

PERMAFROST AND WILDFIRE INFLUENCES ON STREAM NUTRIENT
DYNAMICS AND METABOLISM IN BOREAL FOREST WATERSHEDS OF
INTERIOR ALASKA

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Abstract

In the boreal forest of interior Alaska, both permafrost and wildfire impact stream solute concentrations, but their effect on stream function is unknown. This research focused on the effects of wildfire and permafrost on stream nutrient dynamics and metabolism in the Caribou Poker Creeks Research Watershed (CPCRW) in interior Alaska. Wildfire impact on chemistry was determined through comparison of pre- and post-fire stream chemistry in a control and a burned watershed. We predicted that increased nutrient and decreased dissolved organic carbon (DOC) delivery to streams after fire would stimulate gross primary productivity (GPP) and reduce ecosystem respiration in burned watersheds. Fire resulted in higher stream nitrate, sulfate, and cation concentrations, and lower DOC concentration, at the burned site than at the control site. Streams draining burned watersheds had higher summer GPP ($2.4 \text{ gO}_2 \text{ m}^{-2} \text{ day}^{-1}$) than the unburned sites ($1.2 \text{ gO}_2 \text{ m}^{-2} \text{ day}^{-1}$). Respiration was also higher in burned than unburned watersheds (3.9 and $3.0 \text{ gO}_2 \text{ m}^{-2} \text{ day}^{-1}$, respectively). Metabolism was not correlated with the increased nutrient concentrations observed after fire. Instead, we suggest GPP was stimulated through increased soluble reactive phosphorus availability after fire, whereas respiration was likely controlled by pre-existing differences in stream physical or chemical characteristics.

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Chapter 1: Background

Permafrost and hydrology

Permafrost, defined as ground that has remained below 0°C for two or more years, plays a major role in shaping ecosystem structure and function in the boreal forest of interior Alaska (Davis 2001). Whereas continuous permafrost is present over much of northern Alaska, interior Alaska is underlain by discontinuous permafrost, with the distribution largely determined by aspect, vegetation and winter soil temperature (Davis 2001, Hinzman et al. 2006; Fig. 1). The presence and extent of permafrost in catchments is a defining condition for boreal forest streams, influencing hydrological flowpaths and, hence, nutrient inputs to streams.

Permafrost inhibits infiltration of precipitation to groundwater and results in restriction of flow to soil organic horizons (Kane et al. 1992, MacLean et al. 2002). Streams draining catchments underlain by a large extent of permafrost are characterized by high dissolved organic matter (DOM) and low nutrient concentrations. In contrast, streams in catchments with low extents of permafrost typically have low DOM and high nutrient concentrations (MacLean et al. 1999, Petrone et al. 2006; Fig. 2). DOM and nutrient concentrations appear to be closely linked to depth of the active layer. The active layer is the upper region of the soil profile that freezes and thaws annually and in the region of discontinuous permafrost is generally 0.5 to 3 m deep (Davis 2001). During early summer, the depth of thaw is generally shallow but progressively increases as air temperature rises and heat is conducted from the soil organic layer. As the depth of thaw

increases, hydrologic flowpaths transition from the soil organic horizon to increased infiltration through the mineral soil horizon (Kane et al. 1992). As flow percolates through to mineral soil, DOM fluxes to streams decline, resulting in a progressive decline in DOM concentration during summer in high permafrost watersheds (Petrone et al. 2006).

Boreal forest and fire

Boreal forest stand type and permafrost extent are intimately linked and generally maintained through negative feedbacks. Soil underlying black spruce stands (*Picea mariana*) tends to be wetter and more acidic than soil underlying deciduous stands (Baldocchi et al. 2000, Ping et al. 2005), which keeps organic matter mineralization low and promotes maintenance of a thick organic layer (Ping et al. 2006). The thick organic layer acts as an insulator keeping soil temperature low, maintaining underlying permafrost and reinforcing environmental conditions suitable for black spruce growth. Black spruce stands are more flammable than deciduous stands and cause increased occurrence, size and severity of wildfires in the boreal forest (Kasischke et al. 2006, Rupp et al. 2002).

Fire severity is a key factor that will determine the extent of change in boreal forest ecosystems after wildfire (Hinzman et al. 2006). Permafrost degradation after ground fires results primarily from an altered soil thermal regime following fire (Burn 1998). Combustion of the insulating forest organic layer results in greater heat conductance to subsurface soils and continued permafrost thaw (Yoshikawa et al. 2003),

charcoal formation, and reduced albedo (Certini 2005), as well as increased incident radiation via reduced canopy shading (DeBano et al. 1998). In sites with underlying permafrost, post-fire re-accumulation of surface organic matter stabilizes permafrost and potentially permits re-establishment of black spruce stands. Alternatively, in sites with little or unstable permafrost, complete removal of the organic layer and associated change in soil thermal regime may initiate permafrost degradation and facilitate the establishment of deciduous tree stands via successional processes. This latter scenario is most likely to occur where permafrost is unstable (near the point of thawing) and can result in a change in vegetation trajectory (Chapin et al. 2004). In a study of black spruce distribution after wildfire in interior Alaska, Johnstone and Kasischke (2005) found a negative relationship between fire severity and subsequent recruitment of black spruce. After the most severe fires, the authors reported replacement of pre-fire black spruce stands by mixed conifer and deciduous vegetation.

Wildfire has substantially changed solute chemistry in boreal forest soils. Analysis of soil solute concentrations in interior Alaska indicated that fire led to reduced organic carbon and nitrogen and increased calcium concentrations, with observed changes limited to the upper soil horizon (Neff et al. 2005). Similarly, Shibata et al. (2005) found post-fire reductions in surface organic soil DOC and water extractable organic carbon concentrations as well as increases in organic soil sodium and potassium concentrations in the soil organic layer. Forest stand type can also influence post-fire soil nutrient dynamics. In a study of fire effects in four forests types, soil solution nitrogen and phosphorus concentrations after fire were significantly higher in black spruce stands,

whereas soil nitrogen decreased and phosphorus concentration increased only marginally in white spruce and aspen stands (Dyrness et al. 1989).

In the boreal forest, fire impacts permafrost extent, stand vegetation dynamics and soil chemistry and, therefore, may alter stream nutrient concentrations. In an earlier study of the effects of prescribed fire on stream chemistry at the Caribou Poker Creeks Research Watershed (CPCRW; Pertone et al. In Press) the only detectable change was an increase in nitrate concentration during a post-fire rain storm. However, the prescribed fire was small (28% of watershed area) and restricted primarily to uplands. The effect of large-scale wildfires on stream chemistry and stream function in the zone of discontinuous permafrost remains unknown.

Whole-stream metabolism

Whole-stream metabolism is a measure of in-stream primary production and ecosystem respiration. Primary production or fixation of carbon dioxide to reduced organic compounds is achieved via the photosynthetic pathways of macrophytes, bryophytes, algae and cyanobacteria and occurs predominantly at the stream-sediment boundary (Bott 1996). Ecosystem respiration is a measure of energy consumption by both autotrophic and heterotrophic organisms (e.g., bacteria and fungi). Respiration occurs to a small extent in the water column, but is predominately a result of microbial activity at the stream-sediment boundary and in the soils of the hyporheic zone (Allan 1995). Both production and consumption alter dissolved oxygen concentration in streams, with primary production producing dissolved oxygen and respiration consuming

dissolved oxygen. Metabolism can therefore be quantified by monitoring change in stream dissolved oxygen concentration over time (Bott 1996, Odum 1956).

At any point in time, dissolved oxygen concentration in streams (C , gO_2/L) can be quantified using the equation:

$$C = P - R \pm E \quad (1)$$

where P is dissolved oxygen production via photosynthesis, R is oxygen consumption through respiration and E is an evasion term quantifying oxygen exchange with the atmosphere (Bott 1996, Odum 1956). Essential in the calculation of stream metabolism is an estimate of the oxygen evasion term. Oxygen evasion has been estimated from either channel physical parameters such as stream discharge, velocity, depth, and turbidity, or by direct measurement using conservative and volatile tracers (Hibbs et al. 1998, Stream Solute Workshop 1990, Tsivoglou and Neal 1976).

Metabolism is calculated from an evasion-corrected curve of diel oxygen concentration. Periodic measurements of dissolved oxygen concentration are used to generate a rate of change curve, which indicates the change in stream oxygen concentration between two points in time (Fig. 3). By applying the relationship in equation 1 to a rate of change curve, stream metabolism can be expressed as:

$$GPP = NEP - R \quad (2)$$

where GPP , NEP and R ($\text{gO}_2 \text{ m}^{-2} \text{ day}^{-1}$) are gross primary production, net ecosystem production and respiration, respectively (Bott 1996, Odum 1956). The nighttime change in dissolved oxygen provides a measure of heterotrophic respiration (Fig. 3). With the assumption that daytime respiration is equal to nighttime respiration

we can calculate daily respiration rate by multiplying the average hourly rate of nighttime heterotrophic respiration by 24:

$$R = \Delta O_{2(night)} * 24 \quad (3)$$

where $\Delta O_{2(night)}$ is the average hourly nighttime change in dissolved oxygen concentration (Fig. 3). Net ecosystem productivity is calculated as:

$$NEP = \sum \Delta O_2 \quad (4)$$

where $\sum \Delta O_2$ is the sum of total change in dissolved oxygen concentration throughout the 24 hour study period (Fig. 3). From equation 2, GPP is then calculated as net ecosystem production (NEP) minus ecosystem respiration (R).

Factors controlling stream metabolism

Stream ecosystem metabolism is determined by the physical and chemical properties of the stream and its surrounding catchment. Whole stream metabolism reflects the productivity of biota in all stream compartments (pelagic, benthic and hyporheic) under their particular nutritional and environmental constraints. Gross primary production is carried out via photosynthetic pathways and, therefore, requires sufficient light, dissolved carbon dioxide, and inorganic nutrients, such as nitrogen and phosphorus. Due to these basic physiological requirements, observed rates of GPP have been found to be limited by light (Mulholland et al. 2001), nutrients (Elser et al. 1990, Gausch et al 1995), or temperature (Webster and Meyer 1997).

Stream ecosystem respiration encompasses consumption of reduced organic carbon stocks by both autotrophic and heterotrophic organisms. Moreover, all organisms have specific carbon to nutrient requirements, and therefore need to be able to assimilate sufficient nutrients from their environment to balance carbon use. Because of the physiological requirements of respiration, rates are commonly limited by temperature as well as availability of carbon, macronutrients, and oxygen. Respiration has been linked to dissolved organic matter concentration in streams (Kreutzweiser and Capell 2003, Young and Huryn 1999), benthic organic matter (Acuña et al. 2004, Hedin 1990) and presence of coarse woody debris (Houser et al. 2005). In a meta-analysis of benthic respiration across 22 study streams in North America temperature was found to significantly increase respiration rate, with temperature alone explaining 38% of the variability in respiration across sites (Sinsabaugh 1997).

Physical processes that alter primary limiting factors in the stream environment also control metabolism. Environmental factors such as stream discharge, catchment hydrologic flowpath, riparian canopy cover, and season will alter stream solute concentrations, light levels, temperature, and oxygen availability and, subsequently, rates of metabolism. Similarly, alteration of the surrounding catchment can impact important stream physicochemical parameters. For example, forest stand removal can increase stream discharge and light availability while changing the quantity or quality of allochthonous carbon inputs. Several studies have reported that streams in catchments with intact, undisturbed native vegetation received greater organic matter inputs and exhibited significantly higher rates of respiration than streams bounded by agricultural

pastureland (Houser et al. 2005, Young and Huryn 1999). Houser et al. (2005) attributed these lower respiration rates in impacted watersheds to declines in benthic organic matter and standing stocks of coarse woody debris. In a study of land-use impacts on stream health, reduction of stream riparian vegetation led to higher rates of gross primary production and lower respiration rates than in streams draining undisturbed catchments (Bunn et al. 1999).

The effects of wildfire on stream metabolism are largely unknown, but like other types of catchment disturbance, fire has potential to substantially alter stream solute concentrations as well as temperature, discharge, and ambient light conditions. The magnitude of changes to stream physical and chemical properties after fire is directly related to burn severity and to the proportion and location of the affected catchment (Gresswell 1999). Stream discharge can increase dramatically after fire, largely due to reduced interception by vegetation and reduced evapotranspiration (Bayley and Schindler 1991, Prepas et al. 2003). Increased stream flow has potential to reduce water clarity and increase sediment transport, bank erosion, and bed scouring (Minshall et al. 2001). Fire also impacts stream solute concentrations, most commonly leading to increases in stream nitrogen, phosphorus, sulfate and cation concentrations (Bayley and Schindler 1991, Hauer and Spencer 1998, Spencer et al. 2003, Williams and Melack 1997).

Permafrost and wildfire influences on stream function in the boreal forest

Permafrost prevalence in the northern boreal forest is closely tied to soil temperature. As summer soil temperature approaches 0°C, permafrost becomes

increasingly unstable and likely to degrade (Davis 2001). Analysis of permafrost dynamics across Alaska has indicated that subsurface soil temperature has increased over the last few decades, with the permafrost in this region becoming highly susceptible to thawing if air temperature continues to rise (Osterkamp and Romanovsky 1999). Specifically, permafrost extent at the Caribou Poker Creeks Research Watershed (CPCRW), in interior Alaska, has declined 2.1%, and one-third of remaining permafrost has become unstable, i.e. close to point of thawing, over the last century (Hinzman et al. 2006).

Combined with rising air temperatures, climatic warming is predicted to cause an increase in the size and number of wildfires in Alaska's boreal forest (Kasischke et al. 1995, Stocks et al. 1998). The interaction of wildfire and climatic warming will likely have a profound effect on vegetation, permafrost extent, and catchment hydrology in the region of discontinuous permafrost. However, how these environmental changes will affect stream chemistry and function in the boreal forest is unknown, though it is likely that changes in catchment soil solution chemistry after fire will result in immediate changes to stream nutrient concentrations. Additionally, permafrost extent may be reduced after wildfire either immediately (via latent heat transfer) or over successional timescales (via the loss of permafrost and replacement of black spruce stands with deciduous species). Loss of underlying permafrost will alter catchment hydrology, most likely reducing the importance of flows from the soil organic horizon, and further affect stream solute chemistry.

The role of permafrost extent and differing hydrologic flowpaths on whole stream function has not been established. Given the substantial differences in stream chemical properties in watersheds with differing extents of permafrost, functional differences are likely. Furthermore, the relative influence of permafrost versus wildfire effects on stream structure and function has not been determined. This question is becoming increasingly important as boreal forest watersheds are currently in a period of rapid change.

This research addresses the question of how permafrost and wildfire drive differences in stream chemistry among watersheds in the region of discontinuous permafrost, and in turn, how those differences affect rates of in-stream metabolism. The research was conducted in the CPRW. Sub-catchments of CPRW are underlain with varying extents of permafrost, for which stream chemistry data have been collected from 2002 – 2005. Furthermore, one sub-catchment of CPRW was extensively burned during the summer of 2004.

Permafrost effects on stream chemistry were addressed through comparison of patterns in stream chemistry among watersheds. Fire effects on solute concentrations were determined by comparing pre- and post-fire stream chemistry in the burned and a control watershed. The relative effects of fire and permafrost on stream function in the region of discontinuous permafrost were established by measuring rates and determining the drivers of metabolism in CPRW sub-catchments.

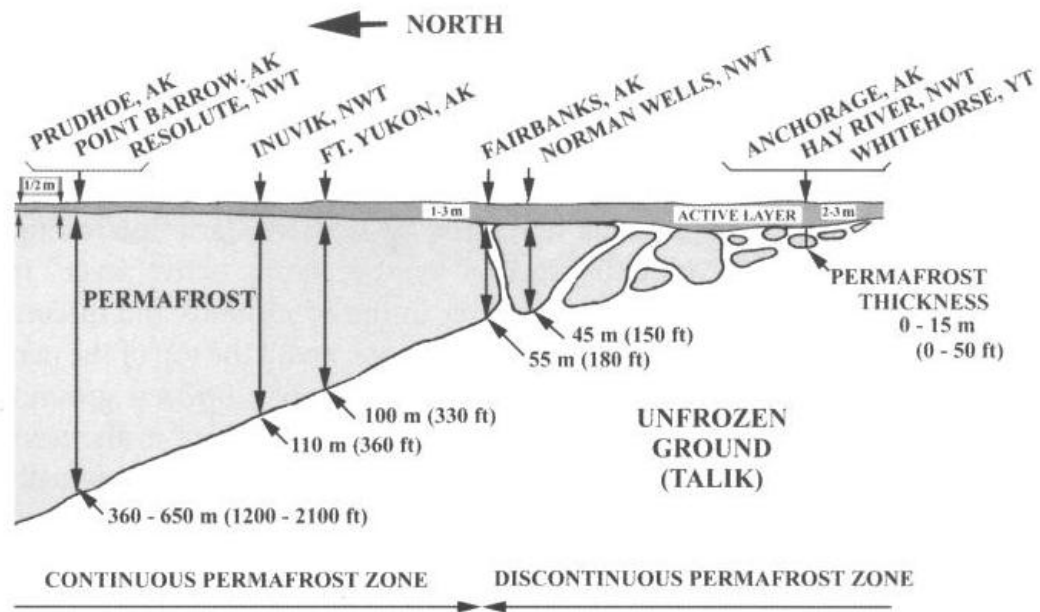


Figure 1. Permafrost thickness and longitudinal distribution in Alaska and Northern Canada. Also illustrated is the transition from continuous to discontinuous permafrost (Davis 2001).

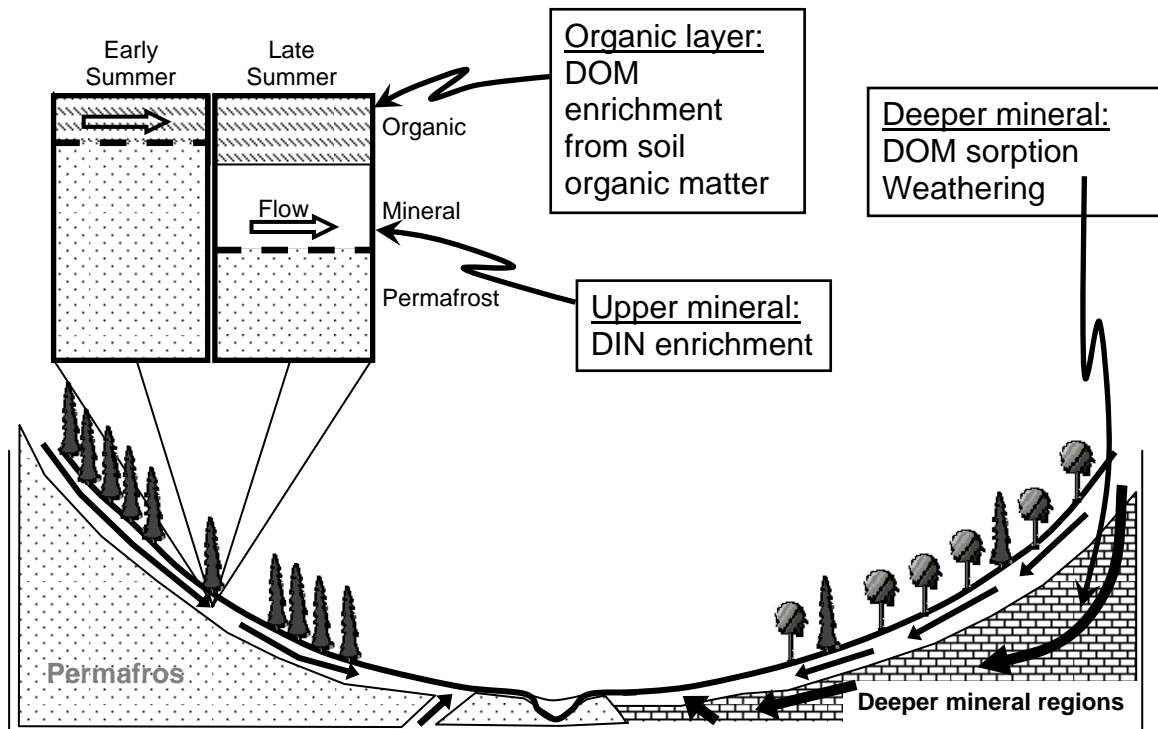


Figure 2. Conceptual model of permafrost control over hydrology and stream chemistry in the region of discontinuous permafrost (modified from MacLean et al. 1999). Streams draining catchments underlain by permafrost are characterized by high dissolved organic matter (DOM) and low nutrient concentrations. In catchments with low extents of permafrost, streams typically have low DOM and high nutrient concentrations.

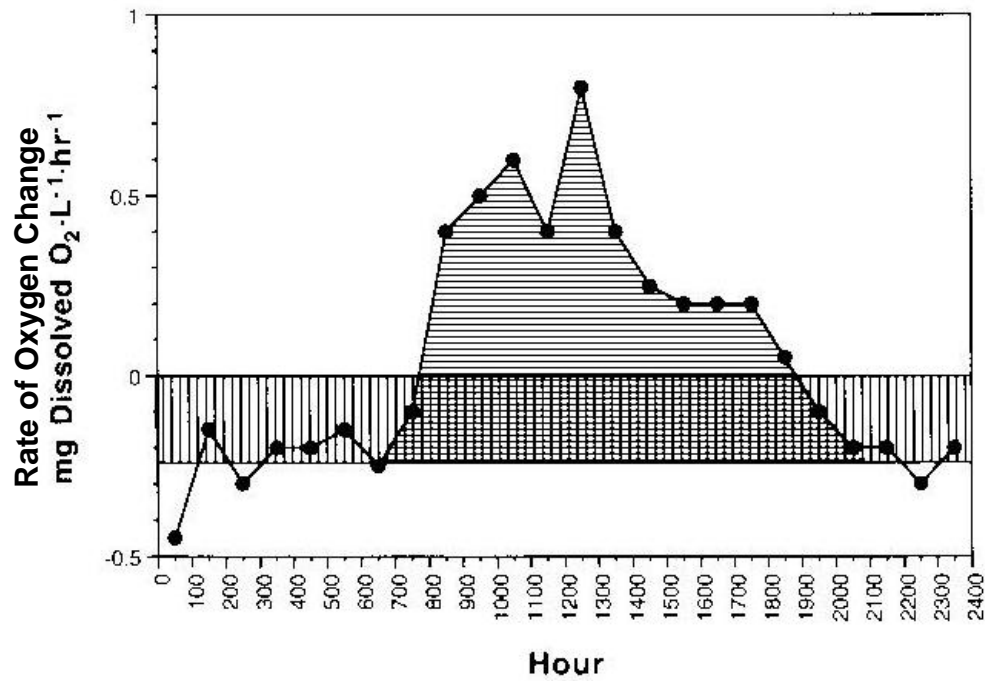


Figure 3. Rate of change curve derived from changes in dissolved oxygen concentration over 24 hours (figure from Bott 1996). Vertical lines = minimum hourly change in dissolved oxygen multiplied by 24 (Ecosystem respiration; R). Horizontal lines = Gross primary production.

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Chapter 2: Permafrost and Wildfire Influences on Stream Nutrient Dynamics and Metabolism in Boreal Forest Watersheds of Interior Alaska¹

Abstract

In the boreal forest of interior Alaska, both permafrost and wildfire impact stream solute concentrations, but their effect on stream function is unknown. We² studied the effects of wildfire and permafrost on stream nutrient dynamics and metabolism in the Caribou Poker Creeks Research Watershed (CPCRW) in interior Alaska. Wildfire impact on chemistry was determined through comparison of pre- and post-fire stream chemistry in a control and a burned watershed. We predicted that increased nutrient and decreased dissolved organic carbon (DOC) delivery to streams after fire would stimulate gross primary productivity (GPP) and reduce ecosystem respiration in burned watersheds. Fire resulted in higher stream nitrate, sulfate, and cation concentrations, and lower DOC concentration, at the burned site than at the control site. Streams draining burned watersheds had higher summer GPP ($2.4 \text{ gO}_2 \text{ m}^{-2} \text{ day}^{-1}$) than the unburned sites ($1.2 \text{ gO}_2 \text{ m}^{-2} \text{ day}^{-1}$). Respiration was also higher in burned than unburned watersheds (3.9 and $3.0 \text{ gO}_2 \text{ m}^{-2} \text{ day}^{-1}$, respectively). Metabolism was not correlated with the increased nutrient concentrations observed after fire. Instead, we suggest GPP was stimulated through increased soluble reactive phosphorus availability after fire, whereas respiration was

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likely controlled by pre-existing differences in stream physical or chemical characteristics.

Introduction

Climate change will have multi-fold impacts on streams in the boreal forest through changes in temperature, permafrost distribution, fire regimes and impacts on stream flows. In the region of discontinuous permafrost, soil temperature is near the point of thawing, and permafrost is susceptible to rapid degradation (Osterkamp and Romanovsky 1999). Rising air temperature over the past century has already caused a reduction in the extent of permafrost (Hinzman et al. 2006, Jorgensen et al. 2001).

Wildfire is a widespread disturbance in the boreal forest, and is fundamental in shaping forest structure, function and successional processes (Chapin et al. 2006). Moreover, with climatic warming, the northern boreal forest is predicted to experience earlier onset of the wildfire season combined with a potential increase in the number and size of fires (Kasischke et al. 1995, Stocks et al. 1998). In regions with permafrost, fire impacts on surface organic layer depth, albedo, and vegetation are likely initiate positive feedbacks on soil temperature, contributing to decreased permafrost extent and increased depth of seasonal thaw (Burn 1998, Yoshikawa et al. 2003).

Catchment hydrology and stream chemistry are regulated by both permafrost extent and wildfire. In the boreal forest of interior Alaska, watersheds are characterized by discontinuous permafrost. Where present, permafrost is an effective barrier to groundwater infiltration and thus restricts watershed flowpaths to organic soil horizons.

Streams draining catchments with extensive permafrost have higher dissolved organic matter (DOM) and lower nutrient concentrations than streams draining catchments with low extents of permafrost (MacLean et al. 1999, Petrone et al. 2006). Wildfire is a ubiquitous disturbance in terrestrial ecosystems and impacts streams through alteration of the physical environment, including temperature and light regimes (Gresswell 1999, Hitt 2003). Streams are closely coupled to their catchments with allochthonous inputs an important source of dissolved organic matter and nutrients. Combustion of terrestrial organic matter has the potential to significantly alter stream solutes, in particular, carbon, nitrogen, phosphorus and cation concentrations (Bisson et al. 2003, Gresswell 1999). Reduced evapotranspiration following wildfire may also result in changes to stream discharge and hydrologic flowpaths through watersheds, thereby altering both the magnitude and timing of flows and solute delivery to streams (Earl and Blinn 2003, Petrone et al. In press, Williams and Melack 1997).

Whole stream metabolism is a product of instream primary production and ecosystem respiration and, as such, is an integrative measure of stream function. Rates of metabolism can have significant ramifications for nutrient uptake from streams, downstream transport of nutrients, and stream food webs (Amon 2002, Bunn et al. 1999, Hall and Meyer 1995, Hall and Tank 2003, Thorp 2002). Temperature, light levels, dissolved organic matter, nutrient availability, and discharge are important determinants of production and respiration rates in streams (Elser et al. 1990, Elser et al. 2000, Gausch et al. 1995, Uehlinger 2000). Wildfire impacts on stream physical and chemical properties are similar to the effects of land management disturbances such as logging,

riparian canopy removal, eutrophication and alteration of flow regime. Post-fire alteration of discharge regime, canopy loss, and nutrient enrichment have been shown to alter invertebrate assemblages and stream food webs (Minshall et al. 1997, Spencer et al. 2003, Vieira et al. 2004), but the effect of fire on whole-stream metabolism is as yet unknown.

The northern boreal forest is experiencing a period of rapid change, yet the importance of permafrost extent and wildfire on stream nutrient concentrations and stream function have not been established. The objectives of this research were two-fold. First, we examined the roles of permafrost and wildfire in determining stream nutrient concentrations by contrasting watersheds with varying extents of permafrost, and by comparing pre- and post-fire stream chemistry in a burned and a control watershed. Second, we assessed how the resulting patterns in stream chemistry impacted stream metabolism. Because permafrost influences DOM and nutrient delivery to streams, we hypothesized that streams draining watersheds with large extents of permafrost, and thus receiving greater DOM and lower nutrient inputs, would support higher respiration and reduced GPP. In addition, we hypothesized that wildfire would lead to increased inputs of inorganic nutrients to streams, thereby alleviating nutrient limitation and causing increased GPP. Finally, we hypothesized that combustion of soil carbon in the surrounding catchments would reduce DOM input to streams, leading to lower rates of ecosystem respiration.

Methods

Study sites

Our study was conducted in the Caribou Poker Creeks Research Watershed (CPCRW; 65.15°N, 147.5°W), located approximately 50 km NE of Fairbanks, Alaska. The CPCRW is a pristine, 104 km² watershed reserved for hydrological and ecological research. The climate of CPCRW is continental, with warm summers (mean = 16.4°C in July), cold winters (mean = -29°C in January), and low precipitation (411 mm, of which 31% falls as snow). The research site is located in the zone of discontinuous permafrost, with the extent of permafrost underlying sub-catchments ranging from 3 to 53%. This distribution of permafrost is largely determined by aspect and winter temperature, with permafrost found predominately on north-facing slopes and in valley bottoms (MacLean et al. 1999). Uplands on south facing slopes are dominated by hardwood forests of Alaskan paper birch (*Betula neoalaskana*) and quaking aspen (*Populus tremuloides*), whereas north-facing slopes generally have black spruce (*Picea mariana*), and feathermosses (*Pleurozium schreberi* and *Hylocomium schreberi*). Several alder species have a patchy distribution in both uplands (*Alnus viridis*) and valley bottoms (*Alnus incana*). Valley bottoms typically have saturated soil and support growth of mosses (e.g., *Sphagnum* spp., *Hylocomium*) and dwarf shrubs (e.g., *Betula nana*, *Salix* spp., *Vaccinium uliginosum*). The CPCRW is underlain by the Yukon-Tanana metamorphic complex, with north-facing slopes dominated by gelisols and south facing slopes by inceptisols (Ping et al. 2005). North-facing slopes generally have a well developed organic horizon and thin, weakly weathered mineral soil overlying fractured bedrock. Soils on north-

facing slopes are typically poorly drained with low pH (4 – 5), high organic carbon standing stocks and a small pool of base cations (Ping et al. 2005). In contrast, south-facing slopes have well drained soils with comparatively shallow organic horizons, thick mineral soil overlying fractured bedrock, and higher pH (5 – 6). Compared with north-facing slopes, the soil of south-facing slopes has less organic carbon but more exchangeable Ca and Mg, which increase with depth (Ping et al. 2005).

We studied a total of five watersheds. Four catchments varied in size from 5.2 to 10.0 km² and permafrost extents from 4 - 26% (C1, C2, C4 and P6 watersheds; Fig. 1). An additional stream was studied, which was located approximately 25 km northeast of the CPCRW boundary (Boston Creek; Table 1; Fig. 1). Within the CPCRW, the C4 catchment was partially burned in 1999 by a prescribed fire to assess fire and climate feedbacks in the boreal forest (FROSTFIRE project; Hinzman et al. 2003). The prescribed burn was of moderate intensity, covered 28% of the watershed area, and was largely restricted to the black spruce-dominated north-facing slopes with the riparian zone left mostly unburned. Two additional study catchments (P6 and Boston Creek) were extensively burned in the Boundary Fire during summer 2004. P6 drains a 7.0 km² catchment with 18% underlying permafrost. Boston Creek drains a 20.0 km² catchment; the permafrost extent is unknown for Boston Creek, but the northeasterly aspect and almost uniformly black-spruce vegetation suggest a high degree of underlying permafrost. The Boundary Fire burned 65% and 85% of the P6 and Boston Creek catchments, respectively, with burning generally restricted to black spruce-dominated slopes. The riparian zone at P6 was undisturbed along the 200 m study reach, whereas at

Boston Creek, over half the riparian zone along the 200 m study reach was burned during the fire.

Study overview

Impacts of fire on stream nutrient concentrations were assessed by comparing pre- and post-fire stream chemistry in the C1 and P6 watersheds which were well matched in catchment area, stream size, permafrost extent and dominant vegetation (Table 1). Similar catchment characteristics and similar pre-fire dissolved organic carbon (DOC) and nitrate concentrations made C1 the most appropriate control watershed against which to compare fire effects at P6. We have no pre-fire data for the Boston Creek catchment, so fire effects on stream chemistry were determined by comparison with the combined variability observed in the unburned watersheds. In addition to monitoring stream chemistry, we measured whole-stream metabolism in four streams (C2, C4, P6 and Boston Creek) during summer 2005 to evaluate fire and permafrost impacts on instream GPP and respiration. The unburned watersheds (C2 and C4) were used as controls against which fire effects on metabolism in P6 and Boston Creek were assessed.

Field methods

Stream chemistry and physical measurements

Stream stage height was measured continuously every 15 minutes using pressure transducers and Campbell Scientific dataloggers at Parshall flumes at the C2, C4 and P6 streams, and at a stream cross-section at Boston Creek. Rating curves were developed for

all sites from velocity and stream cross-section measurements. Campbell Scientific CS547A conductivity and temperature probes were installed at all sites. Stream water chemistry was measured during the summers of 2002 – 2005 for the CPRW sub-catchments. The C1, C2 and C4 streams were sampled each summer (June – August) from 2002 – 2005, and P6 during all but the first two months of 2004. The P6 stream was not sampled for chemistry prior to the 2004 Boundary Fire, but we initiated sampling as soon as access allowed after the fire (July). Boston Creek was sampled once in July, 2004 after passage of the wildfire, then from June – August 2005. Stream water samples were collected daily using ISCO autosamplers and an additional grab sample was collected every 1 to 2 weeks. Previous analyses at CPRW have shown that for most solutes, including nitrate and DOC, the concentration measured in autosampler collected samples stored for two weeks is not significantly different from grab sample concentrations ($R^2 = 0.95$; slope = 0.99; Jones et al. 2005). Samples were collected in acid washed HDPE bottles, stored in a cooler for transport to the lab and filtered within 12 hours using glass fiber filters (Gelman AE, 0.7 μm pore size). Samples were stored at 4°C until analysis (usually within 48 hours). When samples could not be analyzed immediately, samples were frozen until subsequent processing. Several sample dates were missed at each stream due to mechanical failure of the autosamplers. Stream temperature, conductivity and pH were measured every two weeks from 2002 – 2004, and weekly in 2005. Temperature was measured using a field thermometer, calibrated in the lab with a high precision thermometer, and electrical conductivity was measured with a handheld conductivity meter. pH was measured in the laboratory using a Beckman 300

pH meter on grab samples collected in 125 ml HDPE with no head space to prevent degassing of CO₂.

Stream metabolism

In-stream GPP, respiration, and net ecosystem productivity (NEP) were calculated from diel change in dissolved oxygen concentration using the single-station, whole-stream technique (Bott 1996, Odum 1956). Metabolism was measured approximately weekly in 2005 for 24 hours at C2, C4, P6 and Boston Creek. Because interior Alaska experiences up to 20 hours of daylight during the summer, we calculated daily respiration from the minimum hourly change in dissolved oxygen over the day. On each date, dissolved oxygen was measured every 15 minutes using Sensorex dissolved oxygen probes connected to Campbell Scientific dataloggers. Probes were calibrated on each sample date using the spectrophotometric dissolved oxygen method (Roland et al. 1999). Triplicate 250 ml water samples were collected in BOD bottles from the thalweg (i.e. deepest part of the stream channel) of each stream, and immediately fixed in the field with 2.5 ml MnSO₄ and 2.5 mL KOH-KI. To express metabolism on an areal basis, stream widths were measured once at all sites in 2005. All of the study streams have incised stream channels, and, as a consequence, width does not vary appreciably with discharge. Width was measured at 5 m intervals over a 200 m reach and averaged.

Oxygen exchange with the atmosphere was determined using the steady-state injection technique (Hibbs et al. 1998, Mulholland et al. 2001). We conducted eight injections across a range of flows to develop a model relating evasion rate to discharge.

Evasion was quantified in the C2 and C4 streams, which have the lowest and highest stream flows at CPCRW, respectively. Conservative and volatile gas tracers (Cl^- or Br^- , and SF_6) were simultaneously injected into the study stream 300 m upstream from a dissolved oxygen probe using a metering pump (flow rate = 50 ml/min) until steady-state was achieved at the downstream station (approximately 3 hours). Prior to each injection, triplicate water and triplicate gas samples (30 ml stream water collected in 60 ml polyethelene syringes) were collected from the thalweg at sampling stations every 50 m downstream from the injection site (300 m total reach length). Stream discharge (Q) at the time of injection was determined using the dilution gauging technique and calculated as

$$Q = \frac{(C_{inj})}{(C_{pl} - C_{bg})} q_{inj} \quad (1)$$

where: C_{bg} is background Cl^- or Br^- concentration, C_{inj} is the concentration of Cl^- or Br^- in the injectate, C_{pl} is stream plateau Cl^- or Br^- concentration, and q_{inj} is the tracer flow rate (Stream Solute Workshop 1990). Stream velocity (v) was calculated as

$$v = \frac{d}{t_1 - t_0} \quad (2)$$

where d is the distance traveled by the conservative solute, t_0 is the injection start time and t_1 is the time when solute concentration at the downstream station reached half plateau concentration. Stream depth (z) was calculated by dividing discharge by velocity and average stream width.

Evasion of SF₆ ($k_{(SF_6)}$) was calculated as

$$k_{(SF_6)} = \frac{1}{\tau} \ln \frac{G_1 C_2}{G_2 C_1} \quad (3)$$

where τ is the hydrologic travel time between the up and downstream sampling points, G is the steady-state tracer gas concentration, C is the conservative tracer concentration, and the subscripts 1 and 2 are up and downstream sampling locations, respectively (Generaux and Hemond 1992). SF₆ evasion was converted to O₂ evasion using the equation

$$k_{O_2(T_s)} = k_{SF_6} \left(\frac{Sc_{SF_6}}{Sc_{O_2}} \right)^n \quad (4)$$

where, Sc_{SF_6} and Sc_{O_2} are the Schmidt numbers for SF₆ and O₂, respectively (Wanninkhof 1992), and $n = 0.7$ (Generaux and Hemond 1992). Oxygen evasion was converted to evasion at 20°C ($k_{O_2(20^\circ C)}$) before being used to develop the evasion model using the formula

$$k_{O_2(20^\circ C)} = \frac{k_{O_2(T_s)}}{1.024^{(T_s-20)}} \quad (5)$$

where $k_{O_2(T_s)}$ is evasion rate at ambient stream temperature (T_s ; Elmore and West 1961).

To develop an evasion model for CPCRW streams, we compared evasion against velocity and depth via regression analysis using the general form of the equation developed by Bennett and Rathburn (1972)

$$k_{O_2(20^\circ C)} = \frac{av^b}{z^c} \quad (6)$$

where a , b and c are constants.

Laboratory analysis

Stream chemistry - Anions (Cl^- , Br^- , NO_3^- , SO_4^{2-}) and cations (NH_4^+ , Ca^{2+} , Mg^{2+} , Na^+ , K^+) were quantified using a Dionex DX-320 Ion Chromatograph. Soluble reactive phosphorus (SRP) was measured using the molybdate blue method (American Public Health Association, 1998). DOC concentration was measured using a Shimadzu 5000 total organic carbon analyzer, which was plumbed to an Antek 7050 nitric oxide chemoluminescent detector to quantify total dissolved nitrogen (TDN). Dissolved organic nitrogen (DON) was calculated as $TDN - (NH_4^+ + NO_3^-)$. Dissolved oxygen samples were acidified with 2.5 ml 12N H_3PO_4 and the light absorbance at 430 nm read using a Beckman DU 640B spectrophotometer according to the method described by Roland et al. (1999). SF_6 was measured by allowing syringes to warm to room temperature, adding a 30 ml helium headspace, and allowing equilibration for two hours. The headspace was then injected into a Varian CP 3800 gas chromatograph (5 ml) and measured with an electron capture detector.

Data analysis

Differences in stream solute concentrations (DOC, nitrate and sulfate) among CPCRW sub-watersheds were tested using one-way ANOVA. Paired data from the C1

and P6 watersheds were used to establish the summer average difference in solute concentrations between the two sites. The differences obtained were compared with the pre-fire contrast data. The null hypothesis was that there was no change in solute concentration differences between C1 and P6 streams after fire. If a difference between pre- and post-fire solute concentrations was found, the pre-fire concentration difference between C1 and P6 was used to calculate an expected summer average concentration, from which percent change following fire was calculated. To test for the degree of difference between burn and control stream pre-fire chemistry we used the Wilcoxon signed-rank test ($\alpha = 0.05$). Post-fire changes in stream solute concentrations between the burned (P6) and control (C1) streams were examined using the Wilcoxon rank-sum test ($\alpha = 0.05$). Non-parametric tests were used, when testing differences between stream solute concentrations, due to violation of the assumptions of normality and constant variance.

The metabolism data represent repeated measurements taken throughout the summer, but due to the small sample size and lack of temporal trends, a t-test was used to examine differences between burned (P6 and Boston Creek; $n = 10$) and unburned (C2 and C4; $n = 11$) watersheds. Simple linear regression was used to quantify relationships between environmental variables and GPP and respiration. The use of non-replicated treatment and control streams in this study has resulted in pseudoreplication. Pseudoreplication could not be avoided due to the limited number of streams available for study in the CPRW. Natural log transformations were used when required to correct non-normal distributions and unequal variance of environmental and metabolism data.

Results

Stream solute chemistry

Across the four year study period, stream nitrogen concentration varied significantly among watersheds. Comparing stream chemistry prior to the Boundary Fire, the mean nitrate concentration in the low-permafrost watershed (C2), ranged from 515 – 568 $\mu\text{gN/L}$ compared with 236 – 292 $\mu\text{gN/L}$ in the high-permafrost watershed (C1; Fig. 2). The medium-permafrost watersheds had variable nitrate concentrations. C4 stream nitrate concentration was similar to the low-permafrost watershed (567 – 606 $\mu\text{gN/L}$), whereas the nitrate concentration in P6 was similar to the high-permafrost watershed (282 – 292). In 2002, stream nitrate concentrations in the C1 and P6 streams were not statistically different. During the remainder of the study (2003 – 2005), nitrate concentrations were statistically different in all streams ($p < 0.05$; Fig. 2). Ammonium, in contrast, exhibited greater inter-annual variation before the Boundary Fire within individual sites than among sites. Pre-fire stream ammonium concentrations ranged from 16 – 32 and 12 – 21 $\mu\text{gN/L}$ at C2 and C4, respectively. Ammonium concentration was marginally lower at P6, ranging from 10 – 14 $\mu\text{gN/L}$, whereas ammonium concentration ranged between 14 – 20 $\mu\text{gN/L}$ at C1.

Stream DOC concentration varied considerably in streams of CPRW sub-catchments. DOC concentrations in the C2 and C4 streams (low- and medium-permafrost watersheds, respectively) were not significantly different throughout the four year study period ($p > 0.05$; Fig. 3). Summer average DOC concentrations ranged from 2.8 to 3.8 mgC/L in C2 and from 2.3 to 4.1 mgC/L in C4. From 2002 – 2004, DOC

concentration in C1 stream was significantly higher than both the C2 and C4 streams and ranged from 3.5 to 5.5 mgC/L ($p < 0.05$; Fig. 3). In 2005, average DOC concentration in C1 was 4.3 mgC/L, and was not significantly different from DOC concentration in the C2 and C4 streams. Pre-fire (2002-2003) DOC concentration in P6 ranged from 6.2 to 6.6 mgC/L, and was significantly higher than DOC concentration in the other study streams ($p < 0.05$; Fig. 3). Interestingly, DON concentration exhibited less variation among streams than did DOC. Pre-fire DON concentration ranged from 272 to 399 $\mu\text{gN/L}$ across the four CPCRW sub-catchments (Table 2).

Sulfate concentration varied significantly among CPCRW streams. In 2002, sulfate concentration was not significantly different in the C4 and C1 streams ($p > 0.05$; Fig 4). In contrast, from 2003 – 2005, sulfate concentration was significantly different in all streams ($p < 0.05$; Fig. 4). Average summer sulfate concentration was lowest in the high-permafrost watershed (C1), and ranged from 2.7 to 3.4 $\text{mgSO}_4^{2-}/\text{L}$ (Fig. 4). Sulfate concentrations were intermediate in the C2 and C4 streams across the four year study period, ranging from 5.8 to 6.1 in C2 and ranging from 6.0 to 7.1 $\text{mgSO}_4^{2-}/\text{L}$ in C4 (Fig. 4). Pre-fire (2002 – 2003) average sulfate concentration in P6 varied from 10.2 to 13.0 $\text{mgSO}_4^{2-}/\text{L}$, or almost double the concentration in the other study streams. At all sites, across all pre-fire years, stream soluble reactive phosphorus concentration was at or below detection level ($< 1 \mu\text{gP/L}$).

Stream cation concentrations varied among study streams, but always occurred in the order calcium > magnesium > sodium > potassium (Table 2). Total cation concentration (Ca^{2+} , Mg^{2+} , Na^+ and K^+) was lowest in the stream draining the high-

permafrost watershed (C1). The medium-permafrost watersheds (C4 and P6) had similar concentrations of all cations (Table 2). Stream magnesium, potassium and sodium concentrations were similar in the low (C2) and medium-permafrost watersheds (C4 and P6). Calcium concentration in the low-permafrost watershed (C2) was intermediate between that in the medium (C4 and P6) and high-permafrost (C1) watersheds.

There were marked differences in stream nitrate, sulfate and DOC concentrations among study streams. In particular, stream DOC and nitrate concentrations indicated two chemically similar groups of streams in CPCRW. The first group consists of the C2 and C4 streams, which despite having low and medium-permafrost extents, had the same pattern of high nitrate and low DOC concentrations throughout the study period. The second group, the C1 (high-permafrost) and P6 (medium-permafrost) streams, were considerably higher in DOC and lower in nitrate than C2 and C4. Sulfate concentration behaved similarly to nitrate in all streams except P6, which had substantially higher pre-fire sulfate concentration than the other streams. The combination of solutes in each group suggests considerable difference in dominant catchment hydrologic flowpaths contributing to streamflow.

Wildfire impacts on stream chemistry

The C1 stream had pre-fire chemistry most similar to P6 and was used as a control stream to detect changes in nutrient chemistry after the fire. Pre-fire (2002 – 2003) stream nitrate and ammonium concentrations were not significantly different in the C1 and P6 streams ($p < 0.05$). Dissolved organic carbon concentration tended to be

higher in the P6 stream than in the C1 stream, with summer average concentration of 5.1 and 6.4 mgC/L for C1 and P6, respectively ($p > 0.05$). These values were higher than the summer averages for C2 and C4 (3.8 and 4.1 mgC/L).

Wildfire had a dramatic effect on stream nitrate concentration (Fig. 5). In 2004, immediately following the fire, nitrate concentration was 22% higher in the burned than the control stream ($p < 0.05$), and in 2005 nitrate concentration was 45% higher in the burned watershed than the control ($p < 0.05$; Fig. 7). Moreover, in 2005, nitrate concentration was as high as 1200 $\mu\text{gN/L}$ in P6, almost double the concentration in the control stream. When compared with the control watershed, stream ammonium concentration in the burned watershed increased immediately after fire but was lower than the pre-fire concentration the following summer. In 2004, ammonium increased by 40% to 44 $\mu\text{gN/L}$ at P6 compared with 57 $\mu\text{gN/L}$ at C1 ($p < 0.05$), whereas in 2005 ammonium concentration had declined by 67% to 37 $\mu\text{gN/L}$ in P6 compared with 72 $\mu\text{gN/L}$ in C1 ($p < 0.05$). Throughout the study period, stream ammonium concentration was higher in the control than the burned watersheds.

After the fire, DOC concentration declined in the stream draining the burned watershed relative to the control ($p < 0.05$; Fig. 7). Pre-fire mean DOC concentration was 1.7 mgC/L higher in P6 than in C1, whereas the post-fire the difference was 0.9 mgC/L. Wildfire resulted in an increase in stream DON the year of disturbance, followed by a net decrease one summer post-fire. In 2004, stream DON concentration had increased by 32 $\mu\text{gN/L}$ in the burned watershed compared with the control. In contrast,

in 2005, DON concentration was 92 $\mu\text{gN/L}$ lower in P6 than in C1, equal to a 36% decline in DON.

Stream sulfate concentration increased in the burned watershed. Sulfate concentration increased by 24% in 2004 and 18% in 2005 at the burned site when compared with the control stream ($p < 0.05$; Fig. 4). Wildfire had a variable effect on stream concentrations of cation species (Table 2). Stream sodium concentration was elevated immediately following the fire, but no difference was detected the subsequent year. In contrast, potassium concentration in the burned stream was unchanged the year of the fire but was elevated in 2005. Compared with the control stream, calcium concentration in P6 was marginally elevated the year of the wildfire, and significantly reduced twelve months following fire.

Boston Creek post-fire solute concentrations were all within the range observed across CPCRW streams in summer 2005. Average nitrate, ammonium and DON concentrations were 447, 56 and 221 $\mu\text{gN/L}$, respectively. The average post-fire concentration of DOC at Boston Creek was 4.9 (mgC/L), which was higher than the maximum for unburned streams (3.6 mgC/L) but lower than in P6 (5.6 mgC/L). Mean sulfate concentration in Boston Creek ($10.3 \text{ mgSO}_4^{2-}/\text{L}$) was also higher than the maximum observed in the unburned study sites ($7.1 \text{ mgSO}_4^{2-}/\text{L}$), but lower than in P6 ($13.9 \text{ mgSO}_4^{2-}/\text{L}$).

The 2004 Boundary Fire resulted in increased concentrations of nitrate, sulfate and cations and reduced DOC, DON and ammonium concentrations in the burned stream (P6) when compared with the control. Fire effects on stream solute concentrations in

Boston Creek cannot be conclusively established due to the lack of pre-fire data.

However, the combination of stream solute concentrations in Boston Creek is consistent with the pattern observed for the P6 stream.

Ecosystem metabolism

Across the two years of evasion measurements, the model we developed incorporated discharges from 19 to 60 L/s and velocities from 0.13 to 0.24 m/s. The measured oxygen evasion was best described by the equation

$$k_{O_2(20^\circ C)} = \frac{0.0008v^{-0.6031}}{z^{1.7813}} \quad (7)$$

Across study sites, summer discharge ranged from 28 to 134 L/s, including nine sample dates that fell outside the model's range of inference (discharge 19 – 60 L/s). To calculate evasion over the full range of observed discharges we used equation 7 when discharge was less than 60 L/s, but when discharge was greater than 60 L/s, we used the evasion model developed by Bennett and Rathburn (1972)

$$k_{O_2} = \frac{0.0039v^{0.607}}{z^{1.689}} \quad (8)$$

GPP was more correlated with burn history than with extent of permafrost (Fig. 9a). Despite differing extents of permafrost, GPP in the C2 and C4 watersheds was similar, with a summer average of 1.5 and 0.9 gO₂ m⁻² d⁻¹ at C2 and C4, respectively. GPP was 50% higher in the burned streams, averaging 2.4 gO₂ m⁻² d⁻¹ for both P6 and Boston Creek (p < 0.05), than in the unburned sites. Maximum summer GPP at P6 and

Boston Creek was 3.4 and 3.6 $\text{gO}_2 \text{ m}^{-2} \text{ d}^{-1}$, respectively. GPP remained fairly constant at C2 and C4 throughout the summer, whereas at the burned sites GPP exhibited considerable variation (Fig. 7a).

Ecosystem respiration in streams draining burned watersheds was significantly higher than in the unburned watersheds ($p < 0.05$; Fig. 7b). Respiration was lowest in the C4 stream, which had an average respiration rate of $1.20 \text{ gO}_2 \text{ m}^{-2} \text{ d}^{-1}$. C4 maximum respiration occurred early to mid-summer (1.4 and $1.7 \text{ gO}_2 \text{ m}^{-2} \text{ d}^{-1}$) then declined to a low of $0.4 \text{ gO}_2 \text{ m}^{-2} \text{ d}^{-1}$ at the end of the study period. The summer average respiration rate at C2 was $4.5 \text{ gO}_2 \text{ m}^{-2} \text{ d}^{-1}$ with the lowest values on the first two sample dates (3.3 and $2.9 \text{ gO}_2 \text{ m}^{-2} \text{ d}^{-1}$), after which respiration ranged between 4.4 and $5.8 \text{ gO}_2 \text{ m}^{-2} \text{ d}^{-1}$. Respiration was slightly lower in the Boston Creek stream than in the C2 watershed, with a summer average of $3.9 \text{ gO}_2 \text{ m}^{-2} \text{ d}^{-1}$. A minimum rate of $2.4 \text{ gO}_2 \text{ m}^{-2} \text{ d}^{-1}$ was recorded in mid-summer at Boston Creek, after which respiration remained between 3.7 and $4.7 \text{ gO}_2 \text{ m}^{-2} \text{ d}^{-1}$. The highest respiration rate was recorded in the P6 stream, with a summer average of $6.6 \text{ gO}_2 \text{ m}^{-2} \text{ d}^{-1}$, and a range of 4.9 to $7.9 \text{ gO}_2 \text{ m}^{-2} \text{ d}^{-1}$. Net ecosystem productivity was not significantly different between the burned and control watersheds ($p > 0.05$; Fig. 7c). The C2, P6 and Boston Creek streams had negative net production rates throughout the summer because respiration always exceeded GPP, whereas the C4 stream had negative production for the first half of the study period and positive net production on the last two sample dates.

Regression analysis revealed that the changes to stream solute concentrations following fire were not the underlying cause of the different metabolism rates in the

burned and control streams. In all sites both respiration and GPP were unrelated to stream temperature ($p > 0.05$; Fig. 8a and 9a, respectively). There was no correlation between respiration rate and either DOC or DIN concentration in any of the study streams ($p > 0.05$; Fig 8b and 8c). Similarly, stream DIN concentration was not correlated with GPP in the burned watersheds, but was negatively correlated with GPP in the control streams ($p > 0.05$; Fig. 9b). ($p < 0.05$, $R^2 = 0.53$; Fig 9). Also, in the control streams, DIN was positively correlated with stream discharge ($p < 0.05$, $R^2 = 0.56$; Fig 10), while GPP was negatively related to discharge ($p < 0.05$, $R^2 = 0.53$, Fig. 10). Clearly, the observed correlation between GPP and DIN may be a function of their individual relationships with discharge. Therefore, these results suggest that DOC, nitrate, and ammonium do not regulate metabolism rates in CPCRW streams, and fire-induced alteration of these nutrients did not impact production or respiration in streams draining burned watersheds.

Discussion

Permafrost and stream chemistry

Stream solute concentrations in sub-catchments of CPCRW varied widely among watersheds. Permafrost serves as a barrier to the infiltration of precipitation and has been proposed as an important regulator of stream solute concentrations in the region of discontinuous permafrost (Carey 2003, MacLean et al. 1999, Petrone et al. 2006).

Streams in catchments underlain by a large extent of permafrost receive greater DOC and

reduced nutrient inputs, whereas streams in low-permafrost watersheds receive relatively greater nutrient and reduced DOC inputs (Jones et al. 2005, MacLean et al. 1999, Petrone et al. 2006). Permafrost is thought to control the depth of subsurface flowpaths and, thus, the chemical character of inputs to streams. High DOC concentration in the surface soil solution has been attributed to DOC leaching from leaf litter and high biological activity, whereas the reduction of DOC as flow passes through mineral soil is thought to be due to abiotic sorption of DOC to soil particles (McDowell and Likens 1988). In CPRW, nitrate, sulfate, sodium, and calcium concentrations are considerably higher in groundwater than in shallow soil, which suggests weathering is the primary source of these solutes to streamflow (Petrone et al. 2006).

In this study, despite significant differences in DOC and nutrient concentrations among streams, solute concentration did not vary predictably with permafrost extent. The low DOC and high nutrient concentrations in streams draining low (C2) and medium (C4) permafrost watersheds were considerably different from the high DOC and low nutrient concentrations of the high-permafrost (C1) stream. However, although the P6 watershed had a similar permafrost extent to C4, stream DOC and nitrate concentrations in the P6 watershed were more similar to those in high-permafrost watershed (C1). Permafrost extent may contribute to variation in stream chemistry among the sub-watersheds of CPRW; however, other variables also appear to be important in driving the observed pattern of solute concentrations. In a previous study of catchment hydrology and solute dynamics in CPRW, a low-permafrost watershed was found to have a significantly smaller catchment area (1.6 – 4.4%) contributing to stream flow

during two storms than a high-permafrost watershed (11.8 – 14.5%; Petrone et al. In press). Catchment contributing area, and the location of permafrost in relation to contributing area may be important sources of variation in stream chemistry in regions of discontinuous permafrost. Also, soil nutrient concentrations vary significantly between black spruce and mixed hardwood (birch and aspen) stands in interior Alaska (Troth et al. 1976), and undoubtedly contribute to differences in stream chemistry among CPCRW streams.

Wildfire and solute dynamics

Comparison of pre- and post-fire control (C1) and burned (P6) stream chemistry indicates that wildfire caused immediate and sustained changes to stream solute concentrations in the burned catchment. The reduction in DOC concentration (15%) following fire could be due to a number of processes, including combustion of soil carbon stocks, transformation of soil carbon to more recalcitrant and less soluble carbon compounds, or decreased microbial mineralization of organic carbon due to a post-fire reduction in microbial biomass (Certini 2005). Following the prescribed burning of the C4 watershed in 1999 (FROSTFIRE; Hinzman et al. 2003), both soil water DOC and water extractable organic carbon from the organic horizon of burned slopes declined (Shibata et al. 2003). Similarly, the reduction in stream DOC concentration in the burned watershed in the present study was likely due in part to combustion and conversion of soil organic carbon stocks.

Elevated stream nitrogen concentration, as observed in this study, is a common response to fire (Bayley and Schindler 1991, Hauer and Spencer 1998, Minshall et al. 1997, Williams and Melack 1997). Increased soil ammonium concentration, commonly observed following fire, is potentially due to increased microbial or thermal decomposition of vegetation and soil organic matter (Chorover et al. 1994, Wan et al. 2005). Any increase in available ammonium has potential to be rapidly nitrified and in turn increase soil nitrate availability (Certini 2005, Wan et al. 2001). Despite evidence that soil nitrification is not stimulated in black spruce stands of CPRW following fire (Lyle 2006, Smith et al. 2000), stream nitrate concentration increased in this study, suggesting decreased microbial competition with vegetation for nitrate and hence increased nitrification. Alternatively, the pattern of increased nitrate concentration may be due to increase in soil active depth. Increased thaw depth initiated by wildfire (latent heat transfer and lower surface albedo) may have improved soil drainage and increased aeration, thus reducing nitrate removal via denitrification.

Comparing the results of our study with a previous study conducted in CPRW on fire impacts on stream chemistry (FROSTFIRE; Hinzman et al. 2003), stream nitrate in the former study increased for a short period during a single post-fire storm, but was not elevated at any other time (Petroni et al. In press). The difference between the fires in the response of stream chemistry is likely due to extent of watershed burned. The 2004 wildfire at P6 burned 65% of the catchment, compared with 28% of C4 burned during FROSTFIRE. In addition, whereas the riparian zone was not burned in the P6 watershed, the fire burned closer to the stream than during the FROSTFIRE burn. The

type of vegetation and soil burned may also play an important role in nutrient export to streams. In a study of soil chemistry following wildfire in interior Alaska, soil solution nitrogen was slightly elevated in black spruce forests, in contrast to soil in aspen, birch, and white spruce stands where soil nitrogen concentrations declined following fire (Dyrness et al. 1989). The 1999 and 2004 fires both burned black spruce predominately, indicating that the difference in catchment area burned and proximity to the stream are the likely causes of the sustained elevation of stream nitrate at P6.

Stream phosphorus concentration has been shown to either dramatically increase (Earl and Blinn 2003, Prepas et al. 2003), albeit for a brief period, or be unchanged by fire (Minshall et al. 1997, Stephens et al. 2004). Prior to the 2004 wildfire, SRP concentration in the C1 and P6 streams was near detection limit ($< 1 \mu\text{gP/L}$), and we observed no detectable post-fire increase in stream SRP concentration at P6. Other studies in the boreal forest, however, have reported a change in stream phosphorus concentration. In a study in the boreal forest of Canada, total dissolved phosphorus was unchanged after fire, but particulate phosphorus increased significantly during periods of high discharge (Prepas et al. 2003). In northwest Montana, stream phosphorus concentration was over five-fold higher in burned than control streams, and this difference was sustained for a period of several years (Hauer and Spencer 1998). The observed increase in stream nitrate, sulfate, and cation concentrations in the burned stream in this study may have been accompanied by increased SRP concentration. However, any difference in SRP concentration after fire was not undetected because of very low ambient stream SRP concentration and/or rapid biological uptake.

Metabolism, wildfire and permafrost

Stream respiration and productivity have been shown to be influenced by various environmental factors, including light (Acuña et al. 2004, Mulholland et al. 2001, Young and Huryn 1996), temperature (Lamberti and Steinman 1997, Sinsabaugh 1997), nutrient concentrations, organic matter supply (Acuña et al. 2004, Kreuzweiser and Capell 2003), discharge (Lamberti and Steinman, 1997, Uehlinger 2000, Young and Huryn 1996), and catchment disturbance (Houser and Mulholland 2005). In this study, we aimed to determine fire effects on stream nutrient dynamics, and then in turn, how altered nutrient concentrations may have affected rates of metabolism. We hypothesized that increased nitrogen and phosphorus concentrations would alleviate nutrient limitation and stimulate primary production. Also, we hypothesized that decreases in stream DOC concentration would result in reduced respiration rate in burned catchments. When we grouped streams by burn history, both GPP and respiration were higher in streams draining burned catchments than unburned catchments. There was no significant correlation between respiration and GPP with nutrients and DOC. Therefore, the higher rates of metabolism in the burned sites do not appear to be correlated with the fire effects on stream nutrient concentrations that we observed.

Gross primary productivity in CPCRW streams does not appear nitrogen limited, but rather production rates may be controlled by phosphorus availability. Despite the large increase in stream nitrogen concentration that occurred following the Boundary Fire, this did not stimulate GPP. In fact, GPP and DIN were not correlated in the burned watersheds but were negatively correlated in the control watersheds. A similar negative

relationship between metabolic activity and nitrogen concentration was found in a study of eleven small streams in Wyoming (Hall and Tank 2003). Whereas the nitrate concentration was very low in the streams in Wyoming (mean 2 - 24 $\mu\text{gN/L}$; Hall and Tank 2003), stream nitrate concentration was consistently over 200 $\mu\text{gN/L}$ in CPCRW. Moreover, the C2 and C4 streams had the highest stream nitrate concentrations and the lowest rates of gross primary productivity. The negative association we found between DIN and GPP is likely an artifact of the negative relationship between GPP and discharge (Fig. 10). The periods of higher discharge during this study were not associated with flooding or bed-moving spates, and hence declines in GPP are not likely caused by loss of algal standing stocks. Alternatively, the reduced GPP during higher flows may be due to light attenuation from increased flow depth and turbidity (Young and Huryn 1996). Given the very low concentration of SRP in CPCRW streams, limitation of production by this nutrient is highly likely, as was found in another study in interior Alaska (Slavik et al. 2004). Considering the low ambient concentration of SRP in CPCRW streams, even a minor SRP increase may have been sufficient to stimulate GPP in streams draining burned watersheds.

Stream respiration rates in CPCRW were unrelated to the physical and chemical parameters measured during this study, but instead may be related to stream DOC bioavailability (Findlay et al. 2003, Kreutzweizer and Capell 2003) or extent of the hyporheic and transient storage zones (Fellows et al. 2001, Mulholland et al. 2001). Carbon bioavailability has been found to be both positively (Striegl et al. 2005) and negatively (Kawahigashi et al. 2004) correlated with permafrost extent. The correlation

between permafrost extent and carbon bioavailability in CPCRW streams is unknown, but significant differences in stream carbon concentration and catchment stand type suggest differences in carbon lability among streams are likely. Similarly, the variability in catchment permafrost extent and soil saturation among CPCRW watersheds may result in differences in size of the hyporheic zone and stream transient storage among watersheds. Although the mechanism behind differences in metabolism among streams remains unresolved, we propose GPP and respiration are linked to differences in stream SRP concentration following fire and to differences in carbon bioavailability due to differences in stream and catchment hydrology.

Warming, wildfire, permafrost and ecosystem metabolism

Climatic warming over the last century has resulted in a 2.1% reduction in permafrost extent at CPCRW, and over one-third of the watershed now has either unstable or actively thawing permafrost (Hinzman et al. 2006). The black spruce forests typical of areas underlain by permafrost are highly flammable, and susceptible to disturbance by wildfire (Kasischke et al. 2006). Fire is likely to alter permafrost extent by dramatically changing the thermal balance of soil through increasing soil temperatures via loss of the insulating organic mat, reduced vegetation cover, and associated increases in solar radiation and through reduced albedo from soil blackening and charcoal formation (Hinzman et al. 2006, Ping et al. 2006). After fire, black spruce may reestablish, leading to stabilization of permafrost, or, alternatively, black spruce forests may be replaced with deciduous stands (Chapin et al. 2004) under which further

degradation of the underlying permafrost is likely. Fire effects on permafrost extent could be substantial under a changing climate. Where permafrost is near the point of thawing, wildfire may accelerate the rate of degradation.

In the region of discontinuous permafrost, wildfires may impact stream chemistry and function in two ways. First, immediate changes to stream solute concentrations after fire may indeed alleviate nutrient limitation and increase rates of whole-stream metabolism. Second, fire-permafrost feedbacks may substantially alter the extent of underlying permafrost, with an as yet unknown effect on stream chemistry and ecosystem function. The precise role of permafrost extent on catchment hydrology and stream chemistry and function has not been clearly established, but during this period of rapid change in the region of discontinuous permafrost remains a question of significant importance.

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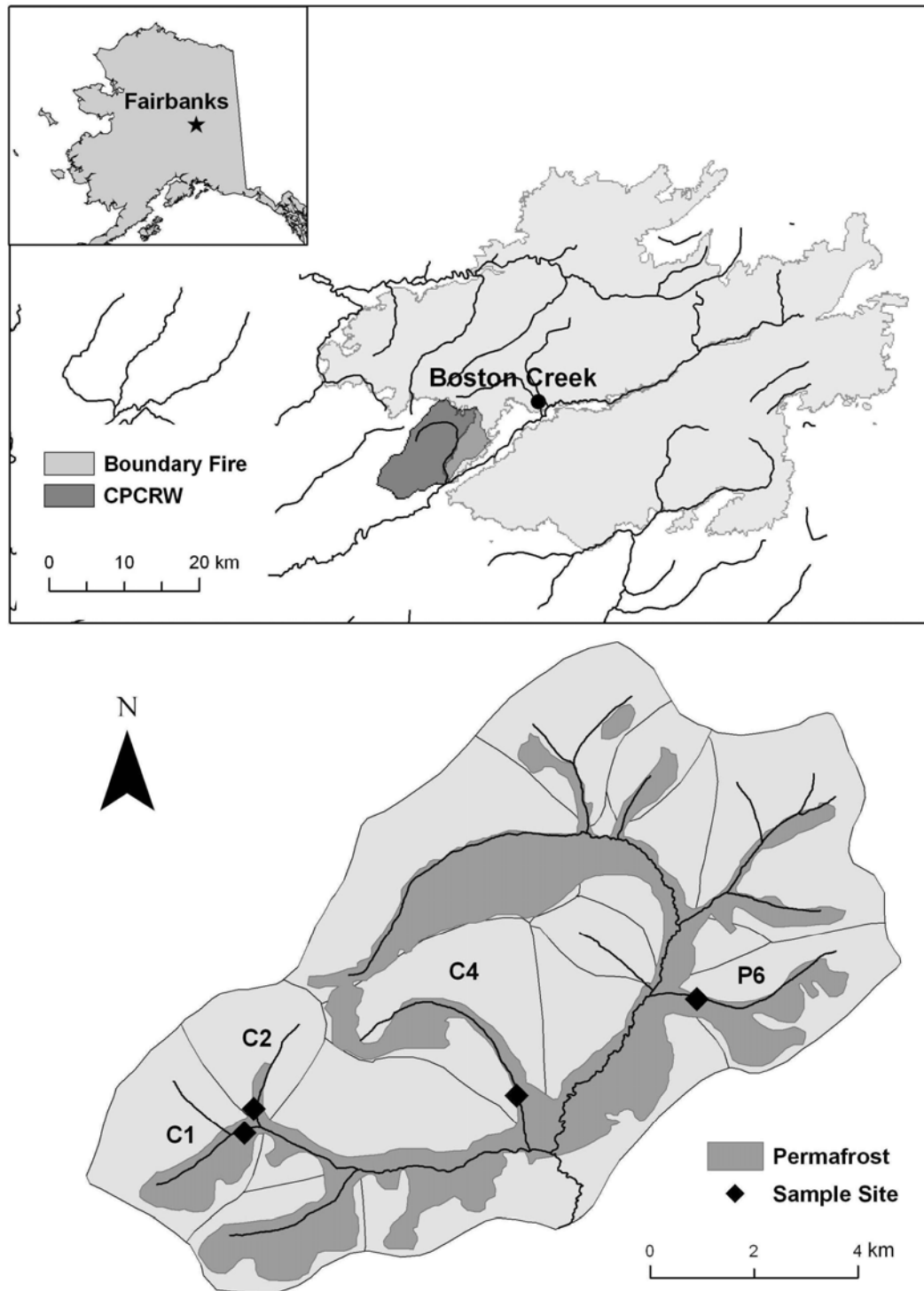


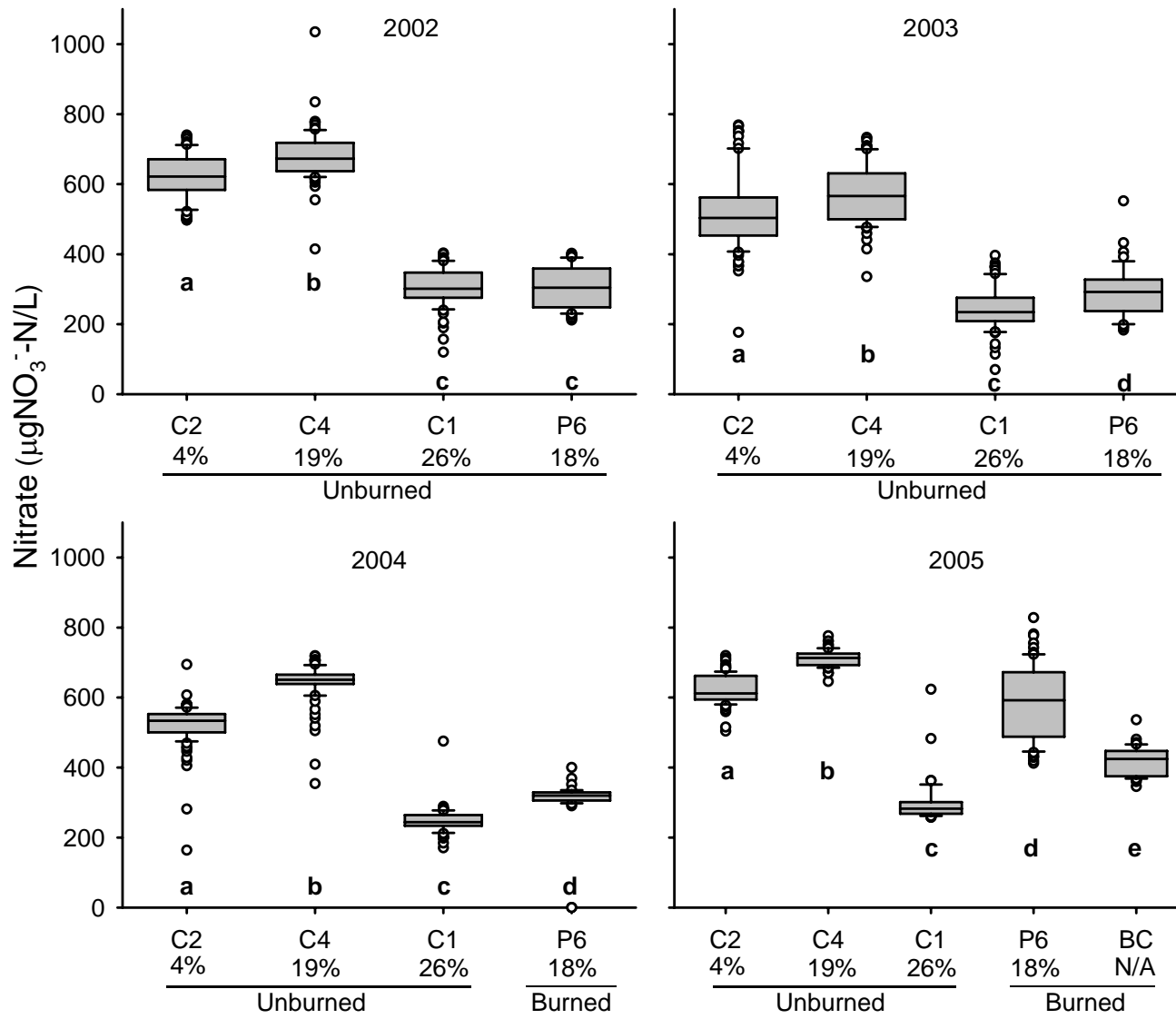
Figure 1. Caribou Poker Creeks Research Watershed (CPCRW) northeast of Fairbanks, Alaska. Location of the Boundary Fire and the Boston Creek research site in relation to the CPCRW. Map of permafrost distribution and study sites in CPCRW.

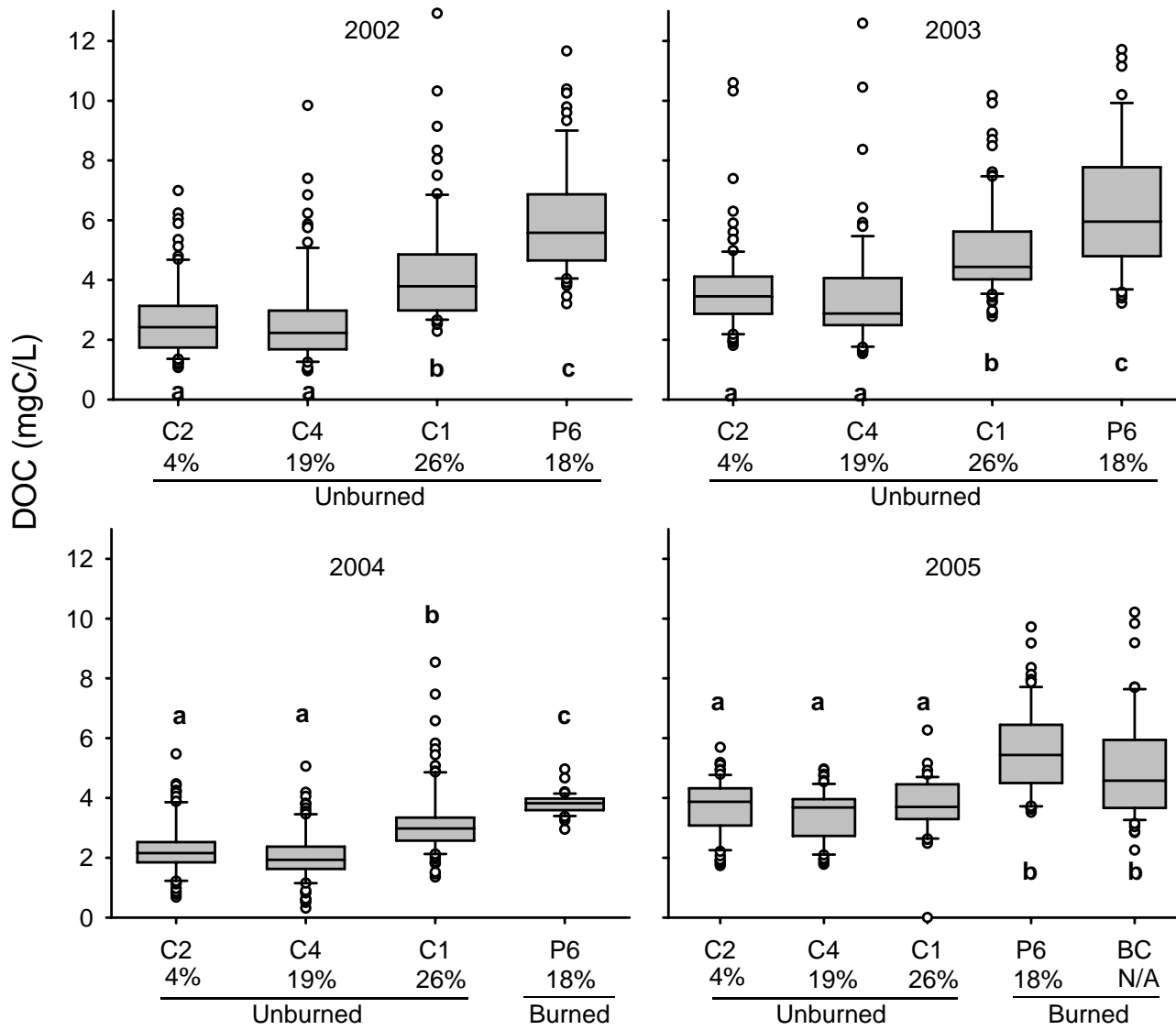
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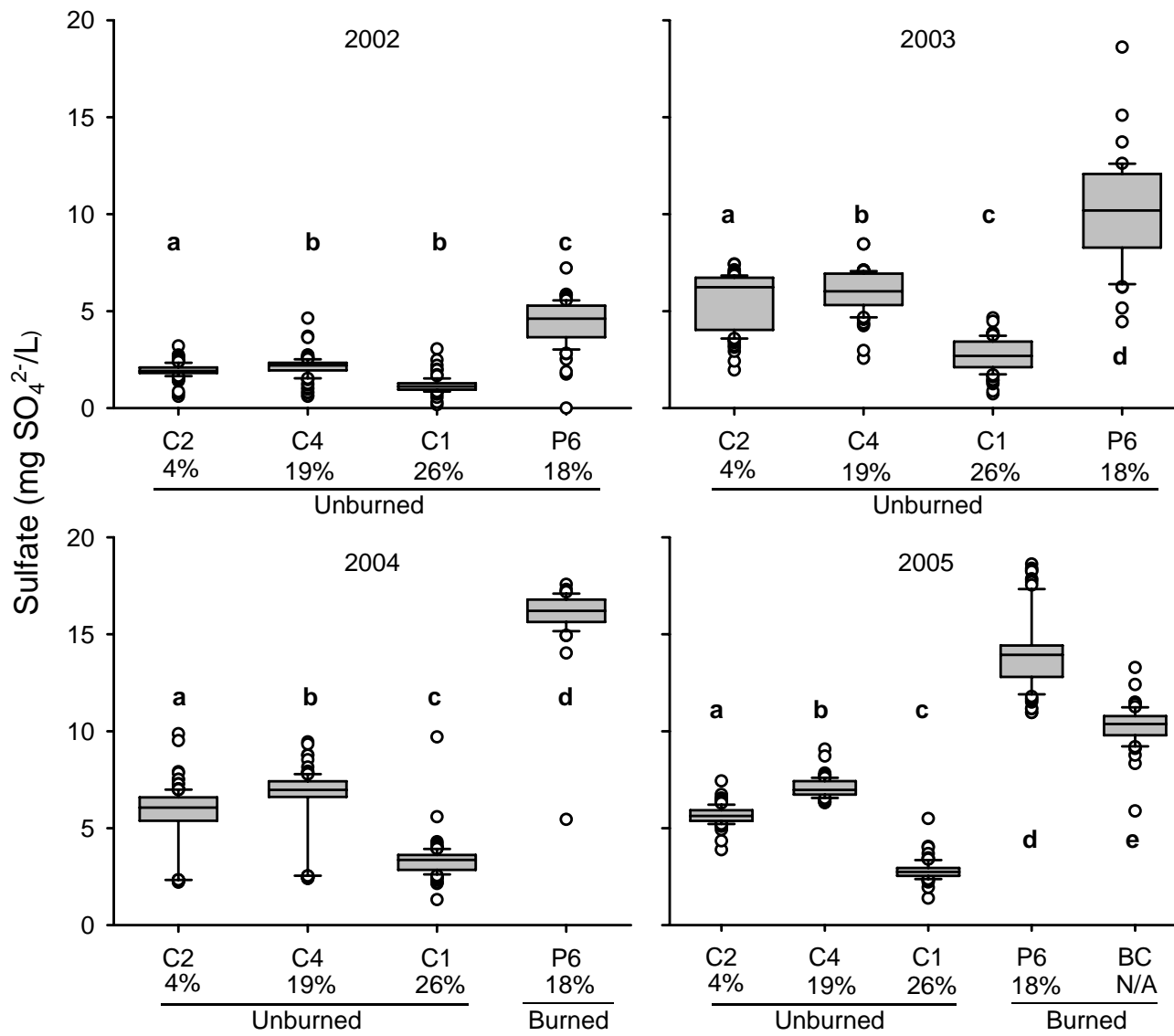
Figure 2. Stream nitrate concentration in four sub-catchments of the Caribou Poker Creeks Research Watershed (June – August, 2002 – 2005) and Boston Creek (June – August, 2005). Boxplot lines indicate lower, median and upper quartiles, vertical lines show inter-quartile range and open circles are values outside the inter-quartile range. Caribou Creek sub-watersheds are arranged by increasing permafrost extent (4 – 26 %), followed by the burned P6 and Boston Creek (BC) watersheds. Letters indicate streams with statistically similar solute concentration ($p > 0.05$).

Figure 3. Stream dissolved organic carbon concentration in four sub-catchments of the Caribou Poker Creeks Research Watershed (June – August, 2002 – 2005) and Boston Creek (June – August, 2005). Boxplot lines indicate lower, median and upper quartiles, vertical lines show inter-quartile range and open circles are values outside the inter-quartile range. Caribou Creek sub-watersheds are arranged by increasing permafrost extent (4 – 26 %), followed by the burned P6 and Boston Creek (BC) watersheds. Letters indicate streams with statistically similar solute concentration ($p > 0.05$).

Figure 4. Stream sulfate concentration in four sub-catchments of the Caribou Poker Creeks Research Watershed (June – August, 2002 – 2005) and Boston Creek (June – August, 2005). Boxplot lines indicate lower, median and upper quartiles, vertical lines show inter-quartile range and open circles are values outside the inter-quartile range. Caribou Creek sub-watersheds are arranged by increasing permafrost extent (4 – 26 %), followed by the burned P6 and Boston Creek (BC) watersheds. Letters indicate streams with statistically similar solute concentration ($p > 0.05$).







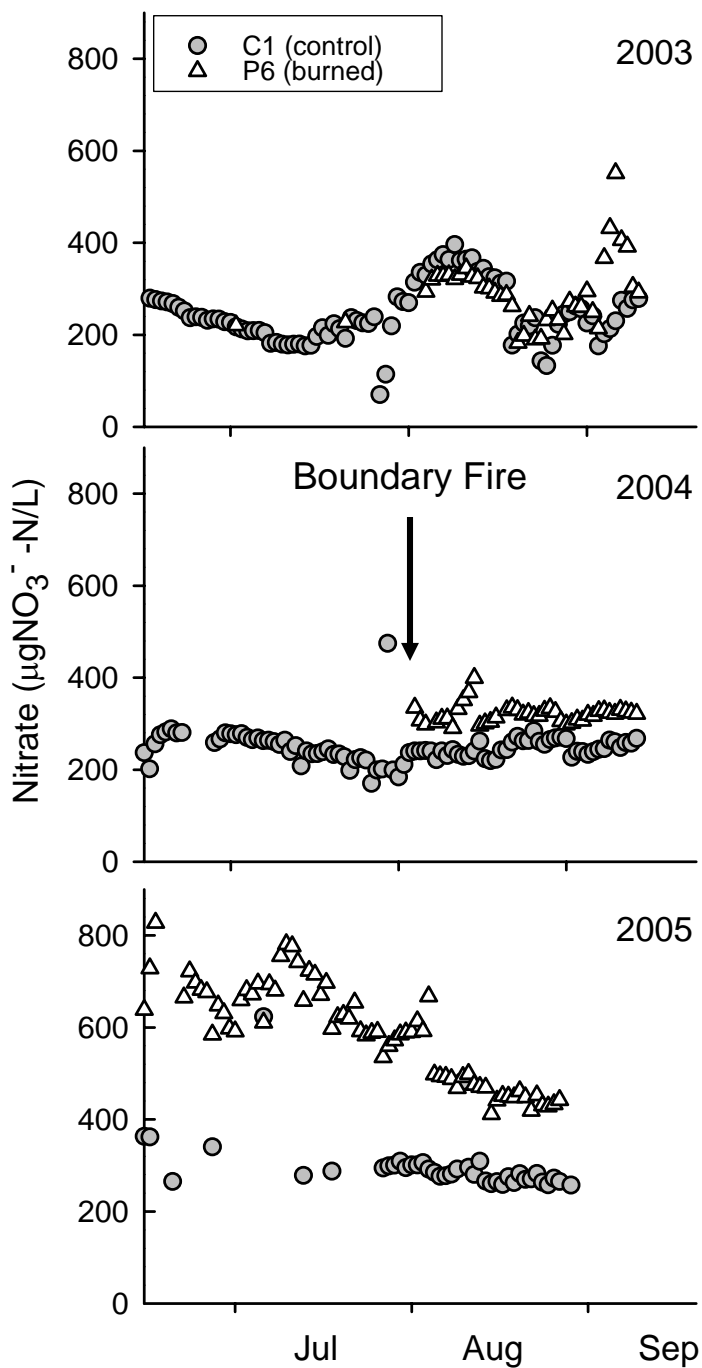


Figure 5. Pre- and post-fire nitrate concentration in burned (P6) and control (C1) watersheds of Caribou Poker Creeks Research Watershed for 2003 – 2005 (2003: $n = 37$; 2004: $n = 41$, $p < 0.05$; 2005: $n = 65$, $p < 0.05$, Wilcoxon rank-sum).

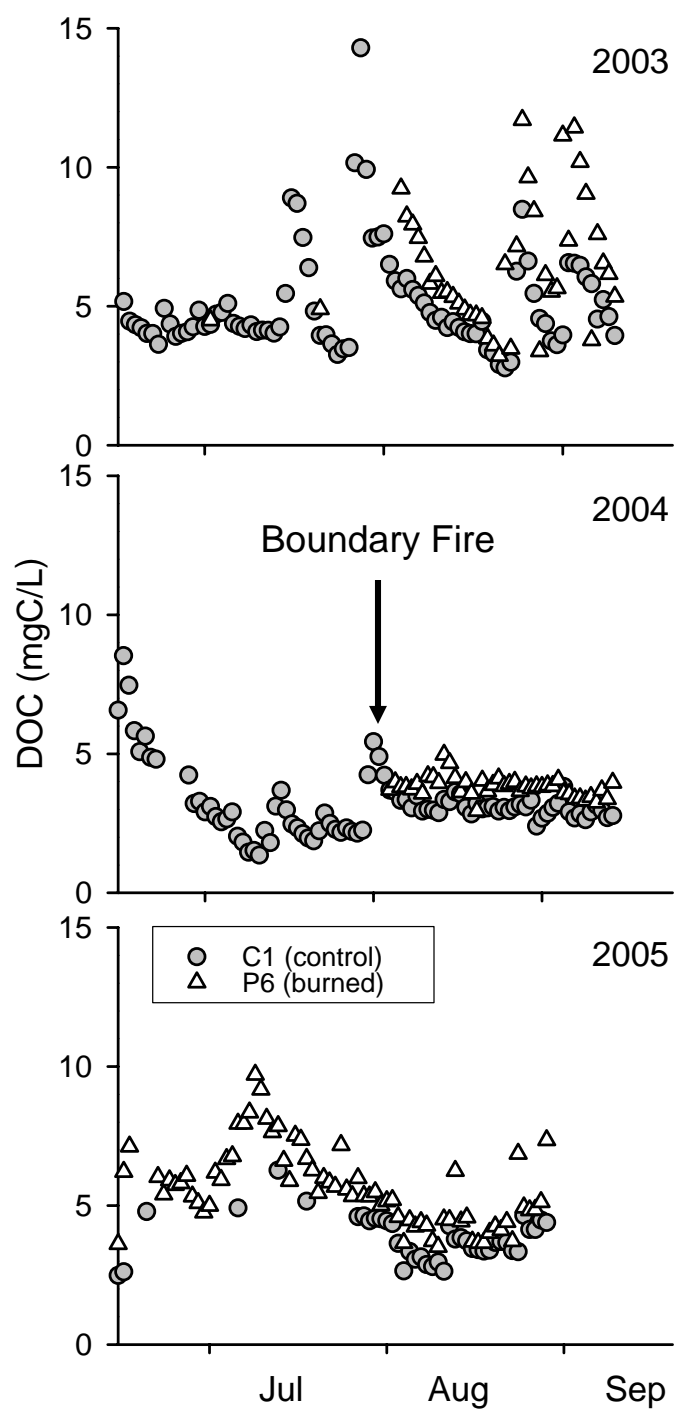


Figure 6. Pre- and post-fire dissolved organic carbon concentration in burned (P6) and control (C1) watersheds of Caribou Poker Creeks Research Watershed for 2003 – 2005 (2003: $n = 39$; 2004: $n = 41$, $p < 0.05$; 2005: $n = 65$, $p < 0.05$, Wilcoxon rank-sum).

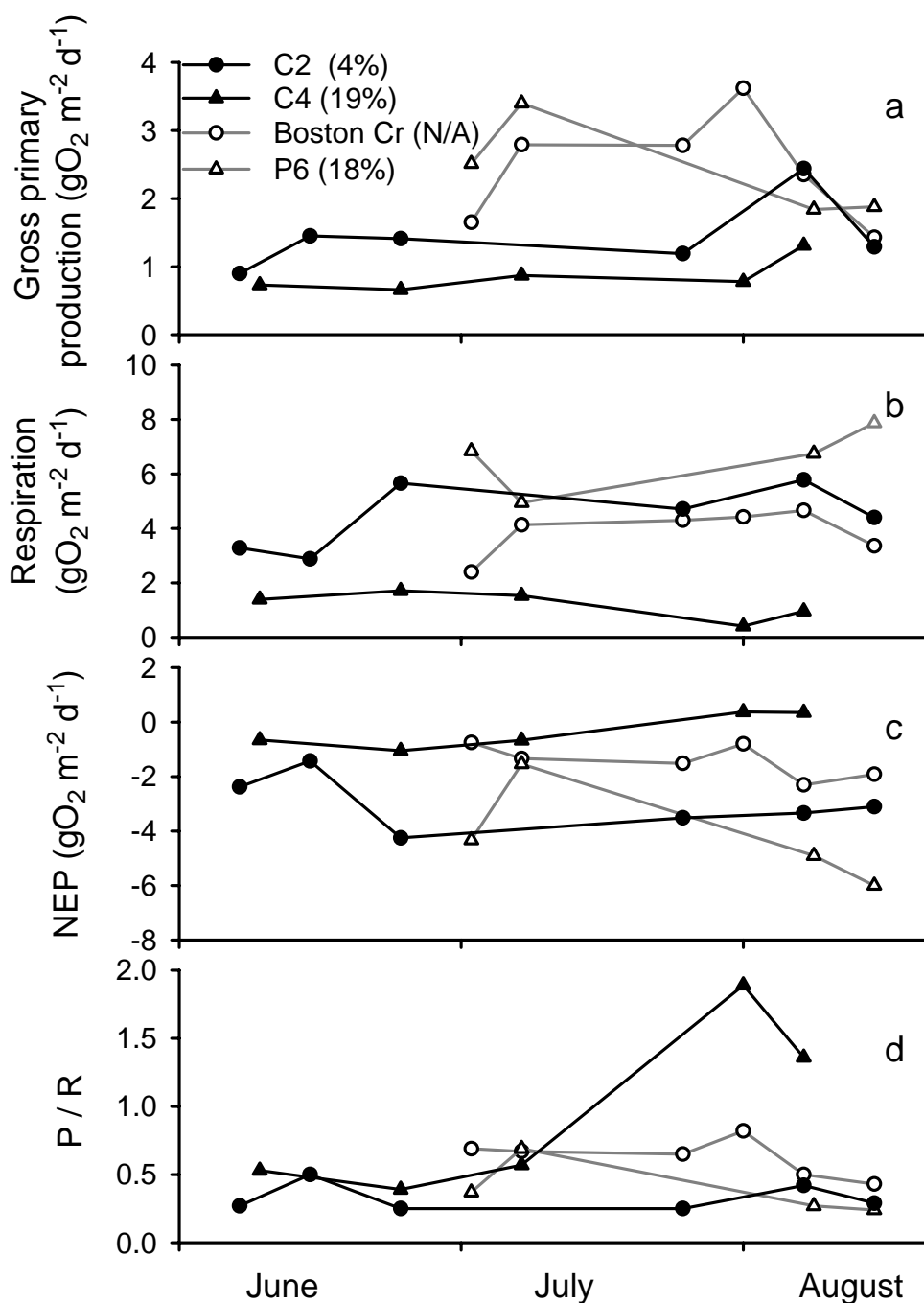


Figure 7. Summer 2005 rates of (a) gross primary production (b) ecosystem respiration, (c) net ecosystem productivity and (d) production to respiration ratio (P/R) for three sub-catchments of the Caribou Poker Creeks Research Watershed and the Boston Creek watershed. Permafrost extent in parentheses (N/A indicates data is unavailable).

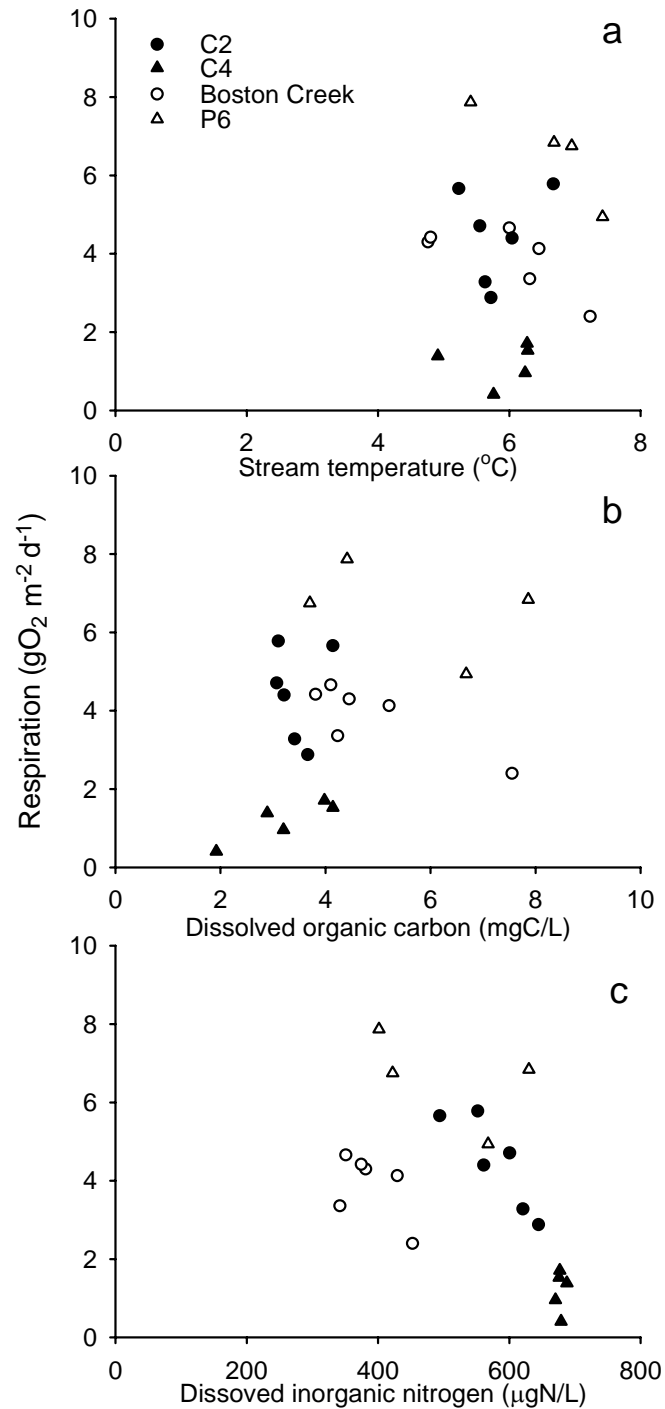


Figure 8. Relationship between stream respiration and (a) stream temperature ($p > 0.05$), (b) dissolved organic carbon concentration ($p > 0.05$) and (c) dissolved inorganic nitrogen (control, $p < 0.05$, $R^2 = 0.53$, closed circles; burned, $p > 0.05$, open circles).

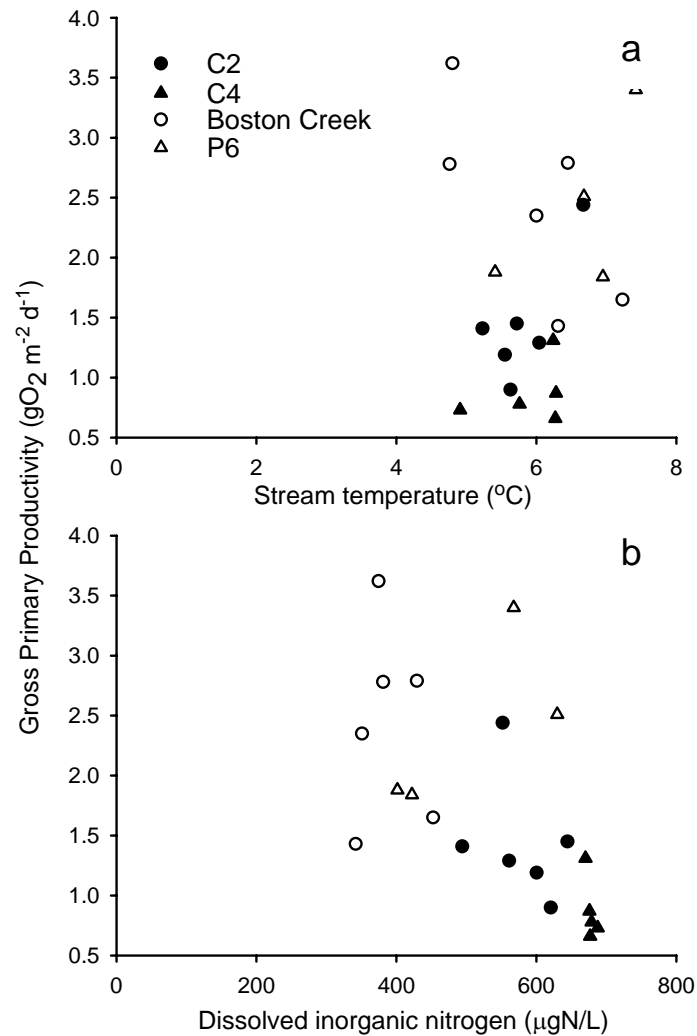


Figure 9. Relationship between gross primary productivity and stream temperature ($p < 0.05$) and dissolved inorganic nitrogen concentration (control $p < 0.05$, $R^2 = 0.49$, closed circles; burned $p > 0.05$, open triangles) in burned and control catchments of the Caribou Poker Creeks Research Watershed.

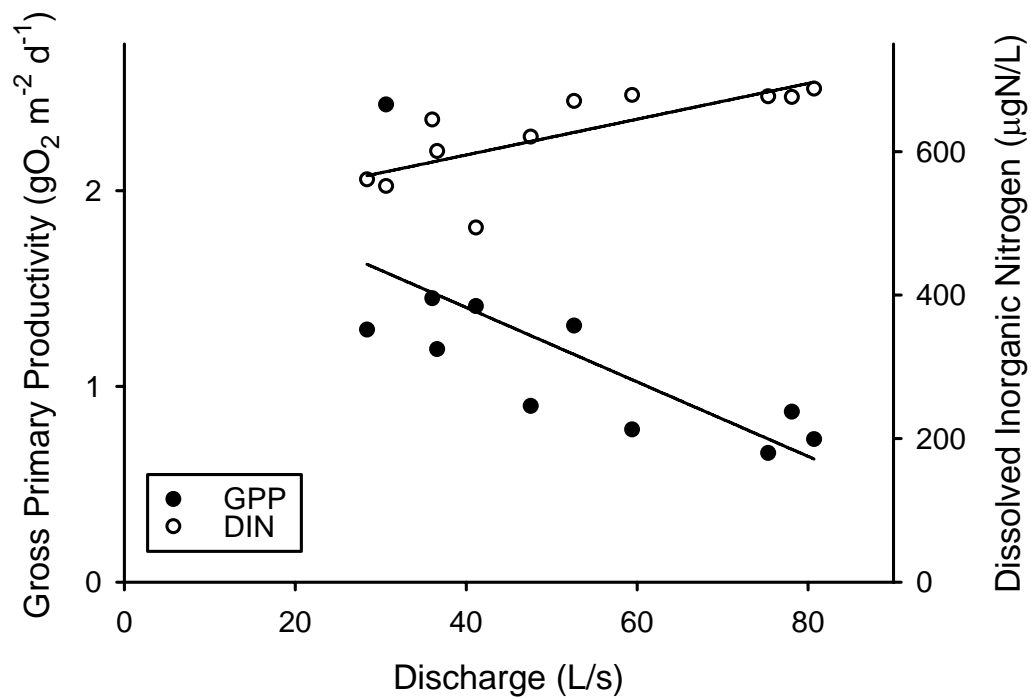


Figure 10. Relationship between discharge and gross primary productivity ($p < 0.05$, $R^2 = 0.53$, closed circles) and dissolved inorganic nitrogen ($p < 0.05$, $R^2 = 0.56$, open circles) in control streams of the Caribou Poker Creeks Research Watershed.

Table 1. Selected physical characteristics of study catchments at the Caribou Poker Creeks Research Watershed. Area burned calculated using GIS mapping (personal communication, D. Verbyla). Discharge and temperature are summer averages (C2 and C4; 2002 – 2005; P6 and Boston Creek, 2005). Riparian canopy is classified as open, closed or a combination of open and closed (mixed), based on visual assessment.

Watershed	Area (km ²)	Area Burn (%)	Permafrost (%)	Stream channel Width (m)	Temperature (°C)	Discharge (L/s)	Riparian Canopy	Dominant Vegetation
C1	6.7	0	26	N/A	4.9 (1.3)	N/A	Open	Coniferous
C2	5.2	0	4	0.65	5.4 (1.1)	31 (21)	Open	Deciduous
C4	10.0	0	19	0.94	5.7 (1.7)	75 (64)	Closed	Deciduous
P6	7.0	65	18	0.66	5.6 (1.5)	40 (13)	Mixed	Mixed
Boston Cr	20.0	85	N/A	1.18	5.7 (0.6)	109 (17)	Open	Coniferous

N/A = data unavailable

Table 2. Stream conductivity, pH and cation concentrations for the study catchments of Caribou Poker Creeks Research Watersheds and Boston Creek. Mean (\pm SD). nutrient concentrations calculated from water samples collected daily. C1, C2 and C4, May – August 2002 – 2005; P6 May – August 2002 – 2003 and 2005, and July – August 2004, Boston Creek samples collected July 2004, and June – August 2005. Statistical comparison of post-fire C1 and P6 sulfate and cation concentrations ($\alpha = 0.05$; Wilcoxon Rank Sum).

Watershed		Conductivity (μ s/cm)	pH	DON (μ g/L)	NH ₄ ⁺ (μ g/L)	Na ⁺ (mg/L)	Mg ²⁺ (mg/L)	K ⁺ (mg/L)	Ca ²⁺ (mg/L)
C2	2002	73.9 (11.5)	7.1 (0.30)	311 (95)	17 (13)	1.1 (0.3)	2.7 (0.4)	0.5 (0.3)	12.4 (2.0)
	2003	73.9 (9.3)	7.4 (0.1)	399 (94)	32 (30)	1.0 (0.3)	2.8 (0.5)	0.3 (0.1)	11.3 (1.8)
	2004	76.3 (5.9)	7.2 (0.2)	336 (370)	93 (89)	1.3 (0.5)	3.1 (0.5)	0.5 (0.2)	13.2 (0.9)
	2005	79.7 (3.1)	7.2 (0.1)	167 (107)	38 (47)	1.0 (0.7)	3.1 (0.2)	0.5 (0.1)	10.2 (0.7)
C4	2002	96.4 (16.7)	7.3 (0.3)	331 (122)	22 (21)	1.4 (0.5)	2.7 (0.5)	0.6 (0.3)	19.1 (3.9)
	2003	94.9 (15.7)	7.5 (0.1)	376 (89)	12 (14)	1.1 (0.3)	2.5 (0.4)	0.4 (0.5)	16.5 (2.6)
	2004	102.7 (7.0)	7.2 (0.2)	274 (289)	34 (26)	1.6 (0.5)	3.0 (0.5)	0.7 (0.2)	20.7 (1.1)
	2005	102.7 (7.2)	7.5 (0.2)	116 (65)	31 (31)	1.3 (0.3)	2.9 (0.3)	0.7 (0.07)	14.9 (1.3)

Table 2 Continued.

Watershed		Conductivity ($\mu\text{s}/\text{cm}$)	pH	DON ($\mu\text{g}/\text{L}$)	NH_4^+ ($\mu\text{g}/\text{L}$)	Na^+ (mg/L)	Mg^{2+} (mg/L)	K^+ (mg/L)	Ca^{2+} (mg/L)
C1	2002	47.0 (8.6)	6.7 (0.3)	272 (73)	14 (12)	1.0 (0.3)	1.3 (0.2)	0.3 (0.2)	8.2 (1.3)
	2003	49.2 (7.0)	7.0 (0.1)	301 (66)	20 (16)	1.0 (0.1)	1.4 (0.4)	0.2 (0.1)	8.4 (3.8)
	2004	58.1 (7.3)	6.9 (0.2)	293 (236)	57 (56)	1.1 (0.1)	1.5 (0.4)	0.4 (0.2)	10.4 (1.6)
	2005	55.2 (6.0)	7.0 (0.2)	204 (132)	72 (56)	1.0 (0.3)	1.6 (0.3)	0.4 (0.1)	7.6 (1.2)
P6	2002	105.3 (14.1)	7.2 (0.1)	312 (95)	14 (13)	1.3 (0.3)	2.7 (0.6)	0.4 (0.2)	22.6 (3.3)
	2003	104.2 (14.4)	7.4 (0.1)	336 (80)	10 (9)	1.1 (0.4)	2.5 (0.3)	0.2 (0.1)	18.5 (3.0)
	2004	125.3 (20.2)*	7.4 (n/a)	307 (66)*	45 (28)*	1.5 (0.1)*	2.8 (0.7)	0.6 (0.4)	26.0 (1.2)*
	2005	117.8 (15.7)*	7.4 (0.1)	189 (96)**	37 (32)**	1.3 (0.3)	3.0 (0.3)	0.7 (0.1)*	17.8 (1.6)*
Boston Cr	2004	115.0 (N/A)	N/A	N/A	N/A	N/A	N/A	N/A	N/A
	2005	84.7 (9.7)	7.4 (0.4)	221 (122)	56 (43)	1.3 (0.4)	2.8 (0.3)	0.5 (0.1)	11.2 (1.6)

*Significantly increased concentration compared with C1 ($p < 0.05$)

**Significantly decreased concentration compared with C1 ($p < 0.05$)

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Chapter 3: Conclusions

In the region of discontinuous permafrost, the extent of underlying permafrost is believed to control catchment hydrology and, hence, nutrient delivery to streams (MacLean et al. 1999, Petrone et al. 2006). Additionally, wildfire in catchments has significantly altered stream nutrient concentrations, in particular, increasing stream nitrogen, phosphorus and cation concentrations (Bisson et al. 2003, Gresswell 1999). The boreal forest of interior Alaska is experiencing a period of rapid change. Climatic warming has already caused a reduction in permafrost extent, and is also predicted to result in increased wildfire occurrence and severity. The effects of both wildfire and permafrost extent on stream nutrient concentrations and ecosystem metabolism, however, remain unclear.

This study focused on determining the influence that wildfire and permafrost have on stream nutrient dynamics and metabolism in the Caribou Poker Creeks Research Watershed (CPCRW) in interior Alaska. We predicted that increased nutrient and decreased dissolved organic carbon (DOC) delivery to streams after fire would stimulate gross primary productivity (GPP) and reduce ecosystem respiration in burned watersheds.

The presence of permafrost in catchments did seem to influence catchment hydrology and stream nutrient concentrations; however, the relationship did not vary predictably with percent area of underlying permafrost. Two catchments (low and medium-permafrost extent) had high stream nitrate and low DOC concentration, which contrasted with catchments (medium and high-permafrost) with higher DOC and lower nitrate concentration. Despite significant differences in patterns of solute chemistry, the

relationship between permafrost coverage and solute concentration was not linear. Other factors such as catchment contributing area, location of underlying permafrost, geology, and catchment vegetation characteristics are likely to be contributing to the observed patterns in solute chemistry among watersheds.

The stream draining the burned watershed had higher stream nitrate, sulfate, and cation concentrations, and lower DOC concentration, than the control stream. Average summer GPP was higher in streams draining burned watersheds; however, this does not appear to result from the increased concentrations of nutrients we noted after fire. Instead, we suggest higher GPP is related to an undetected but potentially important increase in soluble reactive phosphorus. Stream ecosystem respiration was also higher in burned than unburned watersheds, but fire is unlikely to have been the cause. Rather, ecosystem respiration in CPCRW streams may be driven by pre-existing differences in stream physical or chemical characteristics.