0	0	37	0	0	0
Baetidae	0	0	0	1	0	0	56	2	45	8	1	7
Acentrella	0	0	0	0	1	0	0	0	0	0	0	0
Acerpenna sp.	0	0	0	1	0	2	0	0	3	0	0	0
Acerpenna pygmaea	0	0	0	0	0	0	0	0	1	0	0	0
Baetis	0	4	0	13	75	33	80	7	15	1	7	4
Baetis tricaudatus group	0	0	0	5	23	1	3	0	1	3	0	0
Callibaetis sp.	0	29	0	0	0	0	0	0	0	1	0	0
Ephemera	0	0	0	0	0	0	0	0	1	0	0	0
Ephemera simulans	0	0	0	0	0	0	0	0	1	0	0	0
Heptageniidae	0	8	0	2	6	0	11	2	26	0	0	0
Maccaffertium	0	0	0	0	0	0	0	0	1	0	0	0
Leptophlebiidae	0	4	0	1	1	3	1	42	7	13	2	3
Paraleptophlebia	0	0	0	0	0	0	0	1	0	0	0	0
Capniidae	0	0	0	0	0	3	0	1	0	0	0	
Chloroperlidae	0	0	0	1	1	29	2	7	0	0	0	8
Nemouridae	0	0	0	0	68	57	4	54	0	7	0	47
Amphinemura	0	0	0	0	1	0	0	3	0	0	0	0
Nemoura	0	0	0	0	0	0	7	7	0	7	0	4
Zapada	0	0	0	1	0	0	0	0	0	0	0	0
Zapada cinctipes	0	0	0	0	0	2	0	0	0	0	0	0
Perlodidae	0	0	0	0	0	0	0	0	1	0	0	0
Pteronarcys	0	0	0	0	0	0	0	0	1	0	0	0
Trichoptera	0	0	0	0	0	0	0	1	0	0	0	0
Apatania	0	0	0	0	0	0	0	0	2	0	0	0
Brachycentridae	0	0	0	0	0	0	1	3	0	0	0	0
Brachycentrus	0	0	0	0	0	0	2	0	0	0	0	0
Brachycentrus americanus	0	0	0	0	0	0	0	1	0	0	0	0
Micrasema	0	0	0	0	0	0	0	1	0	0	0	23
Glossosomatidae	0	0	0	0	0	0	0	2	0	0	0	0
Glossosoma	0	4	0	0	2	2	4	2	0	0	0	4
Hydropsychidae	0	4	0	2	12	0	3	0	0	0	0	1
Cheumatopsyche	0	0	0	0	0	0	0	0	0	0	0	0
Hydropsyche	0	4	0	0	2	0	10	0	1	0	0	0

Hydroptilidae	0	0	0	3	0	0	0	4	0	0	4	6
Hydroptila	0	0	0	0	0	0	0	0	0	0	3	1
Oxyethira sp.	0	0	0	0	1	7	0	5	0	0	20	7
Lepidostomatidae	0	0	0	0	0	0	0	0	0	0	0	0
Lepidostoma	0	4	0	0	3	3	3	0	5	0	1	0
Leptoceridae	0	0	0	0	0	1	0	0	0	0	1	0
Ceraclea	0	0	0	0	0	0	0	0	2	0	0	0
Limnephilidae	0	0	0	0	0	0	0	0	0	5	0	1
Chimarra sp.	0	0	0	0	0	0	0	0	4	0	0	0
Ptilostomis	0	0	0	0	0	0	0	0	0	1	0	0
Rhyacophila	0	0	0	1	0	0	0	0	0	0	0	0
Oligophlebodes	0	8	0	0	0	0	0	0	0	0	0	0
Elmidae	0	4	0	0	0	0	5	0	0	0	4	6
Optioservus sp.	0	8	0	0	0	4	8	0	2	0	0	0
Haliplus sp.	0	4	0	0	0	0	0	0	0	0	0	0
Berosus	0	0	0	0	0	0	0	0	0	1	0	0
Staphylinidae	0	0	0	0	0	0	0	0	0	1	0	0
Diptera	0	0	0	0	0	0	0	1	0	0	0	2
Ceratopogonidae	0	0	7	0	0	0	0	0	0	20	6	0
Alluaudomyia sp.	22	4	0	0	0	0	0	0	0	0	0	0
Atrichopogon	0	4	0	0	0	0	0	0	0	4	0	0
Bezzia/ Palpomyia	6	63	0	0	0	0	0	0	0	14	2	2
Culicoides	0	0	0	0	0	0	0	0	0	7	0	0
Dasyhelea sp.	0	29	0	0	0	0	0	0	0	0	0	0
Probezzia	0	0	0	0	0	0	0	1	0	1	2	1
Chironomidae	33	29	10	12	15	1	18	4	1	10	3	2
Chironomus	0	92	0	0	0	0	0	0	0	0	0	0
Dicrotendipes	0	0	0	0	0	6	0	0	0	0	21	0
Endochironomus	0	0	0	0	0	0	0	1	0	0	0	0
Paralauterborniella nigrohalterale	0	0	0	0	0	0	0	0	0	3	0	0
Paratendipes	0	0	0	0	0	0	0	0	0	6	0	2
Phaenopsectra	0	0	0	0	0	0	0	1	0	0	0	0
Polypedilum sp.	22	21	0	1	0	0	0	0	2	42	4	2
Stictochironomus	0	0	0	0	0	0	0	0	0	0	12	0
Tanytarsini	0	0	6	13	0	4	0	8	0	0	31	0
Cladotanytarsus	0	4	0	0	0	0	0	0	0	0	0	0
Constempellina sp. C	0	0	0	2	0	3	0	28	0	3	0	0
Micropsectra	0	4	18	57	9	19	5	1	0	6	10	9
Neostempellina reissi	0	0	0	1	0	0	0	0	88	0	0	0
Paratanytarsus	0	17	0	0	0	0	0	0	0	1	0	1
Rheotanytarsus	0	0	0	40	0	7	2	9	0	0	35	0
Stempellinella	0	92	0	5	0	0	1	0	0	0	22	1
Tanytarsus	0	4	3	10	1	5	0	0	0	4	5	4

Lappodiamesa	0	0	10	0	0	0	0	0	0	0	0	0
Arctodiamesa	39	0	0	0	0	0	0	0	0	0	0	0
Pseudokiefferiella	39	0	0	0	0	0	0	0	0	0	0	0
Potthastia longimana group	0	0	0	0	0	0	0	2	0	0	1	0
Brillia	0	0	0	1	0	0	0	1	0	0	0	0
Cardiocladius	0	0	0	0	0	0	1	0	0	0	0	0
Corynoneura	0	0	3	1	0	1	0	1	0	13	0	1
Diplocladius cultriger	0	0	0	0	0	0	0	0	0	2	0	0
Eukiefferiella	0	0	0	8	3	0	14	9	0	0	0	3
Euryhapsis	0	0	0	0	0	0	1	0	0	0	0	0
Heterotanytarsus	0	0	0	0	0	0	0	1	0	0	0	0
Heterotrissocladius	0	0	0	1	0	6	0	1	0	9	0	1
Hydrosmittia	0	0	3	0	0	0	0	0	0	0	0	0
Limnophyes	0	0	0	0	0	0	0	0	0	5	0	0
Metriocnemus	0	0	0	0	0	0	0	0	0	1	0	0
Nanocladius	0	0	0	0	0	0	1	0	0	0	0	0
Orthocladius complex	389	4	91	50	0	23	10	2	0	6	38	83
Parakiefferiella	0	8	0	1	0	0	0	0	0	2	0	0
Parametriocnemus	0	0	1	0	1	0	0	0	2	0	0	0
Paraphaenocladius	0	0	0	3	4	0	0	0	0	2	0	0
Psectrocladius	0	8	1	0	0	3	0	0	1	0	5	0
Rheocricotopus	0	0	35	0	0	0	0	0	0	0	0	0
Rheosmittia	0	0	0	0	0	0	0	0	2	0	0	0
Synorthocladius	0	0	0	1	0	2	0	0	0	0	0	0
Thienemanniella	0	0	2	0	0	0	0	4	0	1	0	5
Tvetenia	0	4	3	7	55	20	42	1	1	1	1	14
Zalutschia	372	13	0	1	1	0	0	0	0	0	0	3
Ablabesmyia	0	4	0	0	1	4	0	1	0	4	11	1
Nilotanypus	0	0	0	0	1	0	0	0	0	0	0	0
Paramerina sp.	0	0	0	0	0	0	1	5	0	0	0	0
Zavrelimyia	0	0	0	0	0	0	0	2	0	0	0	0
Pentaneurini	0	0	0	0	0	0	0	0	0	0	0	0
Thienemannimyia group	22	4	0	28	5	22	0	29	4	0	44	29
Procladius	0	67	0	0	0	0	0	0	2	5	8	0
Culicidae	0	4	0	0	0	0	0	0	0	0	0	0
Dixidae	6	0	0	0	0	0	0	0	0	0	0	0
Empididae	0	0	0	0	2	0	1	0	16	0	0	5
Chelifera/ Metachela	0	0	0	0	0	1	0	0	0	0	0	2
Hemerodromia sp.	6	4	0	1	0	0	0	0	0	0	0	0
Simuliidae	6	0	2	0	16	5	6	5	0	0	0	6
Helodon sp.	11	0	0	0	0	0	0	0	0	0	0	0

Simulium	467	0	45	6	25	2	3	5	1	0	2	10
Chrysops	0	0	0	0	0	0	0	0	0	2	0	0
Tipulidae	0	0	0	0	0	0	0	0	1	0	0	0
Dicranota	0	0	0	0	0	0	0	0	0	1	0	0
Limnophila	0	0	0	0	0	0	0	0	1	0	0	0
Corixidae	0	4	0	0	0	0	0	0	0	0	0	0
Callicorixa	0	0	0	0	0	0	0	0	0	1	0	0
Sigara	0	4	0	0	0	0	0	1	0	1	0	0
Aeshnidae	0	0	0	0	0	2	0	0	0	0	0	0
Aeshna sp.	0	0	0	0	0	0	0	0	0	0	1	0
Corduliidae	0	0	0	0	0	1	0	2	0	0	0	0
Coenagrionidae	0	8	0	0	0	0	0	0	0	0	0	0
Gomphidae	0	0	0	0	0	0	1	0	8	0	0	0
Ophiogomphus	0	0	0	0	0	0	0	0	5	0	0	0
Malacostraca	0	0	0	0	0	0	0	0	0	0	0	0
Amphipoda	0	0	0	0	0	0	0	0	0	0	0	0
Gammaridae	0	0	0	0	0	0	0	0	0	0	0	0
Gammarus	0	0	0	0	0	0	0	0	0	0	1	0
Hyalellidae	0	0	0	0	0	0	0	0	0	0	0	0
Hyalella	0	0	0	1	4	0	1	0	0	0	1	0
Pisidiidae	0	83	0	0	0	0	0	0	30	0	27	0
Pisidium	0	121	0	0	0	0	0	0	0	0	0	0
Sphaerium sp.	0	0	0	0	0	0	0	0	1	0	0	0
Gastropoda	61	121	10	3	1	14	2	3	0	7	30	2
Fossaria	0	0	0	0	0	0	0	0	0	12	0	0
Lymnaea	0	63	0	0	0	0	0	0	0	0	0	0
Stagnicola	0	0	3	0	0	8	0	0	0	0	0	0
Physa	0	0	0	0	0	0	0	0	0	0	0	1
Physella	0	0	0	0	0	0	0	1	0	0	0	0
Planorbidae	11	0	0	0	0	0	0	0	0	0	0	0
Gyraulus	28	129	17	5	0	5	0	5	1	0	6	23
Planorbella	0	0	0	0	0	0	1	0	0	0	0	0
Valvata	0	142	0	2	0	4	0	13	2	0	0	0
Pomatiopsis sp.	6	0	0	0	0	0	0	0	0	0	0	0
Erpobdellidae	0	0	0	0	0	0	1	0	0	0	0	0
Erpobdella	0	0	0	0	0	0	1	0	0	0	0	0
Helobdella stagnalis	0	0	0	0	0	0	0	1	0	0	0	0
Lumbriculus	0	0	0	0	0	0	0	1	0	26	0	0
Enchytraeus	156	54	36	0	1	1	0	1	0	0	0	1
Naididae	28	46	0	0	0	0	0	0	0	0	1	0
Nais	0	0	0	0	0	0	0	0	0	30	0	0
Stylaria lacustris	0	21	0	0	0	0	0	0	0	0	0	0
Tubifex	0	0	0	0	0	0	0	0	0	4	0	0