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The forest fire induced impacts on phosphorus, nitrogen and chlorophyll *a* concentrations in boreal sub-arctic lakes of northern Alberta.

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Abstract

Forest fire has long been recognized as an important disturbance structuring Canada's boreal forest. The effects of fire transcends forest succession altering biogeochemistry of soils, hydrology and thus the flux of solutes from landscape to streams and lakes. To better understand the role of forest fire in altering boreal streams and lakes, the Sustainable Forest Management Network (SFMN) began an investigation of fire effects in aquatic systems under Legacy 1 (formerly Ecological Basis for Sustainability). Fire effects were also of interest to aboriginal peoples in northern Alberta due to their traditional use of fire as a management tool and because of recent large fires on their traditional lands. In this paper, we present data from the Caribou Mountains, a boreal sub-arctic plateau in the discontinuous permafrost region of northern Alberta. 129 000 ha of this plateau burned during the summer of 1995, presenting a unique opportunity to study the effects of large-scale forest fire on aquatic systems.

Two years following forest fire, biogeochemistry of 11 headwater lakes in burnt peatland-conifer catchments was altered relative to 17 lakes in unburnt catchments. Cation exchange in burnt catchments resulted in flux of protons and 1.3-fold lower mean pH and 2-fold reduction in total alkalinity. Lakes in burnt catchments had 3-fold higher mean total phosphorus (TP), 1.5-fold higher total nitrogen (TN) and 2.6-fold higher dissolved organic carbon (DOC) concentrations. Secchi depths and light extinction averaged 52 and 160% of the reference lake mean. TP explained 86% of the variance in reference lake chlorophyll (CHL) but was not related to CHL in burnt catchments. Residual analysis and nutrient stimulation experiments suggested algae in burn-impacted lakes were light- and nitrogen-limited.

General models that could explain the dependence of water chemistry on watershed characteristics were explored. Over a range of catchment-to-lake area ratios, Caribou Mountain lake chemistry was related to a fraction of total catchment areas. Catchments were subsequently reduced in size to effective areas (EA) and a method for predicting the mean size of an EA was contrived using a mass-balance for ^{18}O and sodium. EA divided by lake volume explained 20 to 69% of variance in nutrient concentrations among lakes, performing better than other physical parameters. With the addition of five lakes burnt between 1961 and 1985, time-since-disturbance and %disturbance, combined, explained 58% of the variance in TP.

Introduction

Interest in fire as a natural disturbance impacting catchment biogeochemistry and surface water quality is rekindling. The growing interest in fire centers on better management practices for forested lands, however, studies on the affects of fire suggest impacts are regionally or fire specific. One common theme is that the magnitude of ion flux from catchments is dependent on fire severity because of modifications to cation exchange and biochemical reactions in the organic soil layer (Grier 1975; Stark 1977; Schindler et al. 1980). In fires where a third or less of the organic soil layer was burnt, no changes in soil water and stream water carbon, nitrogen, phosphorus, their inorganic fractions or major cations were reported (*e.g.* Richter et al. 1982). In more severe fires, where bedrock has been exposed, post fire increases in nutrient flux are usually short lived (<5 y) and changes in surface water concentrations do not exceed inter-annual variation (McColl & Grigal 1975; Bayley et al. 1992; Minshall et al. 1997). These studies have occurred primarily in granitic regions with thin organic soils compared to permafrost - peatland dominated systems such as those in northern Alberta.

Peatlands have been studied in granitic regions. For example, a comparison of burnt and unburnt portions of a *Sphagnum fallax / Picea mariana* mire demonstrated no detectable long - term (8 y) impact of fire on water chemistry (Vitt & Bayley 1983). In a review of circumpolar studies MacLean et al. (1983) reported fire rarely burnt the entire organic layer and changes in soil water chemistry did not result in nutrient flux from peat-derived soils. The results from previous studies give little indication of potential impacts from forest fire in catchments with deep peat soils (0.5 m or more), that are underlain by glacial till and contain permafrost or seasonal frost lasting a majority of the summer. From a management perspective, lack of data for northern Alberta makes it difficult to assess impacts from forest management practices designed to mimic fire.

A unique opportunity to examine extensive damage to peatlands was provided when 129 000 ha of the Caribou Mountains, a Sub-arctic plateau, were razed by fire in 1995. In this single event, one - third of the Plateau was consumed, equaling 50% of the mean annual area burnt in Alberta between 1994 and 1998 (Alberta Environmental Protection). Unlike other studies, impacts were expected because of high fire severity, large proportion of catchments burnt (between 60 and 100%), hydrology dominated by flow through the peatlands, and relatively low contributions from aerial deposition to overall lake nutrient budgets (Shaw et al. 1989). We hypothesize that the 1995 fire in the Caribou Mountains impacted lake water chemistry by affecting cation exchange in peat and mineralization of nutrients to produce: a) increased base cation and nutrient concentrations in surface waters; b) corresponding increases in algal biomass; and c) increased suspended organic and inorganic seston. We also looked for indications of long-term (decades) impacts from fire as well as relationships between catchment caracteristics and lake water chemistry.

Methods

Site description

The Caribou Mountains (59°N 115°W) are an erosional remnant of the Keewatin Glacier forming a large plateau 500 m above the Peace River valley. Peatlands cover 56% of the Plateau (Vitt et al. 1996) and are predominately underlain by poorly drained Cryosolic and Brunisolic soils (Strong & Leggat 1992). Solis overlay shale, feldspathic sandstone, and siltstone of deltaic and marine origin (Alberta Geologic Survey). Vegetation is dominated by open forest of black spruce (*Picea marianna*) and an understory of mixed *Sphagnum* spp., feathermoss, brown moss and lichen. Aspen (*Populus tremuloides*) is concentrated in upland areas. Mean percent uplands per catchment is less than 30% but is 83% in one burnt catchment. Permafrost is between 0.5 and 0.75 m beneath the organic soil surface within continental bogs (Strong & Leggat 1992), but is not present in collapse scars (Vitt et al. 1994). Mean peat depths are estimated between 0.75 and 1 m (Zoltai 1993). Mean May through August temperatures are approximately 10° C with less than 800 growing degree days and between 400 and 450 mm of annual precipitation (Strong & Leggat 1992).

Eleven headwater lakes were selected from catchments where between 50 and 100% of the tree cover was killed by fire (mean 83%, median 90%, s.d. 15%). These lakes plus 17 headwater lakes in unburnt reference catchments and five headwater lakes in catchments burnt between 1961 and 1985 were sampled monthly after ice out in late June, July and August/early September (Fig. 1). Lakes from previously burnt catchments are used only in time-since-disturbance and %disturbance analyses. There was a large range in surface area (2.6-1428 ha) and mean depth (0.2 – 11.6 m) among study lakes but ranges in these parameters were similar among burnt and reference treatments (Table 1). Three reference lakes contained volumes of water an order of magnitude higher than the largest burnt lake. However, burnt and reference lakes had a similar range in drainage ratios, catchment vegetation patterns and all were polymictic. In the fire, understory herbaceous, sedge, and lichen cover were incinerated but damage to moss and peat was varied. Burning was mostly limited to 0 to 20 cm of peat that is on average greater than 0.75-m deep. In some locations burning continued within deeper peat until the following summer. Late August testing of some catchments revealed no permafrost in the peat layer (down to mineral soil) of burnt areas, whereas it was common in unburnt areas.

Groundcover in each catchment was classified as upland-aspen, peatland or water. Peatlands were subdivided into veneer bog, peat plateau, poor fen and rich fen based on vegetation and slope characteristics identified from 1:20000 aerial photos (Halsey *et al.* 1997). Percentage of fire disturbance per drainage basin was estimated from aerial photos (1996) and flight surveys. Bathymetric maps were constructed from depth measurements along 5 to 15 transects on each lake. Depth measurements were recorded by echo sounding at equal intervals along transects and used to interpolate depth contours in ArcInfo. Catchment slope was calculated by computing elevation gain divided by linear distance to lake shore (CS1: D'Arcy & Carignan 1997) with 10-30 transects selected by picking topographic high points and intermediate saddles around the watershed.

At the deepest site in each lake, temperature and dissolved oxygen (YSI 50B), light penetration (Li-Cor LI-185, LI-192SB flat sensors) profiles were recorded at 0.5-m intervals. Transparency was recorded with a 20-cm Secchi disc. A 12-L composite water sample was collected with multiple hauls of a PVC tube to the depth of 1% light penetration or 1 m above the lake bottom. Three opaque 2-L acid-washed polyethylene bottles were rinsed and filled with the composite sample. The pH of the composite sample was measured with a Hanna HI9025 calibrated before each use. Water samples were also collected from three streams and four fen sites throughout the summer. Samples were collected from a deep groundwater well throughout the summer of 1998. One stream, MS1, was gauged for discharge calculations. An automated sampler collected storm-event water in MS2.

Triplicate 25-mL aliquots were pipetted into 30-mL glass tubes used directly in the digestion process and nutrient analysis. Unfiltered water was used for total nitrogen (TN), total phosphorus (TP) and ammonium (NH_4^+-N) analyses. Filtrate (Millipore HA, 45 µm) was used for dissolved nitrogen (DN), dissolved phosphorus (DP) and nitrate-nitrite (NO_3^--N) analyses. Nitrogen samples were preserved with 10 µL of 40% H_2SO_4 for storage (1-2 wk), then neutralized with equivalent NaOH prior to digestion in the laboratory. TP and DP were analyzed spectroscopically (5-cm cell) from persulfate-oxidized samples by molybdate blue absorption (Prepas & Rigler 1982). TN and DN concentrations were determined by second derivative spectroscopic analysis of persulfate oxidized samples (Crumpton *et al.* 1992). Soluble reactive phosphorus (SRP) concentrations were measured by molybdate blue absorption from membrane filtrate within 24 h of collection. Ammonium and nitrate samples were frozen in 125-mL polyethylene bottles for analysis by indophenol blue (NH_4^+-N) and cadmium reduction (NO_3^--N) with automated colorimetry (Technicon methods 100-700 W/B and 155-71W). Nitrate and ammonium concentrations were combined into a single dissolved inorganic nitrogen value (DIN). NH_4^+-N averaged 74% of DIN, ranging from 53 to 94% (s.d. 13%).

Total Suspended Solids (TSS), Nonvolatile suspended solids (NVSS), and volatile suspended solids (VSS) were collected on pre-ashed GF/F filters (mean particle retention 0.7 µm) and analyzed in duplicate after APHA (1993). Alkalinity was determined by the Hach phenolpthaline/ bromocresol method (APHA 1993). Filtrate from GF/F filters for cations (Ca^{2+}, Mg^{2+}, Na^+, K^+), dissolved organic and inorganic carbon (DOC/DIC), and anions (Cl^-, SO_4^{2-}) analysis were stored in 60-mL polyethylene bottles. Cation samples were preserved with 40% H_2SO_4 , DOC/DIC and anion samples were refrigerated. Cations were analyzed by atomic absorption flame spectroscopy (Perkin-Elmer AS90, AA3300), DOC/DIC by high temperature catalysis (Shimadzu TOC-5000), and SO_4^{2-} and Cl^- concentrations by chromatography (Dionex 2000i/SP).

Total suspended chlorophyll was collected on Gelman A/H filters (mean particle retention 1.2 µm) in duplicate from each of the three composite sample bottles. Filters were subsequently dessicated with silica-gel and frozen. Total chlorophyll *a* (CHL) concentrations were determined with fluorometric methods (Sartory & Grobbelaar 1986).

To test nutrient limitation of phytoplankton we had expected to select one burnt and one unburnt headwater lake. One of the study lakes (C24) was chosen to represent burnt lakes, however, a non-study lake, Margaret Lake was selected to represent reference conditions due to flight costs. Our base camp was situated at Margaret Lake, which had approximately a third of its catchment burnt, but was non-headwater and therefore contained a phosphorus concentration representative of reference lakes (16 µg•L-1) and TN:TP (38) at the high end for reference lakes. Each experiment consisted of 10-L microcosms augmented with N, P, or N+P, plus an untreated control, replicated in three sets and suspended at half the Secchi disk depth for 4 d (Knowlton & Jones 1996). Response was determined by dividing final CHL by initial CHL and then comparing responses among treatments with replicate means (ANOVA, Bonferoni-Dunn comparison). The Nutrient Stimulation Experiment (NSE) distinguishes three potential nutrient outcomes: limitation by N or P; reciprocal limitation first by one nutrient then the other; and concurrent limitation by both N and P. Each response was determined by comparison to response in control containers. Response to favorable light was inferred from a positive response in control containers compared to initial CHL and final lake CHL.

Measurement of the stable isotope composition of lake water (δ18O and δ2H) was used as a comparative index of lakewater budgets. Based on a simple steady-state isotope-mass-balance model (Gibson et al. 1993) the water residence time t (years) for each lake was estimated as:

$$t = \frac{V}{E}\left(\frac{h}{1-h}\right)\left(\frac{d_L - d_P}{d* - d_L}\right) \qquad \text{eq. 1}$$

where V is the average lake depth (mm) determined by bathymetry, and E and h are annual small lake evaporation (mm/year) and humidity, respectively, interpolated from the Environment Canada climate network. Also, δL, δP, δ^* are the isotopic compositions of lake water (measured 3 times/yr), precipitation (interpolated from GNIP (1996) data), and limiting enrichment rate. The latter parameter reflects the atmospheric limitation on enrichment which is determined by climatic factors (Gibson et al. 1993).

When transport of solutes is attenuated by soils of low hydraulic conductivity, modeling catchment contributions of solutes can be achieved by reducing the size of the drainage basin to an effective area (Sorrano et al. 1996). The effective area (EA) represents the land area that contributes solutes to surface waters which can contract and expand with dry and wet periods. Mean effective areas were constructed for each lake by calculating the area required to account for lake concentrations of sodium ions. The formula:

$$EA = Q_o\,[Na^+] / F \qquad \text{eq. 2}$$

expresses EA as a function of lake sodium concentration, flux rate (F) and discharge (Q). The equation assumes Na^+ is a conservative element at equilibrium in the lake ($Q_I C_I = Q_O C_L$) and its concentration is directly dependent on loading from the catchment ($Q_I C_I = EA*F$). A flux rate of 0.189 $meq \cdot L^{-1} \cdot m^{-2}$ [Na^+] was estimated from reported values for an arctic basin (Buttle & Fraser 1992) and from north-central Alberta (Shaw 1990). The EA calculation is dependant on the flux rate of Na^+ (eq. 2) which we held constant for both burnt and reference lakes. Cation flux rates measured from soils after forest fire are typically elevated for Ca^{2+}, Mg^{2+}, and K^+ but changes in flux of Na^+ are reported as insignificant (Stark 1977; Feller & Kimmins 1984). Discharge was calculated from ^{18}O and deuterium signatures. EA was not calculated for two lakes because isotope signatures returned ambiguous retention times. This approach assumes that the land area contributing Na^+ is the same or proportional to the area contributing all other elements.

Summer mean (June-August) values are used in all analyses. Data are presented in Box & Whisker plots where the box contains 25% to 75% of the data and whiskers denote 90% of the data. The line within each box represents a median value. pH values were converted to [H^+] prior to averaging and analysis (Aquatic Chemistry, Stumm & Morgan 1970) and converted back to pH for reporting. Normally distributed data were analyzed by unpaired two group *t*-test, Pearson's correlation and univariate least-squares regression. Non-normal data were either tested by non-parametric tests (*Mann-Whitney U*) or were $\log_{10}$-transformed for regression analysis (Snedecor & Cochran 1989).

Results

Forest fire and surface water chemistry

Lake water in the Caribou Mountains had dilute total base cation concentrations (Ca^{2+}, Mg^{2+}, Na^+, K^+), all were below the world average for freshwater (2.33 $meq \cdot L^{-1}$). The mean total base cation concentration for reference lakes was 0.776 $meq \cdot L^{-1}$. There were no detectable differences in mean total base cation concentrations between burnt and reference lakes (P = 0.25). As a percentage of total cations, Ca^{2+} was lower and K^+ was higher in burnt lakes compared to reference lakes (Fig. 2).

Reference lake water was moderately alkaline (median pH = 7.8), while fire caused an influx of protons (median pH = 6.9, P = 0.02) and a lower mean total alkalinity of 15.5 $mg \cdot L^{-1}$ $CaCO_3$ (P= 0.01) compared to 33.2 $mg \cdot L^{-1}$ $CaCO_3$ in reference lake water (Fig. 3). Sulfate concentrations were not different between burnt and reference lakes (Mann-Whitney P = 0.09) despite higher median concentrations (median 2.1 and 5.4 $mg \cdot L^{-1}$, respectively). Fire did not effect chloride concentrations, which averaged 0.17 and 0.14 $mg \cdot L^{-1}$, respectively, for burnt and reference lake water (P = 0.82). Total anions (SO_4^{2-} , Cl^- , HCO_3^-, NO_3^-) balanced total cations in reference lake water (0.790 $meq \cdot L^{-1}$). All lakes in burnt catchments had a deficit in total anions averaging 26% below total base cations, suggesting a large pool of organic acids existed in burnt lakes. Most *Sphagnum*-derived acids contain –COOH groups which act as anions in solution while contributing protons (Clymo 1984). Indeed, anion deficits were linearly correlated with DOC in burnt lakes ($r^2 = 0.79$, $P << 0.01$, $n = 11$). Fire thus appeared to increase lake acidity by causing elevated flux of organic acids.

Lake water in burnt catchments had elevated concentrations of TP, TN and their fractions. In reference lakes, mean values of TP, DP and SRP were 33, 14 and 5 $\mu g \bullet L^{-1}$, respectively (Fig. 4). Lake water in burnt catchments contained 2.8-fold higher TP, 3.6-fold higher DP and 5.4-fold higher SRP than in reference lakes ($P < 0.05$). In reference lakes, mean TN, DN, NO_3^--N, and NH_4^+-N were 640, 480, 4 and 14 $\mu g \bullet L^{-1}$, respectively (Fig. 5). Burn-impacted lake water contained 1.3-fold higher TN ($P = 0.01$) and 1.7-fold higher DN ($P = 0.003$) compared to reference lake water. NO_3^--N and NH_4^+-N concentrations were $\log_{10}$ -normally distributed. Burn-impacted lake water contained higher concentrations of both inorganic nitrogen fractions (Fig. 6; $P = 0.01$). Median NO_3^--N and NH_4^+-N values were 6 and 16 $\mu g \bullet L^{-1}$, respectively, for burnt lakes and 2 and 12 $\mu g \bullet L^{-1}$, respectively, for reference lakes. Mean TP and DP in the two reference streams (MS1 & MS2) were 48 and 28 $\mu g \bullet L^{-1}$ while the burn-impacted stream (MS3) contained 5.4-fold higher TP and half the DP. Mean TN and DN concentrations for water in MS1 and MS2 were 710 and 650 $\mu g \bullet L^{-1}$, respectively, while MS3 contained 2.2-fold higher TN and 1.2-fold higher DN concentrations. Increased export of all measured forms of phosphorus and nitrogen from burnt catchments relative to reference watersheds was inferred.

Caribou Mountain lake water contains high concentrations of DOC which were further exacerbated by fire. The mean DOC concentration for reference lake water was 16 $mg \cdot L^{-1}$. Mean DOC in burn-impacted lake water (26 $mg \cdot L^{-1}$) was 1.6-fold higher ($P << 0.01$) than in reference lakes. DIC averaged 8 $mg \cdot L^{-1}$ for both reference and burnt lakes. In MS1 and MS2, DOC concentrations averaged 35 mg·L^{-1} and DIC averaged 8 mg·L^{-1}. In MS3, DOC averaged 28 mg·L^{-1} and DIC averaged 14 mg·L^{-1}. High DOC in burn-impacted lake water may be a result of conversion of elevated DIC exported from catchments into DOC in fens prior to entering lakes and within the lakes themselves.

All Caribou Mountain lakes contained highly coloured waters, especially within burnt catchments. Mean colour was 143 mg·L^{-1} Pt and ranged from 26 - 388 mg·L^{-1} Pt in reference lake waters. These values are comparable to lake water in the boreal mixedwood ecoregion to the south where colour ranges from 8 - 388 mg·L^{-1} Pt (Prepas unpublished data). In keeping with increased flux of DOC from peat after forest fire, mean colour in burn-impacted lake waters (360 mg/L Pt) was 2.5-fold higher (P<<0.01) than in reference lakes. Colour was related to DOC in all lake waters (r^2 = 0.90, P<<0.01). Variance in light penetration was most closely associated with colour of lake water; Secchi disk depths were negatively related to colour (r^2 = 0.74; P<< 0.01). Light extinction coefficients were positively related to colour (r^2 = 0.55; P<< 0.01). As a result, mean Secchi depths in burn-impacted lakes (0.73 m) were 52% of those in reference lakes (P <<0.01), while the mean light extinction coefficient (0.811 m^{-1}) was1.6-fold higher in burn-impacted lakes (P <<0.01) compared to reference lakes. Elevated DOC and colour in burn-impacted lakes reduced transparency and likely enhanced the potential for light limitation of algal growth after fire.

Fire affected inorganic suspended solids concentrations. Mean nonvolatile suspended solids (NVSS) in reference lakes was 1.1 mg·L^{-1}, whereas burnt lakes contained 2.6-fold higher NVSS (Mann-Whitney, P = 0.01). Volatile suspended solids (VSS) were not different between reference and burnt lakes, averaging 4.6 and 5.9 mg·L^{-1} respectively (P =0.45). Neither Secchi depth nor light extinction were related to either NVSS or VSS. Increased transport of inorganic particles from burnt catchments is inferred.

Water residence times in reference lakes averaged 322 d and ranged from 50 d to 2.4 y. Mean residence time for burn-impacted lakes (61 d) was lower (Mann-Whitney $P< 0.01$) ranging from 26 to 170 d. Despite the lower residence time, there were no detectable differences in water yield from burnt and reference catchments (Mann-Whitney $P = 0.80$), due to the high standard error of these estimates (Table 2). For all lakes combined, water yield averaged 100 mm/y, or 25% of measured precipitation and estimated snow water equivalent in 1997.

Fire did not appear to affect lakewater CHL concentrations. Among reference lakes, mean CHL was 11 $\mu g \cdot L^{-1}$ and $\log_{10}$-transformed values were strongly related to TP (Fig. 7, $r^2 = 0.86$, $P<<0.01$). Despite the much higher phosphorus concentrations observed in burn-impacted lake waters, CHL was not elevated, averaging 13.4 $\mu g \cdot L^{-1}$. There was no detectable relationship between CHL and TP concentrations for burn-impacted lakes ($r^2 = 0.08$, $P= 0.4$). We predicted CHL concentrations for burn-impacted lakes with the CHL-TP model for reference lakes. The residuals between predicted and observed values were positively related to Secchi depth ($r^2 = 0.51$, $P <0.01$) and to TN:TP ratio ($r^2 = 0.67$, $P <<0.01$). A one unit decrease in TN:TP in burn-impacted lake water resulted in a 1.21 decrease in the CHL:TP ratio. The lack of response in CHL to elevated phosphorus in burn-impacted lake waters is attributed to light and nitrogen limitation.

Nutrient stimulation experiments (NSE) suggested algal biomass in lake water from C24 (burnt catchment) was limited by nitrogen and light (Fig. 8). The NSE in Margaret Lake demonstrated phosphorus limitation of algal biomass and no indication of light limitation. Margaret Lake had a TP concentration (15 $\mu g \bullet L^{-1}$) representative of the mean for reference lake waters while C24 had a TP concentration (70 $\mu g \bullet L^{-1}$) slightly lower than the burn-impacted mean but similar to the median. Margaret Lake water had a high TN:TP ratio (38) compared to reference lakes while the TN:TP ratio in C24 (8) was representative of burn-impacted lakes (Table 2). Margaret Lake and C24 had similar mean Secchi disk depths (1.3 and 1.2 m, respectively) however, only C24 responded to light over the incubation period.

Modeling land-water interactions

There were no detectable relationships between nutrient concentrations of lake waters and drainage ratio (P >0.3). Three reference and three burn-impacted lakes had drainage ratios larger than 25. When these lakes were removed, colour and DOC were strongly related to drainage ratio in the remaining 22 lakes (r^2 = 0.40 and 0.38 respectively, P <0.01). The observed linear relationships for colour and DOC to drainage ratio only in those lakes with small drainage ratios (<25) suggests that contributions to lakes from the landscape are restricted to a fraction of catchment area.

Transport of solutes is likely to be attenuated by both low relief and flow through peatlands. Watershed slope ranged from 1.6 to 6.1% with a mean of 3.5% for all study lakes. Under conditions where attenuation occurs, catchment loading models can employ an effective area approach (Soranno *et al.* 1995) to represent restricted contributing areas. Effective areas were calculated for 26 of the 28 reference and 1995 burn-impacted lakes. Effective areas were a mean 40% of total catchment areas for reference lakes and 16% for burn-impacted lakes; however, the median was 8% for both groups (Mann-Whitney P = 0.77). Among lake variation in mean TP, DP, SRP, DN and DOC was linked to landscape using EA normalized by lake volume (EA/V). Linear regression with EA/V explained 69% of the variance in $\log_{10}$ TP, 63% of variance in $\log_{10}$ DP, 52% of variance in $\log_{10}$ SRP, 42% of variance in $\log_{10}$ DN and 62% of variance in DOC (P <<0.01). EA/V explained 26% of the variance in $\log_{10}$ DIN (P = 0.01) and 20% in $\log_{10}$ TN (P = 0.03). Drainage ratio and watershed area divided by lake volume explained less than 10% of variation in nutrient concentrations (P >

0.4). Effective area appears to better represent transport of materials from peatland catchments because transport is reduced to a sub-area within a catchment.

Percent disturbance and time since disturbance

Both phosphorus and nitrogen concentrations in lakes were correlated with percent disturbance and time-since-disturbance (Table 3). Percent disturbance demonstrated a positive-linear relationship with TP and DP. Time-since-disturbance data were not evenly distributed over the 44 y for which we have data, however, TP and DP appeared to decline with the natural logarithm of time-since-disturbance. We combined both disturbance measures into a single disturbance index (DI) based on the linear and hypothesized exponential relationships:

$$DI = \%Disturbance * e^{-Kt} \qquad \text{eq. 3}$$

where t = time since disturbance and K = decay constant. The value for K was predicted from the relationship between $\log_{10}$ DP and TP with the natural logarithm of time-since-disturbance where K = 0.693/T and T = half-life. The half-life (T) was 7 y for DP and 10 y for TP. The disturbance index explained 58% (*P*=<<0.01) of the variance in TP and DP among all 33 lakes (Table 4).

Discussion

The movement of groundwater to lakes and streams in the discontinuous permafrost region of boreal Alberta likely occurs by: (i) flow in contact with mineral soils or (ii) flow through extensive organic peat. Groundwater in contact with mineral soils enters through stream beds and fens as they appear to be the only places without permafrost. Inputs from the peat layer would be restricted to riparian areas for sheetflow and vadose water by low hydraulic conductivity of peat (less than $0.1\ cm \cdot hr^{-1}$) or larger distances through macropores (e.g. Hill 1993). Fens can have greater influence on surface water chemistry than bogs by virtue of their high discharge (Halsey et al. 1996). Base cation concentrations for Caribou Mountain fens were characteristic of bog/fen complexes (Halsey et al. 1996) and poor fens (Vitt & Chee 1990) in northern Alberta and were 20 to 60% of the mean for Caribou lakes. Groundwater from mineral soils of the Caribou Mountains had base cation concentrations 20-fold higher than found in lakes. The low discharge expected through peat directly, the similarity between fen and lake chemistry and the high cation concentrations of mineral soil water suggests groundwater contributions to lakes were restricted to waters from the organic layer flowing through fens.

Determining hydrologic flowpaths in flat peatlands similar to the Caribou Mountains is difficult. Deuterium and ^{18}O proved an effective alternative, predicting water yields of 100 mm, similar to the 85 mm derived from discharge measurements in MS2. We had predicted increased water yields from burnt catchments due to reduced evapotranspiration (Johnson & Needham 1966; Tiedmann *et al.* 1978). Similarity in water yield between burnt and reference watersheds could be a result of increased volume for water retention as permafrost melted, however, this is unlikely as melting frost usually results in subsidence of peat (Vitt et al. 1994). An alternate explanation follows observations by Rouse (1976) who demonstrated that microclimate changes in sub-arctic catchments should lead to increased evaporation and decreased snow accumulation following fire. Higher deuterium enrichment rates in burnt areas would cause an underestimation of water yield. However, decreased snow accumulation could likewise account for no observed increase in water yield. The processes of isotope fractionation require further study as isotopes may offer the only feasible method of acquiring hydrologic data in poorly drained peatland areas such as the Caribou Mountains.

The flat terrain and poor drainage of the Caribou Mountains would attenuate solute transport so that only areas close to water bodies deliver solutes. This is unlike the Canadian Shield where bedrock and thin soils allow the prediction of solute concentrations in lake waters from mean catchment export rates (Dillon & Kirchner 1975, Chapra *et al.* 1978). The influence of transformation on transport of nitrogen and phosphorus as water flows from uplands through valley bottoms can be incorporated into models with catchment slope as an indicator of inundated area (D' Arcy & Carignan 1997). In areas where gradient is low, an alternate approach is to truncate the catchment to include only those transforming areas that give surface water its chemical signature. In the Caribou Mountains this area (EA) was inferred from lake Na^+ concentrations, assuming transformations of nutrients are also limited to these same areas. EA was the best physical correlate for nutrient concentrations in Caribou lakes (Table 2). These encouraging results suggest improvements in modeling land-water linkages are possible by investigating solute transport and transformation as water flows through vegetation categories that comprise an EA. In northern Alberta we hypothesize that fens and inundated bogs, permanently or intermittently linked to surface water bodies, comprise the EA and that these areas produce water relatively consistent in their ion chemistries.

Time-since-disturbance and percent disturbance were important factors in nutrient enrichment among lakes explaining 58% of the variance in TP and DP concentrations (Table 4). An exponential decay model for time-since-disturbance impacts was used because recovery should initially be rapid as new growth and microbial communities re-establish themselves. Recovery rates should be reduced through time as the pulse of readily available organic matter from the fire is used by microbial communities both within the lake and catchment or are flushed from the lake and lost to sediments. In our dataset, a linear model for time-since-disturbance explained 65% of the variance in TP and DP but was weighted towards lakes burnt in 1995 and produced larger prediction errors for lakes burnt between 1961 and 85. A larger range in time-since-disturbance data is required to test the hypothesis that attenuating impacts of fire on watershed nutrient export through time is decays exponentially.

Increase in the flux of divalent cations and potassium from organic soils usually occurs after fire (Tiedmann *et al.* 1978). Increased flux rates of divalent ions from burned peat likely occurred in the Caribou Mountains. However, the cation exchange capacity of both living and dead peat would remain intact and liberate protons as cations were exchanged (Clymo 1984). The result was a 7-fold median increase in [H^+], decreased pH and reduced alkalinity in burnt lakes while base cation concentrations did not change. Organic acids associated with the elevated DOC in burn-impacted lakes added to overall acidification while contributing organic anions (presumably $R\text{–}COO^-$).

General affects of fire on in-lake processes

Catchments in the Caribou Mountains are fundamentally different from catchments where fire effects have historically been studied. The large proportion of inundated peatlands in the Caribou Mountains created a unique nutrient responses following fire. For example, forest fire usually augments nitrate export and to a lesser degree, the export of phosphorus from granitic regions (Lewis 1974, McColl & Grigal 1975, Wright 1976). The relatively severe Caribou Mountains fire caused mean 3-fold higher total phosphorus and 1.5-fold higher total nitrogen concentrations which suggests higher phosphorus compared to nitrogen export. Similar findings are reported from wetland systems. Burnt fens exported phosphorus and retained nitrogen while burnt upland catchments exported nitrogen and held phosphorus in the granitic Experimental Lakes Area (Bailey *et al.* 1992). Nitrogen retention in Alberta peatlands typically exceeds 98% (Li & Vitt 1997), a value comparable to the >85% retention reported by Bailey *et al.* (1992) for burnt fens. Surface water in northern Alberta, where peatlands dominate, could be more sensitive to eutrophication from elevated TP following fire than indicated by previous fire studies if phosphorus remains the factor limiting algal biomass.

The relationship between CHL and TP in reference lakes on the plateau indicates a strong association between algal biomass and total phosphorus. The slope for $\log_{10}$-transformed values (2.31) was more positive and the intercept (-2.46) more negative than values reported in temperate locations throughout North America (*e.g.* Dillon & Rigler 1974; Prepas & Trew 1983; Hoyer & Jones 1983). The low intercept for Caribou lakes indicates low phosphorus availability for algal growth at reduced TP concentration compared to lakes in other regions. Algae in Caribou Mountain reference lakes made efficient use of TP as concentrations increased resulting in the 2.31 CHL-TP slope. In the undisturbed condition, these lakes are phosphorus limited and should respond to phosphorus loading with elevated CHL up to at least a concentration of 25 $\mu g \bullet L^{-1}$ [CHL].

Fire removed the phosphorus dependance of algal biomass in Caribou Mountain lakes by reducing nitrogen : phosphorus ratios and increasing light extinction. The relationships between residuals of predicted and observed CHL values with light extinction, transparency, colour and TN:TP indicated that the algal assemblage in burnt lakes shifted from P-limitation to both light and nitrogen limitation post-fire. This finding was further supported by nutrient stimulation experiments (NSE). The switch to nitrogen limitation from phosphorus in systems receiving nutrient loading at low N:P ratios is well documented (*e.g.* Forsberg & Ryding 1980). Nitrogen limitation allows n-fixing phytoplankton to take advantage of increased TP inputs (Barica et al. 1990), however, increased algal biomass was not observed in Caribou Mountain lakes because of the added influence of light limitation.

Our results demonstrate forest fire had a profound impact on surface water quality in lakes of the Caribou Mountains. These lakes responded to fire with elevated nutrient concentrations because phosphorus and to a lesser degree, nitrogen liberated during fire and subsequent decomposition was not retained by peat. Though base cations were likely liberated from burnt material, cation exchange with peat resulted in an increase in flux of protons. We have attempted to explain the linkage between lake nutrient concentrations and export from the catchment using an effective area approach. The simple model was based on ^{18}O and Na^+ mass-balance as a surrogate for detailed hydrologic measurements. As a surrogate, ^{18}O predicted water yields similar to that measured in one gauged watershed (MS2) and resulting effective areas were strong predictors of phosphorus concentrations among lakes. Indeed, ^{18}O and deuterium may be the only practical method of acquiring inferences of hydrology in large peatland dominated catchments. Elevated phosphorus concentrations slowly returned to reference conditions depending on both the magnitude of disturbance (%disturbance) and time-since disturbance. Phosphorus concentrations decreased linearly with %disturbance and on a $\log_e$ scale with time. A single disturbance index combining %disturbance and time proved effective, however, a larger range in time is needed to test the hypothesized exponential and linear components of this model.

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Table 1: Physical parameters for Caribou Mountain study lakes. Abreviations are: watershed area (Wo), lake surface area (Ao), lake mean depth (Zm). Slope is an average from 10-30 transects in each catchment, Upland and Peatland are percent of vegetation type covering the drainage basinarea. Disturb. Is the percent of drainage basin impacted by forest fire.

ID	Wo (ha)	Ao (ha)	Volume (10^3 m^3)	Zm (m)	Wo Slope (%)	Upland (% Ao)	Peatland (% Ao)	Disturb. (% Ao)	Colour Mg$\cdot$L^{-1} Pt
Reference lakes									
C1	663.2	52.3	510	1.07	1.5	1	94	0	317
C2	614.8	93.4	1910	2.17	2.3	2	95	0	209
C5	1068	76.5	568	0.75	2.4	6	79	0	172
C6	260.4	91.1	261	0.30	3.1	16	79	0	76
C7	569.0	90.0	578	0.66	2.9	6	87	0	205
C8	8208	60.8	612	0.99	1.4	5	85	0	389
C9	146.3	86.1	1563	1.83	2.6	13	86	0	34
C11	204.8	134.4	4071	3.06	6.1	65	35	0	26
C30	3218	536.1	55564	11.60	2.0	28	67	0	105
C31	11519	1334	105386	7.90	2.5	20	74	0	107
C32	3018	953.4	27318	2.90	3.0	12	81	0	77
C34	5571	166.5	5840	3.50	4.0	22	77	0	140
C35W	825.0	175.0	18240	5.50	6.1	32	66	0	112
C35E	832.0	268.0	24365	5.50	6.0	35	61	0	31
C40	11570	1428	101388	7.10	3.2	11	85	0	186
C45	4627	1173	93840	8.00	4.0	14	83	0	55
C47	2732	59.0	697	1.18	6.0	72	25	0	196
Lakes in catchments burnt in 1995									
C17	700.9	164.8	2300	1.54	1.9	83	16	80	240
C23	1053	65.8	2436	4.10	1.9	19	80	90	331
C24	1761	159.8	10332	6.36	3.6	35	62	95	250
C25	820.1	151.9	4278	2.87	4.2	25	72	90	112
C26	1520	36.6	523	1.68	3.3	25	73	75	480
C27	581.1	28.1	220	0.85	6.0	30	69	60	396
C41	32.6	2.6	22	0.86	2.0	43	57	100	424
C42	34.3	7.4	53	0.72	3.0	51	49	98	212
C43	691.9	8.3	50	0.60	4.0	51	45	95	456
C44	133.9	10.0	22	0.22	3.0	13	80	95	537
C46	359.4	4.4	25	0.58	6.0	44	51	60	523
Lakes in catchments burnt between 1961 & 1985									
C12	642.9	40.5	440	1.22	2.9	6	88	20	312
C13	483.8	175.5	1745	1.04	1.3	12	81	80	112
C14	268.9	62.4	1779	2.93	4.3	27	73	80	87
C15	2257	128.3	2013	1.67	2.8	6	91	70	454
C16	995.1	53.4	1056	2.04	4.2	42	55	80	344

Table 2: Summary of chemical characteristics for Caribou Mountain study lakes. Mean, standard error of the mean (*SE*), minimum (Min.), maximum (Max.), and median (Med.) values are presented for each lake grouping identified in Table 1. Chemical characteristics are: Total phosphorus (TP), dissolved phosphorus (DP), soluble reactive phosphorus (SRP), total nitrogen (TN), dissolved nitrogen (DN), dissolved inorganic nitrogen (DIN), total suspended chlorophyll *a* (CHL), calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^{+}), potassium (K^{+}), sulfate (SO_4^{2-}), chloride (Cl^{-}), dissolved organic carbon (DOC) and colour.

	Reference Lakes (*n* =17)					Burn-impacted Lakes (*n* =11)					1961-85 Burn-impacted lakes (*n* = 5)				
	Mean	SE	Min.	Max.	Med.	Mean	SE	Min.	Max.	Med.	Mean	SE	Min.	Max.	Med.
TP (µg•L^{-1})	32	2	22	46	32	92	13	40	166	79	53	9	36	80	46
DP (µg•L^{-1})	14	1	9	25	12	50	7	21	97	47	30	10	14	64	19
SRP(µg•L^{-1})	5	1	1	14	5	27	4	4	52	31	18	9	2	51	12
TN (µg•L^{-1})	640	32	412	840	661	825	78	552	1480	757	742	62	588	907	799
DN (µg•L^{-1})	480	25	345	723	470	632	46	421	974	636	581	46	456	728	556
DIN(µg•L^{-1})	17	2	8	43	15	52	23	15	278	28	73	17	29	119	75
CHL(µg•L^{-1})	11	2	4	25	9	13	3	4	39	11	12	2	5	20	12
Ca^{2+} (mg·L^{-1})	10.3	1.5	4.1	27.0	8.5	7.6	0.7	5.3	12.1	6.5	8.4	1.5	5.7	14.3	7.4
Mg^{2+} (mg·L^{-1})	2.4	0.4	0.9	7.7	2.0	1.9	0.2	1.3	3.1	1.8	2.1	0.4	1.5	3.8	1.9
Na^{+} (mg·L^{-1})	1.3	0.4	0.3	6.0	0.6	1.0	0.2	0.5	2.8	0.8	0.9	0.2	0.5	1.3	0.8
K^{+} (mg·L^{-1})	0.4	0.0	0.1	0.6	0.4	0.7	0.1	0.2	1.1	0.6	0.3	0.1	0.1	0.6	0.3
Total alkalinity (mg·L^{-1} $CaCO_3$)	33.2	4.7	9.8	71.8	35.3	15.6	2.7	1.1	27.8	16.0	24.1	8.2	10.0	54.8	16.3
SO_4^{2-} (mg·L^{-1})	5.8	2.7	0.7	48.7	2.1	6.1	1.4	1.6	16.6	5.4	3.3	0.9	1.2	5.3	3.5
Cl^{-} (mg·L^{-1})	0.8	0.0	0.1	0.7	0.1	0.1	0.0	0.1	0.2	0.1	0.1	0.0	0.1	0.2	0.1
DOC (mg·L^{-1})	16.4	1.3	10.0	27.0	16.0	25.5	2.1	14.0	35.0	27.0	20.6	3.2	12.0	27.0	23.0
Colour (mg·L^{-1}) Pt	143	24	26	389	112	360	42	112	537	396	262	70	87	454	312

Table 3: Correlation analysis between nutrient concentrations, percent disturbance and time-since-disturbance in reference, burnt and previously burnt lakes (n=33).

	% Dist.	Ln Time (yr)
Log SRP	0.70 **	-0.65 **
Log DP	0.81 **	-0.77 **
LogTP	0.77 **	-0.78 **
Log DN	0.52 *	-0.50 *
LogTN	0.41^{1}	-0.44^{1}

** $P < 0.0001$; * $P < 0.01$; 1, $P = 0.01$

Table 4: Summary of univariate regression model of Log_{10} total and dissolved phosphorus (TP & DP) on disturbance index (DI - eq. 3).

X	Y	*n*	r^2	Standard error of slope	MS error	*F*	*P*
DI	Log_{10}TP	33	0.58	0.001	0.025	43.34	<0.0001
DI	Log_{10}DP	33	0.58	0.001	0.039	43.19	<0.0001

Fig. 1: Location of Caribou Mountain study lakes. Reference sites contain lakes with no fire in their catchments since 1953, previously burnt sites contain lakes with catchments burnt between 1961 and 1985, burnt sites contain lakes in catchments burnt in 1995.

Fig. 2: Summer mean total base cations and cation composition for lake waters in reference (n=17) and burnt (n=11) catchments. Letters (a & b) indicate differences between reference and burn-impacted lakes ($P < 0.05$).

Fig. 3: Summer mean pH and total alkalinity for reference (n=17) and burn-impacted lakes (n=11). Letters (a & b) signify difference in means at $P < 0.05$.

Fig. 4: Summer mean concentrations of total phosphorus (TP), dissolved phosphorus (DP), soluble reactive phosphorus (SRP), and particulate phosphorus (PP) for reference (n = 17), burn (n = 11) and previously burn-impacted (n = 5) lakes. Letters (a&b) indicate difference between reference and burnt means at $P \leq 0.05$. Analytical statistics were not applied to previously burnt lakes.

Fig. 5: Summer mean total nitrogen (TN), dissolved nitrogen (DN) and particulate nitrogen (PN) for reference (n= 17) and burnt (n=11) lakes. Nitrate-nitrite and ammonium are log transformed. Letters (a & b) indicate differences in mean concentrations between reference and burnt at P 0.01.

Fig. 6: Summer mean nitrate-nitrite (NO_3^{-}N) and ammonium (NH_4^{+}-N) concentrations for reference (n= 17) and burn-impacted (n=11) lakes. Values are log10 transformed. Letters (a & b) indicate differences in mean concentrations between reference and burnt at P=0.01.

Fig. 7: The relationship between total chlorophyll *a* (CHL) and total phosphorus (TP) for lakes in reference and burnt catchments. Values are $\log_{10}$ transformed, A) equation for reference lakes only. B) Equation without C8 which deviates from other reference lakes because of its high colour (388 mg$\cdot$L^{-1} Pt).

Fig. 8: CHL response for nutrient stimulation experiments (10L microcosms) in two lakes. Response is final CHL divided by initial CHL and thus represents growth in response to nutrient amendments and light. The treatments are C= control, N= +1000 µg/L nitrogen, P = +100 µg/L phosphorus and NP = combined N & P. Response in controls is due to improved light conditions in the microcosms relative to the lake. Letters indicate significant differences in response ($P \leq 0.001$).

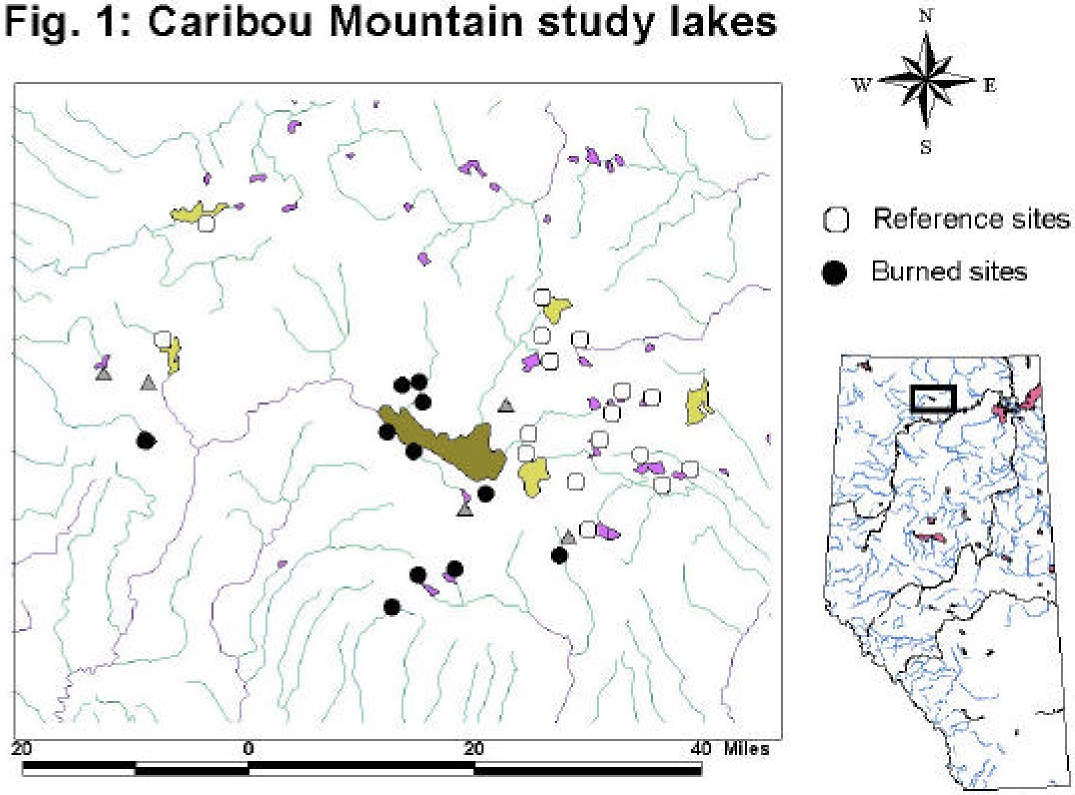

Fig. 1: Caribou Mountain study lakes

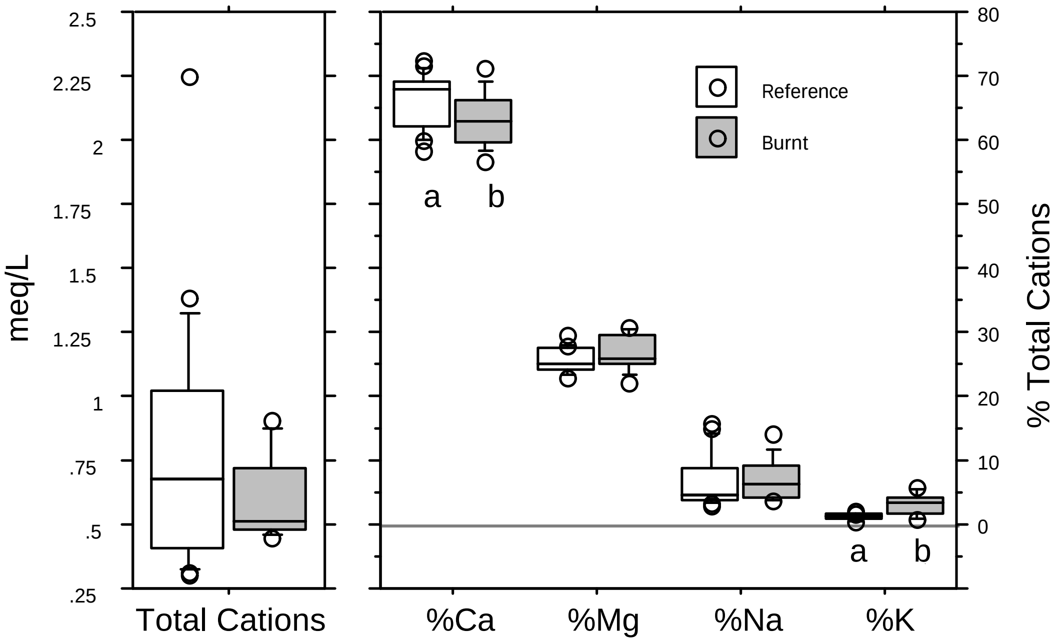

Fig. 2

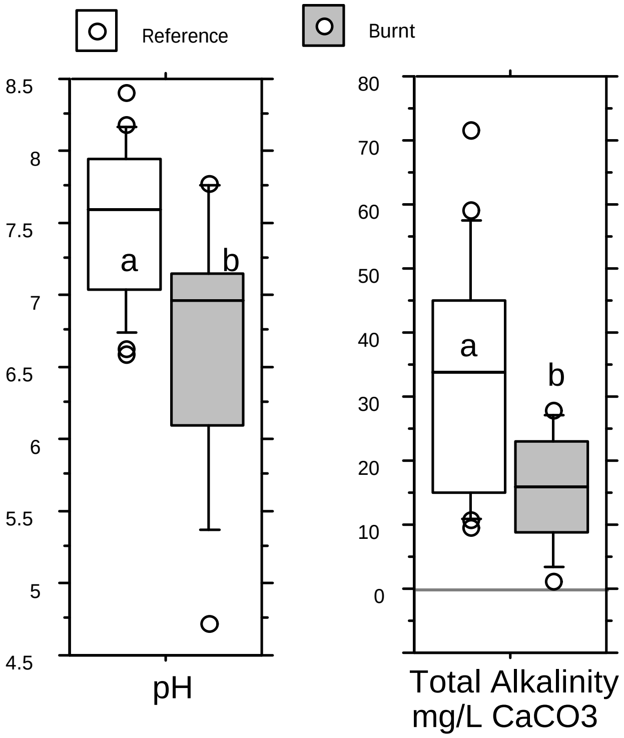
Reference
Burnt
8.5
8
7.5
7
6.5
6
5.5
5
4.5
a
b
pH
80
70
60
50
40
30
20
10
0
a
b
Total Alkalinity
mg/L CaCO3

Fig. 3

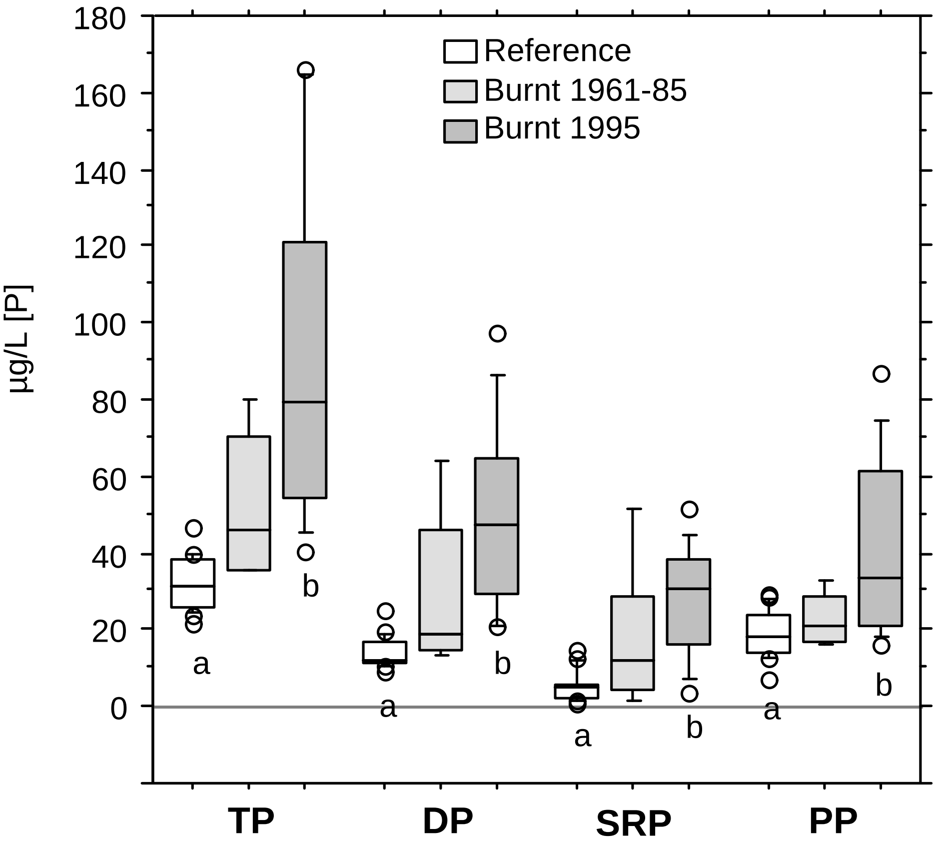
180
160
140
120
100
80
60
40
20
0
µg/L [P]
Reference
Burnt 1961-85
Burnt 1995
a
b
a
b
a
b
a
b
TP
DP
SRP
PP

Fig. 4

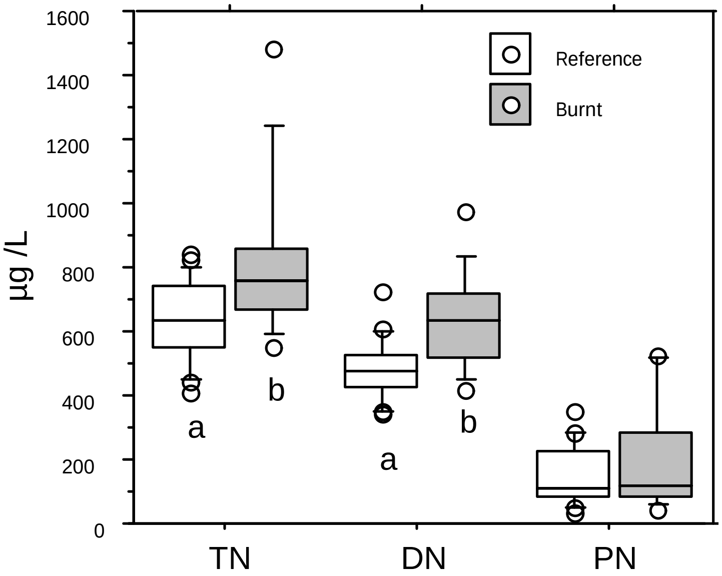
1600
1400
1200
1000
800
600
400
200
0
µg /L
Reference
Burnt
a
b
a
b
TN
DN
PN

Fig. 5

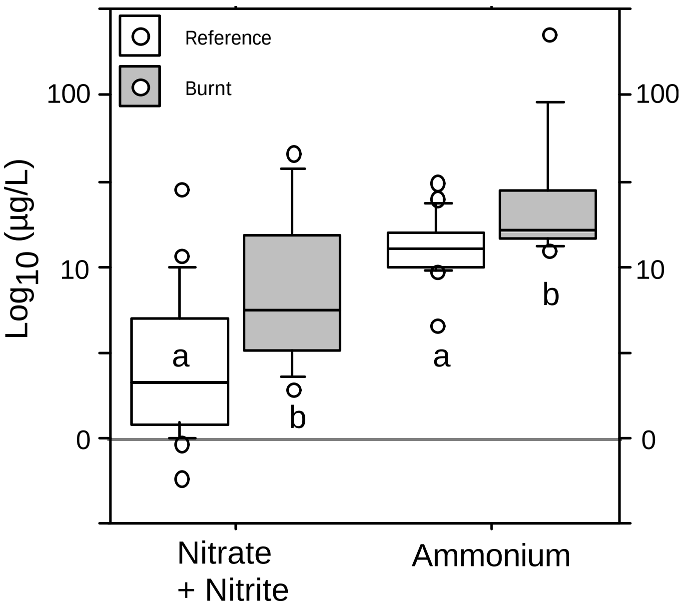
Reference
Burnt
100
10
0
Log10 (µg/L)
a
b
a
b
Nitrate
+ Nitrite
Ammonium

Fig. 6

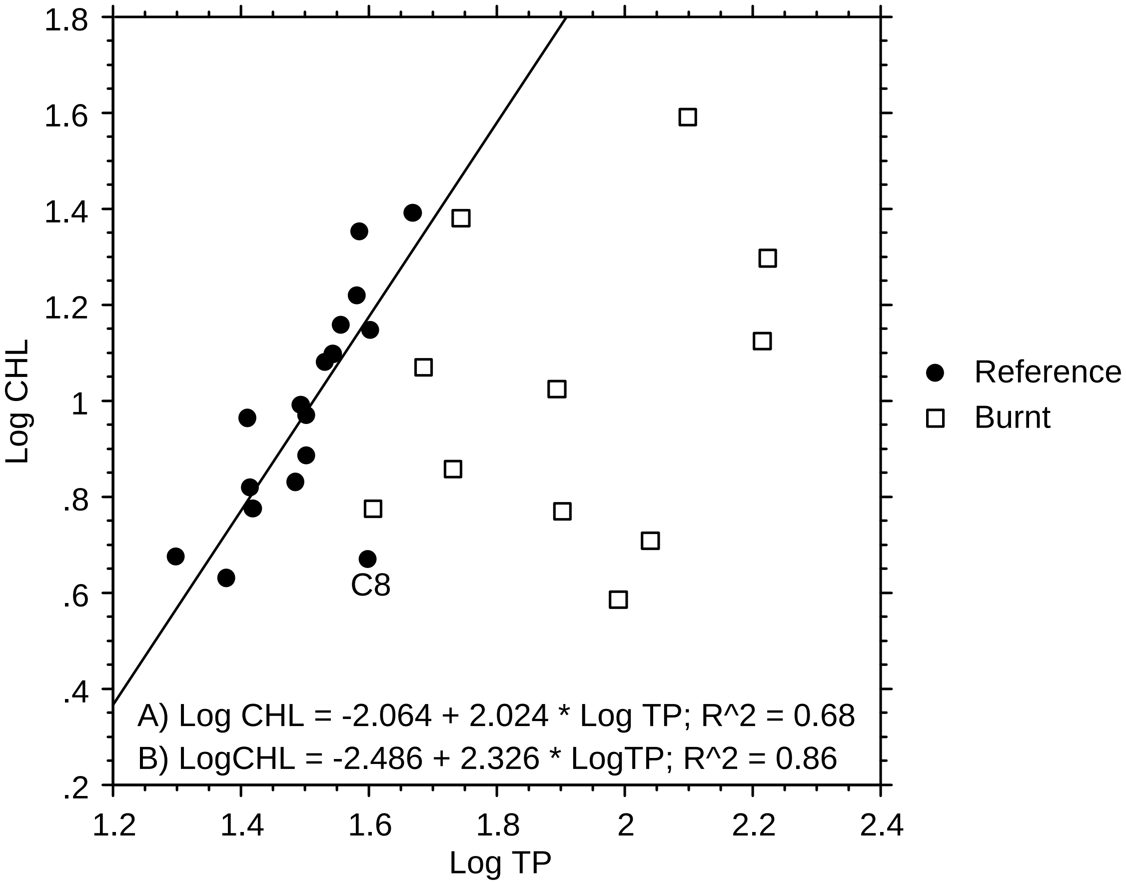
1.8
1.6
1.4
1.2
1
.8
.6
.4
.2
Log CHL
1.2
1.4
1.6
1.8
2
2.2
2.4
Log TP
C8
● Reference
☐ Burnt
A) Log CHL = -2.064 + 2.024 * Log TP; R^2 = 0.68
B) LogCHL = -2.486 + 2.326 * LogTP; R^2 = 0.86

Fig. 7

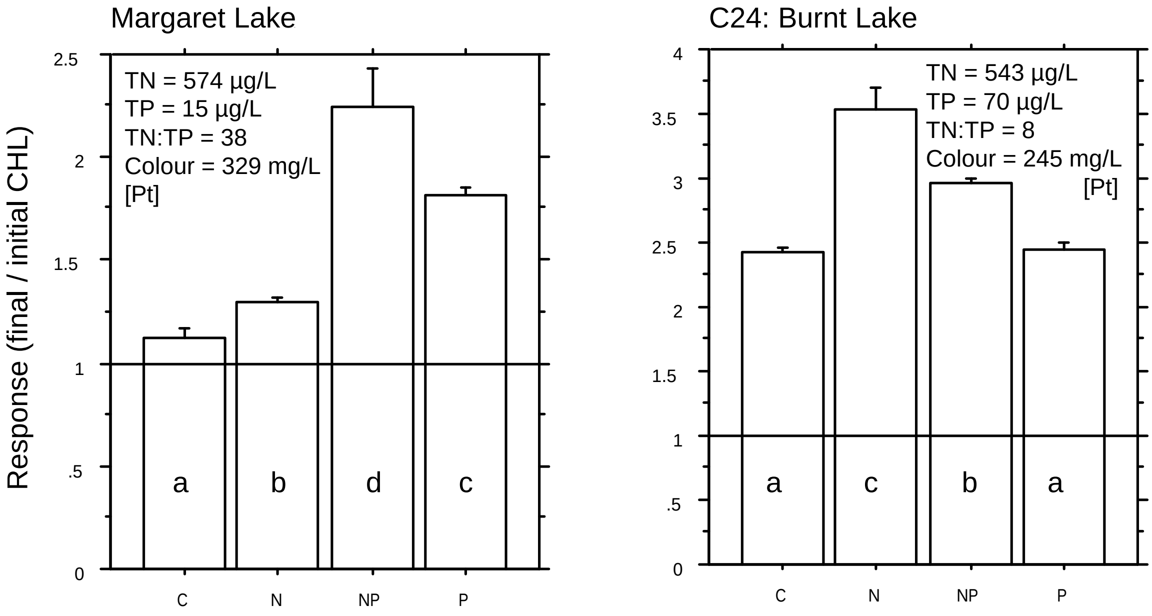
Margaret Lake
Response (final / initial CHL)
2.5
2
1.5
1
.5
0
TN = 574 µg/L
TP = 15 µg/L
TN:TP = 38
Colour = 329 mg/L
[Pt]
a
b
d
c
C
N
NP
P
C24: Burnt Lake
4
3.5
3
2.5
2
1.5
1
.5
0
TN = 543 µg/L
TP = 70 µg/L
TN:TP = 8
Colour = 245 mg/L
[Pt]
a
c
b
a
C
N
NP
P

Fig. 8